UNIVERSITÉ DU QUÉBEC À MONTRÉAL

CHEMINS, COUPES ET PLANTATIONS : UNE EXPLORATION DES STRATEGIES D'AMENAGEMENT FORESTIER A L'ECHELLE DU PAYSAGE

THÈSE

PRÉSENTÉ(E)

COMME EXIGENCE PARTIELLE

AU DOCTORAT EN BIOLOGIE

PAR

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MAI 2024

AVANT-PROPOS ET REMERCIEMENTS

Depuis ma lecture fatidique d'une offre de thèse il y a maintenant 6 ans, j'ai troqué les montagnes aiguisées des Pyrénées pour les flancs paisibles du Mont-Royal ; le bruit constant du Gave d'Azun et de ses torrents pour ceux de la rue Notre-Dame et du port de Montréal ; la compagnie des chèvres et des martres pour celles des humains et des écureuils; le vieux continent pour le nouveau. Mais il n'y a pas que le décor qui s'est transformé. En 6 ans de thèse, je ne suis plus le même quand je me regarde dans le miroir. Je n'ai plus ce regard pétillant, remplis de cette certitude candide que je pourrais changer le monde à force de le vouloir. Mon regard est usé comme des chaussures dans lesquelles on a trop marché. Usé par les écrans, les manuscrits, les articles, par d'incalculables lignes de code; mais aussi par les tourments de mon époque, répétés en milliers de titres affamés d'attention sur le vaste océan du web. Usé par les injures et le rejet vécu en défendant les droits des animaux en parallèle de mes recherches, au sein d'une société dont certaines parties s'éveillent douloureusement à leurs conséquences. Usé de vouloir tout faire, d'aider tout le monde, mais d'arriver à si peu en retour.

En 6 ans de thèse, le monde m'a donné une grande leçon : ça n'est pas si simple de l'aider ou de le comprendre. Et ça n'est pas suffisant de juste le vouloir. C'est un travail interminable et incertain. Un rocher pour lequel Sisyphe à toutes les chances de s'être trompé de montagne en arrivant en haut. Combien d'entre nous, pourtant si bien intentionnés·es, en sont en effet venus·es à défendre l'indéfendable ? A justifer l'injustifiable ? A devenir un problème de plus, là ou l'on voulait en faire un de moins ? Mais «Il faut imaginer Sisyphe heureux», écrivait Camu. «Je vois cet homme redescendre d'un pas lourd mais égal vers le tourment dont il ne connaîtra pas la fin». C'est pour cela que malgré tout, mes yeux restent grands ouverts, et mon visage souriant. Et maintenant que mon rocher est arrivé en haut de la montagne qu'est cette thèse, et avant qu'il ne retombe sur une autre pente, il est temps de remercier toutes les personnes qui m'ont aidé à en endurer l'ascension.

Je remercie ainsi mes doux parents et leur encouragements infatiguables ; mon frêre et toute sa petite famille que je vis grandir avec passion de l'autre coté de l'océan ; et le reste de ma famille, pour leur bienveillance qui surpasse la distance qui nous sépare. Je remercie les amis·es de France qui sont restés en contact avec moi outre-atlantique, et qui m'ont aidé à prendre le temps de faire mon nid au Québec sans me sentir trop seul. Je remercie en particulier Nicolas, l'une des seules personnes au monde avec qui

je me sens à l'aise de piloter un sous-marin dans les mers souterraines d'Europa en rentrant du travail; et Julia, avec qui mes longues discussions ne cessent d'être un baume rafraichissant sur mon cœur.

Je remercie tous les collègues qui m'ont accuillis à la TELUQ; en particulier Alexandre et Rim, dont je n'oublierais jamais le marriage dans la belle erablière d'été et l'annonce de la naissance de Liv; ainsi que Simon L.D., dont j'attends avec impatience sa suite des aventures de Thufir, ainsi que celles de sa fille Bénédicte. À nous quatres, nous aurons exploré les tréfonds de la philosophie de Ludwig Wiggenstein, des mystères de la philonite, de *Cosmo* et *Rebirth* et de la planète Myriade pendant de longues années d'amitié qui continuent encore aujourd'hui. Je remercie également Simon B., qui m'apris à me défendre à l'épée longue allemande sans jamais perdre sa présence tranquille. Des collègues de l'UQAM, je remercie Daniel, Ananda, Michael, Gauthier, Johanna, Élise, Jon, Oriane et tous les autres qui m'ont soutenus durant toute cette thèse. Mes souvenirs de nos voyages au Vermont ou de nos différentes conférences à Québec ou à Chicoutimi me resterons à jamais en mémoire. Je remercie aussi mes collègues du Ministère des Forêts : Jérôme, Isabelle et Benjamin en particulier, pour m'avoir fait passer autant de bons moments dans la machine administrative de l'Atrium et en dehors. Je réserve également beaucoup de gratitude pour Élise Filotas et Christian Messier, dont la bienveillance et les connaissances me font constamment réaliser la chance incroyable que j'ai eu de les avoir comme directeurs·ices de thèse. Je n'oublie pas non plus Yan Boulanger, Dominic Cyr et Osvaldo Valeria qui n'ont pas hésité à partager données, sagesse et humour lors de mon parcours, et sans qui mes recherches n'auraient certainement pas eu la même qualité.

Je remercie également mes camarades d'activisme, avec qui j'ai partagé de longues heures dans la rue chaque fin de semaine pendant cette thèse durant des marches, des manifestations, ou différents types d'évenements de sensibilisation. Je ne pourrais en faire toute la liste; mais je nommerais Tina, Nathe, Laure, Isabelle, JF, Manu, Felix, Gaelan, Sorin, Marie-Pier, Alexia et Nolwenn. Au dela des moqueries, des insultes, du mépris, de la désinformation et des critiques, vous êtes – bien qu'imparfaits·es – parmis les personnes plus braves et bienveillantes que j'ai pu rencontrer. Je remercie également Natalie, dont l'affection m'as soutenu durant mes moments les plus douloureux, et dont le courage ne cesse de m'inspirer depuis que j'ai rencontrée. Enfin, je remercie Kelly et Stelah pour avoir illuminé mon quotidien durant nos années de vies communes, et pour m'avoir forcé à grandir afin de prendre soin de deux personnes qui le méritaient amplement. Vous garderez à jamais un morceau de mon cœur qui vous appartient à présent, et je ne cesserais jamais de vouloir vous voir sourire.

DÉDICACE

À tous les êtres humains ou animaux
exploité, blessés ou tués quotidiennement pour satisfaire
nos envies et nos peurs plutôt que nos besoins.

À toutes les personnes qui se battent courageusement pour
les droits de ces victimes malgré le dénis, le mépris et
l'injustice auxquelles elles font face.

À mon père, qui aurait certainement apprécié être là le jour
de ma soutenance, mais dont la mémoire aura été à mes
coté tout du long.

Aux forêts du Québec et leurs mystères.

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AAC : Allowable Annual Cut ou Volume Récoltable Annuel

ANPP : *Annual Net Primary Productivity* ou Productivité Primaire Annuelle Net.

AQRESEAU+ : Base de données des routes et chemins du Québec, crée par le Ministère des Transports du Québec.

BAU : *Business As Usual*. Type de scénario ou les activités humaines sont similaires aux activités actuelles.

C# : Language de programmation dévellopé par Microsoft.

CC : *Clearcut* ou coupe à blanc.

Clumpy : Indice de fragmentation du paysage.

CPI-CP : Coupe Progressive Irrégullère à Couvert Permanent.

CWM : *Community Weighted Mean*. Moyenne de la valeur d'un trait fonctionnel entre différentes espèces ou individus, pondérée par l'abondance ou la masse des espèces ou des individus respectivement.

DEM : *Digital Elevation Model* ou Modèle d'Élévation Digital. Carte raster qui contient des valeurs d'élévation (par rapport au niveau de la mer) pour tout les pixels de la carte.

FAO : *Food and Agriculture Organisation* des Nations Unies

FLM : *Forest Landscape Model*

FOSS : *Free and Open Source Software* ou Logiciel Gratuit et Ouvert

FRS : *Forest Roads Simulation*. Extension dévellopée pour le modèle LANDIS-II au cours de cette thèse.

GES : Gaz à Effet de Serre

GIEC : Groupe International d'Experts sur le Climat

GitHub : Site internet permettant de créé des « dépots » de données ou de codes sources privés ou public et versionnables via le logiciel Git.

k-NN : *K-Nearest Neighbors algorithm* ou méthode des k plus proches voisins. Méthode mathématique d'assignation d'une entité à une autre basée sur des caractéristiques communes aux deux entités.

LANDIS-II : FLM ouvert et gratuit qui propose différentes extensions, permettant un environnement de modélisation customisable pour les utilisateurs·ices.

MERN : Ministère de l'Énergie et des Ressources Naturelles du Québec. A été remplacé par le MRNF en 2022.

MFFP : Ministère des Forêts, de la Faune et des Parcs du Québec. A été remplacé par le MRNF en 2022.

MPB : *Mountain Pine Beetle*, ou dendrochtone du pin (*Dendroctonus ponderosae*)

MRNF : Ministère des Ressources Naturelles et des Forêts du Québec actuel.

NFI : *National Forest Inventory* ou Inventaire Forestier National du Canada.

NNSA : *Nearest Neighbour Spectral Analysis* ou Analyse Spectrale des Plus Proches Voisins. Méthode mathématique d'assignation d'une entité spatiale à une autre basée sur un ensemble de caractéristiques communes aux deux entités.

MST : *Minimum Spanning Tree*. Problème d'optimisation mathématique dans lequel on tente d'accéder à toutes les cibles possibles au sein d'un graphe avec un cout minimal.

MTAP : *Multiple Target Access Problem*. Problème d'optimisation mathémathique dans lequel on tente d'accéder différentes cibles au sein d'un graphe.

PAAB : *Percentage of annual Area Burned* ou Pourcentage d'aire brulée annuellement.

PAFRAC : *Perimeter-Area Fractal Dimension Index*. Indice de fragmentation du paysage.

PIB : Produit Intérieur Brut.

PICUS : Modèle physiologique permettant de simuler la croissance et la mortalité d'arbres au sein d'un peuplement pour un ensemble de conditions environnementales (p.ex. sol, climat) données.

RAM : *Random Access Memory*. Mémoire vive utilisée dans les ordinateurs actuels.

RCP : *Representative Concentration Pathways* ou Trajectoires Représentative de Concentration. Scénarios climatiques utilisés par le GIEC.

SBW : *Spruce Budworm* ou Tordeuse des bourgeons de l'épinette (*Choristoneura fumiferana*).

SSMTSP : *Single Source Many Target Shortest Path Problem*. Problème d'optimisation mathématique dans lequel on tente d'accéder à connecter plusieurs cible au sein d'un graphe à une nœud « source » tout en minisant les couts.

SSP : *Shared Socioeconomic Pathways* ou Trajectoires Socioéconomiques Partagées. Scénarios climatiques utilisés par le GIEC.

STAP : *Single Target Access Problem*. Problème d'optimisation mathémathique dans lequel on tente d'accèder à une cible au sein d'un graphe.

TCA : *Total Core Area*. Indice de fragmentation du paysage.

TRIAD : Stratégie d'aménagement forestier à l'échelle du paysage utilisant un zonage de trois types de foresteries différentes : foresterie intensive, extensive, ou conservation.

TRIAD+ : Variation de la stratégie d'aménagement TRIAD utilisant des plantations pour augmenter la diversité fonctionnelle des peuplements forestiers traités.

UQAT : Université du Québec en Abitibi-Témiscamingue

RÉSUMÉ

L’aménagement forestier est une activité humaine dont les impacts passés et présents sur les forêts du monde sont profonds. De nombreuses interrogations persistent alors quant à comment continuer de satisfaire les besoins humains en produits du bois, tout en préservant les caractéristiques uniques des écosystèmes forestiers naturels. En conséquence, plusieurs stratégies d’aménagement ont vu le jour à travers l’histoire pour tenter de résoudre ce dilemme. La performance de ces stratégies sur différents indicateurs liés aux écosystèmes forestiers reste cependant relativement peu explorée, en particulier à l’échelle du paysage et sur le long terme. Mais la considération de ces questions à de grandes échelles spatio-temporelles devient extrêmement pressante, alors que les changements globaux (et en particulier les changements climatiques) impactent de plus en plus les forêts du monde. L’objectif de cette thèse consiste ainsi à explorer l’impact de différentes stratégies d’aménagement forestier à l’échelle du paysage, et à élaborer de nouvelles pratiques d’aménagement pour aider les forêts à s’adapter plus rapidement aux conditions climatiques, physiques et biologiques futures.

Pour ce faire, les travaux de cette thèse utilisent le *Forest Landscape Model* LANDIS-II, qui permet de simuler l’évolution des forêts d’un paysage à travers plusieurs siècles. LANDIS-II modélise la dynamique des successions végétales et des perturbations naturelles et humaines qui ont lieu dans le paysage simulé, ainsi que l’effet des changements climatiques sur la croissance des arbres ou sur certaines perturbations. Nous avons simulé un territoire de plus de 4 millions d’hectares situés dans la région de la Mauricie, au Québec, en nous basant sur les données publiques d’inventaire forestier et sur les données d’études passées. En premier lieu, nous avons développé et testé une nouvelle extension pour LANDIS-II nommée « FRS » (*Forest Roads Simulation*), afin de nous permettre de simuler la construction des chemins forestiers nécessaires aux opérations forestières. Cette nouvelle extension nous à alors ensuite permis d’explorer les impacts d’un gradient de stratégies d’aménagements allant de stratégies plus « intensives » basées sur des coupes équiennes (p. ex., des coupes totales) à des stratégies plus « extensives » basées sur des coupes inéquiennes (p. ex., des coupes partielles) à l’échelle du paysage. Plus particulièrement, nous avons pu observer l’impact de ces stratégies sur la quantité de chemins dans le paysage, ainsi que la quantité de vieilles forêts et la fragmentation de ces vieilles forêts. Enfin, nous avons développé une nouvelle stratégie d’aménagement basée sur le concept plus ancien de l’aménagement en TRIAD. Nous y avons néanmoins intégré des plantations fonctionnelles ayant pour but d’augmenter la résilience des peuplements traités à des perturbations futures incertaines influencées par les changements globaux (p. ex., feux, sécheresse, épidémies d’insectes), en augmentant leur diversité de réponses à ces perturbations. Nous avons alors comparé la performance de la TRIAD+ à d’autres stratégies avec ou sans TRIAD et avec ou sans plantations fonctionnelles à travers différents scénarios de changements climatiques. Nous avons mesuré leur effet sur la biomasse d’arbres matures dans le paysage et la diversité fonctionnelle des peuplements, ainsi que différents indicateurs de résiliences pour la biomasse mature du paysage suite à une perturbation extrême que nous avons simulés après 100 ans d’aménagement.

Nos résultats peignent un portrait complexe des impacts de l’aménagement forestier à de grandes échelles spatiales et temporelles. Ils soulignent également le besoin pressant d’explorer de nouvelles stratégies d’aménagement, ainsi que de nouvelles méthodes de modélisation de leurs impacts. Le test de notre nouvelle extension FRS pour LANDIS-II a révélé que son algorithme était capable de reproduire convenablement les principales caractéristiques de réseaux de chemins forestiers existants dans deux paysages au Québec. Cette extension gratuite et *open-source* permet alors à toute équipe de recherche d’intégrer la simulation des chemins forestiers dans des études à l’échelle du paysage dans le futur. Notre exploration d'un gradient de stratégies d'aménagements basées sur des coupes équiennes ou inéquiennes a révélé que les coupes inéquiennes permettaient de préserver plus de vieilles forêts au sein du paysage. Cependant, cette préservation se faisait au prix d'une bien plus grande quantité de chemins forestiers dans le paysage, et d'une plus grande fragmentation des vieilles forêts par ces chemins. La présence de feux de forêt plus fréquents dans le nord de notre zone simulée à réduit les différences entre scénarios, en réduisant naturellement la quantité de vieilles forêts et en augmentant leur fragmentation par des surfaces de forêts plus jeunes. Nos résultats questionnent ainsi le remplacement uniforme des coupes équiennes par des coupes inéquiennes pour des objectifs de conservation, et soulignent l'importance de considérer les régimes de perturbations naturelles dans le choix de leur utilisation. De plus, il est à noter que bien que les stratégies utilisant plus de coupes inéquiennes tendaient à conserver plus de vieilles forêts, celles-ci étaient alors régulièrement perturbées par des coupes partielles périodiques. Enfin, notre nouvelle stratégie TRIAD+ a pu réaliser un bon compromis entre conservation, production et adaptation en comparaison aux autres stratégies testées. En particulier, la TRIAD+ a permis de récolter un volume de bois similaires aux autres stratégies tout en augmentant la taille des aires protégées du paysage, et en augmentant la diversité fonctionnelle des peuplements et la résilience de leur biomasse mature. Cependant, l'augmentation de la résilience de la biomasse mature des peuplements était relativement petite, et même négligeable sur le long terme pour des questions d'aménagement. De plus, toutes nos variables étaient principalement affectées par la présence et l'intensité des changements climatiques au sein de nos simulations, qui ont beaucoup influencé la composition des espèces arbres du paysage simulé. Nos résultats questionnent alors la pertinence des plantations fonctionnelles comme outil pour aider les forêts à s'adapter aux perturbations futures amenées par les changements globaux. Ils questionnent également les limites de modèles comme LANDIS-II pour simuler les processus à l'échelle des peuplements forestiers qui influencent leur résilience, comme ceux en lien avec leur diversité fonctionnelle. Néanmoins, nos résultats démontrent la possibilité d'implémenter des objectifs d'adaptations au sein de stratégies d'aménagement existantes.

Les travaux de cette thèse ouvrent ainsi la porte à des études futures, tout en répondant à certaines questions pressantes dans le domaine de l'écologie forestière. Ils démontrent que les stratégies d'aménagement forestier futures devront certainement utiliser un mélange de différentes méthodes et technique de foresterie, afin de maximiser les compromis qui en découlent sur le long terme et à l'échelle du paysage. Ces travaux révèlent également que ces stratégies devront s'adapter aux changements globaux, et aux bouleversements inévitables qu'ils feront subir aux forêts dans le futur.

Mots clés : Aménagement forestier, aménagement inéquien, chemins forestiers, adaptation, changements climatiques, résilience

ABSTRACT

Forest management is a human activity whose past and present impacts on the world's forests are profound. Many questions therefore persist as to how to continue to satisfy human needs for wood products, while preserving the unique characteristics of natural forest ecosystems. As a result, a number of management strategies have emerged throughout history in an attempt to resolve this dilemma. However, the performance of these strategies on various forest ecosystem indicators remains relatively unexplored, particularly at the landscape scale and over the long term. But consideration of these issues at large spatio-temporal scales is becoming extremely pressing, as global changes (and in particular climate change) increasingly impact the world's forests. The aim of this thesis is therefore to explore the impact of different forest management strategies at the landscape scale, and to develop new management practices to help forests adapt more rapidly to their future climatic, physical and biological conditions.

To achieve this, we used the Forest Landscape Model LANDIS-II, which simulates the evolution of forests in a landscape over several centuries. LANDIS-II models the dynamics of tree succession and natural or human disturbances taking place in the simulated landscape, as well as the effect of climate change on tree growth or certain disturbances. We simulated an area of over 4 million hectares in the Mauricie region of Quebec, based on public forest inventory data and data from past studies. First, we developed and tested a new extension for LANDIS-II called "FRS" (Forest Roads Simulation), to enable us to simulate the construction of forest roads required for forestry operations. This new extension then enabled us to explore the impacts of a gradient of management strategies ranging from more "intensive" strategies based on even-aged cuts (e.g., clearcuts) to more "extensive" strategies based on uneven-aged cuts (e.g., partial cuts) at the landscape scale. In particular, we were able to observe the impact of these strategies on the quantity of roads in the landscape, as well as the quantity of old-growth forest and the fragmentation of this old-growth forest. Finally, we developed a new management strategy based on the older concept of TRIAD management. We have, however, incorporated functional plantings aimed at increasing the resilience of treated stands to uncertain future disturbances influenced by global change (e.g. fire, drought, insect epidemics), by increasing their diversity of responses to these disturbances. We then compared the performance of TRIAD+ to other strategies with or without TRIAD and with or without functional plantations under different climate change scenarios. We measured their effect on the biomass of mature trees in the landscape and the functional diversity of stands, as well as different indicators of resilience for the mature biomass of the landscape following a extreme disturbance that we simulated after 100 years of management.

Our results paint a complex picture of the impacts of forest management at large spatial and temporal scales. They also highlight the pressing need to explore new management strategies, as well as new methods for modeling their impacts. Testing of our new FRS extension for LANDIS-II revealed that its algorithm was able to adequately reproduce the main characteristics of existing forest road networks in two Quebec landscapes. This free, open-source extension now enables any research team to integrate forest road simulation into landscape-scale studies in the future. Our exploration of a gradient of management strategies based on even-aged and uneven-aged cuts revealed that uneven-aged cuts preserved more old-growth forest within the landscape. However, this preservation came at the cost of a much greater number of forest roads in the landscape, and greater fragmentation of old-growth forests by these roads. The presence of more frequent forest fires in the north of our simulated area reduced the differences between scenarios, naturally reducing the amount of old-growth forest and increasing its fragmentation by younger forest areas. Our results thus question the uniform replacement of even-aged cuts by uneven-aged cuts for conservation purposes, and underline the importance of considering natural disturbance regimes when considering their use. In addition, it should be noted that although strategies using more uneven-aged cuts tended to conserve more old-growth forests, these were then regularly disturbed by periodic partial cuts. Finally, our new TRIAD+ strategy was able to achieve a good compromise between conservation, production and adaptation compared with the other strategies tested. In particular, TRIAD+ was able to harvest a similar volume of wood to the other strategies, while increasing the size of the protected areas in the landscape, and increasing the functional diversity of the stands and the resilience of their mature biomass. However, the increase in stand resilience of mature biomass was relatively small, and even negligible over the long term for management purposes. Furthermore, all our variables were mainly affected by the presence and intensity of climate change within our simulations, which greatly influenced the tree species composition of the simulated landscape. Our results thus question the relevance of functional plantations as a tool to help forests adapt to future disturbances brought about by global change. They also question the limitations of models such as LANDIS-II to simulate stand-level processes that influence resilience, such as those related to functional diversity. Nevertheless, our results demonstrate the possibility of implementing adaptation objectives within existing management strategies.

The work in this thesis thus opens the door to future studies, while answering some pressing questions in the field of forest ecology. It shows that future forest management strategies will certainly need to use a mix of different forestry methods and techniques, in order to maximize the resulting trade-offs over the long term and at the landscape scale. This work also shows that these strategies will need to adapt to global changes, and to the inevitable upheavals they will bring to forests in the future.

Keywords : Forest management, uneven-aged management, forest roads, adaptation, climate change, resilience

INTRODUCTION

0.1 L'aménagement forestier et son importance

Environs 12 000 ans avant l'écriture de ces mots, un changement profond se produit au sein des forêts de la période du Néolithique. En attachant des silex coupants à des poignées faites avec des branches, l'espèce que l'on appellera Homo sapiens développe la première hache (Ennos, 2021). Les arbres, la forêt et le bois qu'ils fournissent devinrent ainsi une partie plus essentielle que jamais de l'histoire de notre espèce; et en retour, Homo sapiens devint également une partie essentielle de la leur par le pouvoir de la hache et du feu. Beaucoup de choses ont changé depuis, alors que nos haches rudimentaires se virent remplacées par des véhicules massifs équipés de systèmes GPS, de chauffage interne et de communications radio. Mais ces avancées technologiques indiquent que même si Homo sapiens à entre temps découvert le béton, le plastique, l'acier et les fibres de carbone, notre espèce continue d'entretenir une relation privilégiée avec le bois en tant que matière première.

Entre notre passé lointain et aujourd'hui se trouve le chemin tortueux menant à l'apparition de la foresterie, définie comme la pratique (et ensuite comme la science) de la gestion et de l'utilisation des forêts (Fernow, 1909). On y retrouve ainsi toute une gamme de techniques : l'afforestation, la déforestation, la protection, la récolte d'arbres de différents âges ou tailles, la gestion de la régénération après les coupes, les plantations, les pratiques d'éclaircissement ou de jardinage, etc. (Doucet et Côte, 2009). Aujourd'hui au Québec, on utilise le terme « foresterie » de manière souvent interchangeable avec celui d'« aménagement forestier », qui est défini comme l'application de la foresterie à des aires données (Proulx *et al.*, 2000). Néanmoins, on distingue ces deux termes de celui de « sylviculture », qui concerne alors les interventions humaines au sein des peuplements forestiers (Proulx *et al.*, 2000). Ainsi, là où la foresterie et l'aménagement forestier se font à l'échelle du paysage, la sylviculture s'applique plutôt à l'échelle du peuplement.

Au début de son histoire, la foresterie était loin d'être un champ de connaissances aussi unifié et défini qu'aujourd'hui. Avant la renaissance, elle se basait principalement sur des techniques perpétuées par tradition orale, ou par des informations provenant de textes romains ou grecs dans le monde occidental (Agnoletti, 2006). C'est principalement durant le siècle des Lumières que l'ensemble des savoirs traditionnels liés à la foresterie se voit rassemblé et compilé en Europe. Ce processus mènera au développement de la foresterie en tant que science et à sa rapide progression dans les siècles qui suivent

(Agnoletti, 2006 ; Fernow, 1909). À travers le 19e siècle, la science de la foresterie se diffuse à travers le monde par les pratiques coloniales et impérialistes de l'époque (Agnoletti, 2006). De nombreuses « forêts d'état » voient alors le jour pour exploiter à long terme des forêts allant de l'Inde jusqu'à l'Amérique du Sud, souvent sous la protestation des populations locales (Agnoletti, 2006 ; Radkau, 2012).

Mais c'est durant la deuxième moitié du 20e siècle que la foresterie intégrera les trois composantes qui définissent son état actuel : sa mécanisation, sa relation avec les populations locales (souvent autochtones), et ses impacts importants sur l'environnement. L'évolution des technologies mène alors à l'apparition de grandes industries forestières. Ces industries récoltent et transforment le bois à un prix réduit via des machines toujours plus complexes et efficaces, à une échelle sans précédent. En parallèle, plusieurs programmes internationaux lancés par des organismes comme la Food and Agriculture Organisation des Nations Unies (FAO) vont tenter de populariser une foresterie « sociale ». Celle-ci est alors censée intégrer davantage les populations locales dans les prises de décisions liées à l'exploitation des forêts (Agnoletti, 2006). Cette foresterie « sociale » tentera par la suite de se transformer en foresterie « durable » par le biais des organismes de certifications, tels que le Forest Stewardship Council. Ces organismes découlent en retour de différentes rencontres internationales dans les années 80-90 comme le Montréal Process de 1994, et du travail de groupes environnementaux (Radkau, 2012).

À travers les changements les plus récents des pratiques de foresterie et d'aménagement forestier se cache également une transformation radicale des sociétés humaines, que l'on nomme la grande accélération (Steffen *et al.*, 2005). Depuis 1950, la quantité d'êtres humains sur terre et les ressources qu'ils consomment ont augmenté à un rythme sans précédent dans l'histoire de la planète. La population mondiale est passée de 2,5 à plus de 8 milliards d'individus entre 1950 et 2020, alors que la production de bois a presque doublé dans le même espace de temps (Betts *et al.*, 2021a). Aujourd'hui, la production de bois rond (troncs) mondiale continue d'augmenter, et a atteint 4 milliards de mètres cubes par an; de quoi remplir 4000 fois le volume de l'Empire State Building de New York chaque année avec du bois. À cette transformation des sociétés humaines s'ajoute une transformation importante du climat terrestre, qui affecte progressivement la survie de nombreuses espèces qui composent les forêt (Albrich *et al.*, 2020a ; Bouchard *et al.*, 2019 ; Lee *et al.*, 2023 ; Lyra *et al.*, 2017) (voir section 0.3).

C'est en réponse à ces pressions nouvelles qui pèsent à présent sur les forêts que s'est écrit le chapitre le plus récent de l'histoire de l'aménagement forestier. Depuis le début des années 1990, de nouvelles philosophies d'aménagement se virent ainsi proposées dans le milieu académique. On y trouve par exemple la « nouvelle foresterie » (new forestry) de Franklin (1989), qui propose que l'aménagement doit avoir pour but non pas de préserver les forêts comme des « usines à bois », mais comme des écosystèmes complexes. De ce point de vue, l'aménagement forestier est alors à la fois un outil de production et de régulation des forêts, alors que, selon Franklin, « ce qui est laissé après une coupe sur le site est plus important que ce qui en est pris » (Gillis, 1990). D'autres approches se centrent sur l'adaptation des forêts aux changements globaux (*global-change driven*), sur l'émulation des effets des perturbations naturelles (aménagement écosystémique, *nature-based* ou *disturbance-based*), ou bien sur le fait de répondre à différents besoins sociaux et écologiques en faisant varier les stratégies d'aménagement dans l'espace (*landscape-level*) (Messier *et al.*, 2019).

Depuis ses origines et à travers ses différentes formes dans l'histoire, l'aménagement forestier est ainsi une force majeure qui régule et transforme les écosystèmes et les sociétés humaines à travers le monde (Ennos, 2021). Ses impacts sur la structure et la composition globale des forêts du monde sont aujourd'hui certains, et observés à des échelles spatiales et temporelles variées (Dieler *et al.*, 2017 ; Li *et al.*, 2023 ; Martin *et al.*, 2020). Son interaction avec les autres composantes des changements globaux est également considérée avec préoccupation. L'exploitation des forêts pour leur bois est en effet l'une des causes principales de la déforestation dans les pays tropicaux (Armenteras *et al.*, 2017 ; Pendrill *et al.*, 2019), et les routes générées par l'aménagement forestier peuvent indirectement faciliter la déforestation par d'autres activités économiques (e.g., agriculture ; Kleinschroth *et al.* 2019). En parallèle de la déforestation, l'aménagement forestier peut participer à la fragmentation des forêts (Broadbent *et al.*, 2008), bien que les impacts de cette fragmentation sont encore à établir (Fahrig, 2017 ; Fletcher *et al.*, 2018). L'aménagement forestier peut également mener à la dégradation des forêts en impactant leur structure et leur composition (Betts *et al.*, 2021b ; Matricardi *et al.*, 2020). Plus globalement, plusieurs études suggèrent que l'aménagement forestier contribuerait à la diminution de la quantité des peuplements forestiers plus âgés dans le monde, souvent appelés « vieilles forêts » (Cyr *et al.*, 2009 ; Martin *et al.*, 2020 ; Shorohova *et al.*, 2011). Enfin, il peut entrer en synergie avec l'intensification des perturbations naturelles causées par les changements climatiques, et ainsi altérer d'autant plus les écosystèmes forestiers et les services écosystémiques qu'ils fournissent (voir section 0.3) (Bradford *et al.*, 2013 ; Russell *et al.*, 2018). Malgré ces tendances générales, les pratiques d'aménagement forestier, ainsi que ses impacts, varient grandement par pays et par biome. La plupart des pays occidentaux et industrialisés ont vu leurs surfaces forestières diminuer beaucoup plus légèrement que la plupart des pays tropicaux depuis plusieurs décennies, et certains ont même vu un gain de surface forestière marqué (FAO, 2020). Dans ces pays, la préoccupation se déplace alors du problème de la déforestation vers le problème de la dégradation d'habitat forestier (Betts *et al.*, 2021b).

Bien que l'aménagement forestier soit aujourd'hui une activité aux impacts sociaux et environnementaux importants, cette importance ne devrait pas diminuer dans les années à venir. Au contraire, il risque dans le futur de se placer au centre de conflits liés aux différents usages et bénéfices que les sociétés tirent des forêts, incarnés dans la notion des « services écosystémiques ». Cette notion fut popularisée par le *Millennium Ecosystem Assessment* dans les années 2000 (Reid *et al.*, 2005), avec pour but de mettre en valeur les différentes fonctions des écosystèmes importantes pour l'humanité (Wallace, 2007). Elle fut néanmoins mainte fois critiquée pour leur continuation de la marchandisation de la nature (Schröter *et al.*, 2014) et pour leur occultation des autres valeurs des forêts : leur valeur intrinsèque (Batavia et Nelson, 2017 ; Schröter *et al.*, 2014), existentielle (Batavia et Nelson, 2017 ; Bekessy *et al.*, 2018), ou leur valeur pour d'autres êtres sentients (Batavia et Nelson, 2017). Je me cantonnerais ainsi à les utiliser comme un chemin rhétorique pour explorer des fonctions importantes des forêts aux yeux des sociétés humaines, sans souscrire à la vision du monde qu'ils impliquent.

Le service le plus souvent associé aux forêts est alors la production de bois, employant plus de 50 millions de personnes dans le monde. Bien que 18 % de la population mondiale habite dans des constructions faites de bois (FAO, 2014), la production de bois ne représente cependant que moins de 1 % du PIB mondial, et son utilisation varie grandement entre pays (p. ex., matériaux de construction versus source d'énergie) (Agrawal *et al.*, 2013). Les forêts produisent également une longue liste d'autres produits importants : champignons, plantes médicinales, gommes, miel, baies, sirop d'érable, noix et graines, etc. La valeur de ces produits peut alors rivaliser celle du bois à l'échelle nationale, que ce soit dans les pays riches ou plus pauvres (FAO, 2014 ; Lovrić *et al.*, 2020). Les forêts ont également une valeur spirituelle (Coggins et Chen, 2022), culturelle (Lewis et Sheppard, 2005 ; Trigger et Mulcock, 2005) et récréative (Stier-Jarmer *et al.*, 2021) importante pour les humains. Bien que l'évaluation de cette valeur soit complexe (Kanowski et Williams, 2009), les forêts les plus appréciées par la majorité de la population sont souvent celles « intouchées », « complexes » ou « anciennes » (old-growth), peu ou pas impactées par d'autres utilisations (Clark, 2011 ; Kanowski et Williams, 2009). Cet aspect sacré et transcendant de la forêt « vierge » est particulièrement représenté par des personnalités qui ont grandement inspiré les mouvements environnementaux dans les pays occidentaux comme John Muir, qui déclamait que « Le chemin le plus clair vers l'univers passe par une forêt sauvage » (Muir, 1979).

Au-delà des ressources qu'elles contiennent et de leur valeur culturelle, les forêts régulent également leurs milieux physique et biologique de manière importante. La présence de forêts peut ainsi réduire ainsi l'érosion des sols (Altieri *et al.*, 2018 ; Liu *et al.*, 2020 ; Zuazo et Pleguezuelo, 2009), filtrer l'eau et améliorer sa qualité (Griscom *et al.*, 2017 ; Neary *et al.*, 2009), filtrer l'air et réduire les maladies pulmonaires (Cavanagh et Clemons, 2006 ; Nowak *et al.*, 2014, 2018), influencer le climat local et global (Ellison *et al.*, 2017), et même réduire la prévalence de certaines maladies (Karjalainen *et al.*, 2010). Mais plus important encore aujourd'hui est le rôle des forêts dans les cycles biogéochimiques de la terre, et en particulier dans le cycle du carbone. Les forêts stockent environ 45 % du carbone organique présent sur terre (Bonan, 2008) et entre 75 % et 92 % de la biomasse vivante (Bar-On *et al.*, 2018 ; Pan *et al.*, 2011). Elles absorberaient également plus de CO2 de l'atmosphère chaque année que les océans (Masson-Delmotte *et al.*, 2021), faisant d'elles un des acteurs cruciaux de la mitigation des changements climatiques (Griscom *et al.*, 2017). À ce sujet, les forêts plus complexes et moins aménagées pour de la récolte de bois semblent contenir plus de carbone par le biais d'une plus grande biomasse (Hua *et al.*, 2022). Néanmoins, le potentiel des forêts pour mitiger les changements climatiques en servant de puits de carbone global est aujourd'hui remis en cause (voir section 0.3).

Les forêts du monde contiennent également la plupart de la biodiversité terrestre. Il est estimé que 80 % des amphibiens, 75 % des oiseaux, 68 % des mammifères et la grande majorité des espèces de plantes vasculaires terrestres vivent dans des forêts (FAO et UNEP, 2020). Cette biodiversité est en retour importante pour les autres services fournis par les forêts. Ainsi, les forêts plus diverses dans leur composition peuvent stocker plus de carbone (da Rosa et Marques, 2021 ; Díaz *et al.*, 2009 ; Liu *et al.*, 2018), mieux protéger les sols contre l'érosion (Song *et al.*, 2019), augmenter la productivité des arbres et donc le volume de bois produit (Liang *et al.*, 2016 ; Paquette et Messier, 2011), mais aussi être plus appréciées pour des raisons culturelles, spirituelles ou esthétiques (Brockerhoff *et al.*, 2017 ; Clark, 2011). De ce fait, il existe des corrélations importantes entre l'approvisionnement de différents services, ou entre des services et certaines caractéristiques des forêts. En particulier, les forêts structurellement plus complexes (p. ex. avec des arbres d'âges différents ou des trouées) tendraient à promouvoir une plus grande diversité de services écosystémiques que des forêts structurellement plus simples (Felipe-Lucia *et al.*, 2018 ; LaRue *et al.*, 2019 ; Storch *et al.*, 2023).

Mais bien que des corrélations existent entre certains services, il existe également des tensions entre la récolte de bois et d'autres services écosystémiques liés à la biodiversité, au climat, à l'eau ou à la récréation (Blattert *et al.*, 2023 ; Gutsch *et al.*, 2018 ; Pohjanmies *et al.*, 2021). Ces tensions peuvent alors se traduire en des conflits sociaux parfois importants qui peuvent s'éloigner des considérations techniques (comment récolter du bois ?) pour se rapprocher de considérations politiques et idéologiques (pourquoi récolter du bois ici ? qui possède la forêt ? qui en profite ?) ou même de questions de justice sociale (est-ce que des minorités opprimées souffrent de l'aménagement forestier ?) (Dhiaulhaq et McCarthy, 2020 ; Jakobsson *et al.*, 2021 ; Nousiainen et Mola-Yudego, 2022). À ces tensions qui pèsent sur l'aménagement forestier s'ajoute la responsabilité d'aider les forêts à se préparer à un futur dont les conditions seront sûrement plus hostiles et incertaines pour les forêts (Messier *et al.*, 2015) (voir section 0.3). La dégradation ou la destruction des forêts engendrées par l'aménagement ou par les changements climatiques peut alors impacter de nombreux services : émissions de gaz à effet de serre (Baccini *et al.*, 2017), érosion des sols (Flores *et al.*, 2020), altération du cycle de l'eau (Ellison *et al.*, 2017), perte de biodiversité (Ahrends *et al.*, 2010 ; Betts *et al.*, 2021b ; Venier *et al.*, 2014), impacts culturels (Forest Peoples Programme, 2014), problèmes de santé (Bolton *et al.*, 2022 ; Karjalainen *et al.*, 2010 ; Nowak *et al.*, 2014), perte de productivité primaire, ou encore perte de produits non ligneux (FAO, 2014) pour n'en citer que certains (Blattert *et al.*, 2023 ; Gutsch *et al.*, 2018 ; Schulze *et al.*, 2020).

Ces problématiques environnementales et sociales liées aux forêts expliquent en grande partie pourquoi l'aménagement forestier est resté l'une des préoccupations majeures des mouvements environnementaux du 20e et du 21e siècle, qui mirent en avant l'idée d'une foresterie multi-usage (Agnoletti, 2006). Cette nouvelle vision de la foresterie fue parallèlement justifiée par les théories nouvelles en gestion des écosystèmes de Holling (Holling et Meffe, 1996) et de Ostrom (Lopez et Moran, 2016 ; Ostrom, 2009), qui soulignaient la nécessité d'une gestion des forêts prenant en compte la complexité des écosystèmes et des sociétés humaines. L'aménagement forestier a donc dû évoluer face à cette pression sociale résultant d'une vision nouvelle et plus multidimensionnelle des forêts, en contraste avec une vision passée plus « économique » centrée sur le volume de bois qu'elles contiennent (Sheppard *et al.*, 2020). Cette tension entre la vision « économique » et la vision « multi-dimensionnelle » se reflète cependant dans plusieurs autres débats plus larges liés à la gestion des écosystèmes à travers le monde. En particulier, les discussions entourant l'aménagement forestier s'inscrivent souvent dans le débat entre exploitation intensive et extensive des écosystèmes forestiers, qui est l'un des éléments au cœur de cette thèse (Betts *et al.*, 2021a ; Gladstone et Thomas Ledig, 1990 ; Lindenmayer *et al.*, 2012).

En conclusion, l'aménagement forestier devra conjuguer de nombreuses priorités dans le siècle qui vient, pour répondre aux différentes attentes sociales et environnementales qui l'entoure. Dans les parties suivantes, nous explorerons plus en détail les tensions qui existent entre les objectifs de production, de conservation et d'adaptation que l'aménagement forestier doit atteindre; les potentielles solutions pour les résoudre; mais aussi les obstacles à leur étude liés aux caractéristiques des écosystèmes forestiers.

0.2 Équien et inéquien, intensif et extensif, *sparing* et *sharing* : la dualité de l'aménagement forestier

Les différentes philosophies d'aménagement forestier combinent sur le terrain différentes pratiques de sylviculture, dont les objectifs sont plus restreints dans le temps et dans l'espace. Ces pratiques peuvent être des coupes forestières pour récolter du bois (coupes à blanc, coupes partielles, etc.), de l'éclaircissement (coupes de certains arbres pour aider les autres à mieux grandir), des plantations, ou bien des traitements des sols (scarification, etc.) (Doucet et Côte, 2009). Différentes pratiques de sylvicultures peuvent ainsi créer des structures d'âge et des compositions d'arbres différentes au sein des peuplements forestiers, en influençant les classes d'âge de ceux-ci et les espèces présentes. Le choix et la fréquence d'utilisation des pratiques de sylviculture peuvent alors aboutir à des stratégies d'aménagement plus « intensives », ou plus « extensives » (Betts *et al.*, 2021a). Dans les stratégies intensives, les forêts auront tendance à être affectées par des coupes fréquentes qui récoltent le maximum de bois possible, et qui laissent peu d'arbres (voir aucun) dans l'aire coupée (coupes à blanc ou coupes totales) (Betts *et al.*, 2021a ; Roberge *et al.*, 2016). Elles favoriseront alors la présence de forêts avec des structures dites « équiennes » dans le paysage, où les arbres tendent à avoir le même âge en repoussant en même temps après une coupe totale (Nolet *et al.*, 2018). Ces stratégies intensives peuvent également utiliser des pratiques de plantations, de fertilisation et d'éclaircissement afin d'augmenter la productivité des peuplements du paysage (Messier *et al.*, 2020). Elles peuvent néanmoins créer des peuplements avec une structure d'âge plus simplifiée et une biodiversité réduite (Brockerhoff *et al.*, 2008 ; Burivalova *et al.*, 2014).

À l'inverse, les stratégies d'aménagement extensives tendent à utiliser des coupes dites « partielles », « de rétention » ou de « sélection », qui laissent une partie ou une majorité des arbres dans le peuplement après la coupe (Doucet et Côte, 2009). Elles créent alors des peuplements à structure « inéquiennes », qui présentent une grande diversité de classes d'âge (Boncina, 2011). Parce que ces coupes récoltent moins d'arbres, elles doivent cependant être effectuées sur de plus grandes surfaces pour récolter un volume de bois similaire à celui récolté dans un aménagement intensif (Betts *et al.*, 2021a). En contraste, l'intensification se fait généralement au détriment d'autres services écosystémiques sur les surfaces récoltées, comme l'approvisionnement en produits non ligneux, la récréation, le stockage de carbone ou la qualité d'habitat pour différentes espèces (Pohjanmies *et al.*, 2021 ; Roberge *et al.*, 2016 ; Verkerk *et al.*, 2014). L'aménagement extensif pourrait ainsi représenter une approche plus multifonctionnelle, moins centrée sur la récolte de bois et favorisant une plus grande diversité de services. Comme nous le verrons par la suite, cette vision peut cependant être réductrice, et peut même se révéler fausse lorsque ces stratégies sont observées à l'échelle du paysage.

Au-delà de cette division entre intensif et extensif, on distingue également les stratégies d'aménagement dites « équiennes » et « inéquiennes ». Celles-ci représentent la combinaison de pratiques de sylviculture qui créent des peuplements à structure régulière (équienne) ou irrégulière (inéquienne) (Boncina, 2011 ; Nolet *et al.*, 2018). Ces deux types d'aménagement sont alors parfois présentés comme les deux grandes catégories d'aménagement forestier, en opposition l'une avec l'autre (Hann et Bare, 1979). L'aménagement équien et inéquien sont néanmoins souvent corrélés en pratique à des aménagements intensifs ou extensifs respectivement, comme suggéré plus tôt (Betts *et al.*, 2021a). Toutefois, ils peuvent tous deux varier en intensité, et il n'est pas impossible d'élaborer un aménagement inéquien plus intensif qu'un aménagement équien (Bell *et al.*, 2008). Pour rendre les choses encore plus complexes, la distinction entre les termes de « sylviculture équienne et inéquienne » ou « d'aménagement équien et inéquien » est également devenue floue à travers le temps (Hann et Bare, 1979 ; O'Hara, 2002). Nous garderons alors ici l'idée que la sylviculture se fait à l'échelle du peuplement, et l'aménagement à l'échelle du paysage par une combinaison de pratiques sylvicoles (O'Hara, 2002 ; Proulx *et al.*, 2000). La popularité de la sylviculture inéquienne a ainsi suivi des cycles successifs, et a aujourd'hui regagné en popularité dans le domaine scientifique et public (Diaci *et al.*, 2011 ; O'Hara, 2002 ; Pommerening et Murphy, 2004 ; Schütz *et al.*, 2012). Ce regain de popularité fut en grande partie basé sur la perception sociale négative et les impacts environnementaux des coupes à blanc (ou coupes totales), qui entrent dans la catégorie des méthodes équiennes (Cyr *et al.*, 2009 ; Martin *et al.*, 2020 ; Park *et al.*, 2005).

Les impacts observés de l'aménagement équien ou inéquien dans la littérature sont cependant souvent plus nuancés que ne l'indique leur perception sociale. À l'échelle du peuplement, les pratiques équiennes sont ainsi considérées comme importantes pour permettre la régénération d'espèces pionnières intolérantes à l'ombre, pour lesquelles des méthodes inéquiennes conviendraient moins (Keenan et Kimmins, 1993). Dans la théorie de l'aménagement écosystémique, les coupes équiennes peuvent également reproduire l'impact de feux sévères qui peuvent créer des grandes surfaces de peuplements équiens naturels (Bergeron, 2004 ; Fenton *et al.*, 2009 ; Gauthier et Vaillancourt, 2008). Cette vision a cependant été critiquée, car les coupes peuvent se combiner aux feux au lieu de les remplacer et ainsi générer trop de jeunes forêts (Bergeron, 2004 ; Bergeron *et al.*, 2004 ; Cyr *et al.*, 2009). Les coupes équiennes (c.-à.-d. des coupes totales) ne seraient également pas toujours capables de répliquer les impacts de feux sévères, en particulier sur les sols si du brulage contrôlé n'est pas fait après la coupe (Carleton et Maclellan, 1994 ; Fenton *et al.*, 2009 ; Lafleur *et al.*, 2018 ; Simard *et al.*, 2001). Néanmoins, les méthodes de sylviculture équienne ne sont pas considérées dans la littérature comme incompatibles avec une gestion des forêts basées sur d'autres objectifs que la production de bois (Bergeron, 2004 ; Keenan et Kimmins, 1993). Elles restent néanmoins amplement critiquées pour leur application souvent trop systématique liée à des motivations économiques (Bergeron, 2004 ; Keenan et Kimmins, 1993).

En contraste, de nombreuses études ont observé des effets positifs de méthodes de sylvicultures inéquiennes sur différentes fonctions et caractéristiques des forêts, en comparaison aux méthodes équiennes. Les coupes partielles peuvent ainsi augmenter l'abondance et la diversité de différents taxa en comparaison aux coupes totales, comme les herbacées (Götmark *et al.*, 2005), les lichens importants pour des espèces comme le caribou (Boudreault *et al.*, 2013 ; Stone *et al.*, 2008), les oiseaux (Ruel *et al.*, 2013 ; Tittler *et al.*, 2001), les petits mammifères comme les lapins et différents rongeurs (Ruel *et al.*, 2013), et les insectes (Graham-Sauvé *et al.*, 2013 ; Ruel *et al.*, 2013). Plus globalement, la sylviculture inéquienne semble augmenter l'approvisionnement en services écosystémiques et mieux conserver la biodiversité des peuplements, à l'instar de la sylviculture pratiquée au sein de l'aménagement extensif (Fedrowitz *et al.*, 2014 ; Pukkala, 2016).

Ces conclusions sont cependant remises en doute par une revue de littérature récente de Nolet *et al.*, (2018). Sur 99 études, 23 révélaient une amélioration de mesures liées aux fonctions et caractéristiques importantes des forêts (biodiversité, types de sols, carbone) avec l'aménagement inéquien (Nolet *et al.*, 2018). En contrepartie, 16 études montraient une amélioration de ces mesures avec l'aménagement équien, et 60 études étaient considérées comme « équivoques ». Par exemple, (Pukkala *et al.*, 2011) ont montré que l'aménagement inéquien était plus performant que l'équien pour augmenter toutes leurs mesures (bois fournis, carbone stocké et récolte de myrtilles), sauf dans le cas du carbone stocké dans les peuplements de Pin sylvestre (*Pinus sylvestris*) où l'aménagement équien était plus performant. Ce portrait renforce ainsi l'idée que la sylviculture équienne et inéquienne peuvent toutes deux êtres nécessaires dans différentes situations, et pour des objectifs d'aménagement différents. Ces études contiennent également des limites importantes en ce qui concerne leurs restrictions spatiales, temporelles et expérimentales. La plupart d'entre elles se déroulent ainsi sur des périodes très courtes en comparaison au fonctionnement des écosystèmes forestiers (moins de 10 ans) (Nolet *et al.*, 2018). Cela peut causer des résultats ambigus qui peuvent dépendre du moment de la vie du peuplement qui est considéré (Nolet *et al.*, 2018). Ces études peuvent également cacher d'autres compromis ayant lieu dans le contexte des peuplements étudiés en se limitant à une seule catégorie de mesures (p. ex. biodiversité ou carbone ou productivité). Mais surtout, ces études se réalisent généralement à l'échelle du peuplement, ce qui ne permet pas d'intégrer ou de mesurer des processus ayant lieu à l'échelle du paysage . En particulier, elles ne seront pas suffisantes pour étudier les effets des compromis entre la surface à récolter dans le paysage et l'intensité des traitements sylvicoles. L'exploration de ces compromis s'inscrit dans un débat plus grand qui a pris vie dans l'étude sur l'utilisation des terres pour l'agriculture, et qui oppose les stratégies dites de *land-sparing* et *land-sharing*.

Concrètement, le débat du *land-sparing* et *land-sharing* tente d'établir la stratégie la plus efficace pour protéger les écosystèmes naturels dans un paysage agricole (Green *et al.*, 2005). La stratégie du *land-sparing* consiste alors à utiliser des formes d'agriculture intensive qui conservent relativement peu de biodiversité sur les zones exploitées, mais qui permettent de produire suffisamment de nourriture sur une plus petite surface. Les écosystèmes naturels restants dans le paysage peuvent alors être « épargnés » (*spared*) et conservés intégralement. À l'inverse, la stratégie du *land-sharing* utilise des formes d'agriculture extensive avec moins de pesticides, moins de fertilisants, ou bien des combinaisons de plantes cultivées. Cette agriculture extensive serait moins dommageable à la biodiversité environnante, mais moins efficace en termes de rendement par hectare (Green *et al.*, 2005). À l'instar des stratégies d'aménagement forestier extensives, la stratégie de *land-sharing* demanderait alors de plus grandes surfaces agricoles pour produire la même quantité de nourriture. Elle partagerait (*share*) ainsi le paysage avec les écosystèmes naturels. Actuellement, les données empiriques semblent montrer que la stratégie de *land-sparing* serait globalement plus efficace que le *land-sharing* pour préserver les écosystèmes naturels et les services qu'ils fournissent (Ekroos *et al.*, 2016 ; Phalan, 2018).

Mais depuis son développement, le débat (ou modèle) du *land-sparing* et du *land-sharing* a suscité de nombreuses critiques quant à ses hypothèses sous-jacentes (Phalan, 2018). Les surfaces agricoles intensives ne sont ainsi pas toujours des « déserts » de biodiversité, comme supposé plus haut (Kremen, 2015). De même, l'intensification de l'agriculture ne mène également pas toujours à plus de *sparing* comme il a été observé au cours du 20e siècle (Pellegrini et Fernández, 2018). Ce phénomène est lié au paradoxe de Jevons, processus par lequel l'augmentation de l'efficacité d'une technologie tend à augmenter son utilisation, et donc à augmenter la consommation globale de ressources utilisée cette technologie (Balmford, 2021 ; Giampietro et Mayumi, 2018). Sur un plan spatial, il est aussi à noter que la distinction entre *sparing* et *sharing* peut également devenir floue selon l'échelle spatiale utilisée (Ekroos *et al.*, 2016). En contraste, sur un plan temporel, les changements climatiques pourraient fortement influencer les rendements agricoles intensifs dans le futur (Phalan, 2018). Certaines études suggèrent également que l'augmentation de la production agricole n'est pas la priorité actuelle pour assurer la sécurité alimentaire mondiale, ce qui rendrait la question du *sparing*/*sharing* bien moins urgente qu'elle ne le parait aujourd'hui (Fischer *et al.*, 2014 ; Tscharntke *et al.*, 2012). À titre d'exemple, Poore et Nemecek (2018) estiment qu'une transition vers des régimes alimentaires favorisant les aliments d'origine végétale pourrait amener jusqu'à 75 % de réduction de la quantité de terres nécessaires pour nourrir l'humanité. Certaines études ont également souligné que l'augmentation des populations humaines rendra toute solution au débat ultimement temporaire, impliquant la nécessité de certaines mesures de régulation (Balmford, 2021).

Comme indiqué précédemment, la problématique du *land-sharing* et *land-sparing* a également été formulé dans le domaine de la foresterie (Betts *et al.*, 2021a). Comme en agriculture, la foresterie s'associe à un gradient de milieux résultant des pratiques utilisées, allant des forêts primaires aux plantations en monoculture (Betts *et al.*, 2021a). L'agriculture se fait néanmoins sur des parcelles privées relativement petites où les perturbations naturelles sont généralement contenues et sur des sols fertiles avec une végétation très différente des écosystèmes naturels, en contraste total avec la foresterie (Betts *et al.*, 2021a). Les milieux résultants d'une stratégie intensive en foresterie (p. ex. plantations d'arbres en monocultures) sont ainsi capables d'abriter une biodiversité plus importante que les monocultures intensives agricoles (Brockerhoff *et al.*, 2008). Néanmoins, le modèle du *sharing*/*sparing* peut rester pertinent en foresterie, alors que le bois reste une ressource importante et que moins d'opportunités de réduction des surfaces exploitées y ont été identifiées qu'en agriculture (p. ex. régimes végétariens/véganes, réduction du gaspillage alimentaire, etc.). Notamment, il est possible de voir les stratégies d'aménagement forestier intensives d'un autre œil lorsque vu à travers le prisme du *sparing*/*sharing*. À l'échelle locale, celles-ci peuvent avoir tendance à simplifier la structure des forêts et à favoriser la production de bois au détriment de leurs autres fonctions, comme je l'ai mentionné plus haut. Mais à l'échelle du paysage, elles pourraient permettre de protéger de plus grandes surfaces de forêts naturelles par le biais de réserves, en utilisant moins de surface pour la récolte (Betts *et al.*, 2021a). À grande échelle, cette stratégie pourrait alors être plus efficace que du *sharing* lié à un aménagement extensif (Betts *et al.*, 2021a). Bien entendu, cela ne vaudrait que si ces méthodes intensives n'encouragent pas la coupe de plus bois via le paradoxe de Jevons, ce qui a été suggéré dans la littérature (Messier, 2022). À ma connaissance, il n'existe cependant pas d'études mesurant la présence de ce paradoxe dans le secteur de la foresterie.

Plus globalement, la démonstration de l'efficacité des stratégies de *land-sharing* et *sparing* en foresterie est rare, et ne compte que quelques études qui abordent le problème de manière explicite à ce jour (Balmford, 2021 ; Betts *et al.*, 2021a). La majorité de ces études concernent des modèles ou des mesures empiriques réalisés en contexte de forêt tropicale (Edwards *et al.*, 2014 ; França *et al.*, 2017 ; Griscom *et al.*, 2018 ; Harris et Betts, 2023 ; Mestre *et al.*, 2020 ; Montejo-Kovacevich *et al.*, 2018 ; Runting *et al.*, 2019), avec seulement deux études de simulation en forêt tempérée et boréale (Côté *et al.*, 2010 ; Tittler *et al.*, 2015). Les conclusions de ces études sont également loin d'être unanimes. En forêt tropicale, Edwards *et al.* (2014) et França *et al.* (2017) ont mesuré que le *land-sparing* était plus efficace pour maximiser l'abondance et la diversité de scarabées bousiers, de fourmis et d'oiseaux que le *sharing*. En contraste, Mestre *et al.* (2020) ont observé qu'une stratégie intermédiaire comprenant plus de *sharing* que de *sparing* y était plus efficace pour maximiser la diversité phylogénétique et fonctionnelle des oiseaux. Montejo-Kovacevich *et al.* (2018) et Griscom *et al.*, (2018) ont également observé via des travaux de modélisation spatiale que la stratégie de *sharing* était globalement plus efficace pour maximiser la biodiversité des papillons, la biodiversité globale et le stockage de carbone en forêt tropicale. Cette efficacité du *sharing* était cependant conditionnelle, certaines espèces de papillons préférant le *sparing* (Montejo-Kovacevich *et al.*, 2018). De plus, le *sharing* n'y était réellement efficace que si des méthodes de récolte avec réduction des impacts (p. ex. abatage contrôlé des arbres pour ne pas en endommager d'autres, protection des sols, etc.) étaient utilisées en conjonction avec un respect de la propriété des terres forestières (Griscom *et al.*, 2018). De surcroit, Runting *et al.* (2019) ont observé que l'utilisation de méthodes de coupe avec réductions des impacts était plus importante que des stratégies de *sharing* ou *sparing* pour la diversité et l'abondance de différents carnivores, primates et chauve-souris. Enfin, les études de Côté *et al.* (2010) et Tittler *et al.* (2015) à l'interface de la forêt tempérée et boréale suggèrent qu'un mélange de *land-sparing* et de *land-sharing* est plus efficace pour maximiser la quantité et la connectivité d'habitats pour la faune à l'échelle du paysage.

Ces résultats suggèrent ainsi des compromis complexes entre les stratégies de *land-sparing* et *land-sharing* et les écosystèmes forestiers. Leurs effets changeraient selon les taxa ou les services écosystémiques considérés, les méthodes de sylviculture utilisées, et aussi en fonction du contexte social du paysage étudié. Betts *et al.*, (2021a) proposent également que l'effet de ces stratégies sur les espèces (fauniques ou florales) qui vivent en forêt dépend de leur réponse à des niveaux d'intensité de récolte intermédiaire. De ce fait, le choix d'une stratégie uniforme de *sparing* ou de *sharing* ne serait pas forcément optimal pour favoriser la biodiversité du paysage, car chacune favoriserait l'habitat de certaines espèces au détriment d'autres. Cette idée est renforcée par les travaux de Harris et Betts (2023), qui ont montré que les différentes stratégies (*sparing*, *sharing*, ou un mélange des deux) tendaient à favoriser différents types d'habitats forestiers (forêt jeune, mature ou vieille) en milieux tropicaux.

Face à ce constat, la communauté scientifique a proposé des stratégies alternatives utilisant à la fois du *land-sparing* et du *land-sharing* au sein d'un même paysage. La plus connue en foresterie et souvent citée dans le contexte du *land-sharing*/*sparing* est l'aménagement en TRIAD proposée en 1992 par Seymour et Hunter (1992) (Balmford, 2021 ; Betts *et al.*, 2021a ; Harris et Betts, 2023). Le terme « TRIAD » fait référence à un système explicite de zonage en trois zones différentes au sein du paysage forestier : aménagement intensif, aménagement extensif, et zones de conservation (Himes *et al.*, 2022 ; Messier, 2022 ; Messier *et al.*, 2020). La proportion du paysage occupée par ces trois zones et leur disposition sont alors censés être réfléchis et définis après une analyse globale de celui-ci, amenant ainsi de nombreux avantages. Les zones intensives peuvent ainsi théoriquement permettre d'atteindre le niveau de récolte nécessaire sur une plus petite surface, libérant de l'espace pour des réserves naturelles plus grandes. Une disposition stratégique des différentes zones de la TRIAD peut également optimiser leurs rôles respectifs, réduire l'impact environnemental de la récolte, mais aussi prendre en compte l'acceptabilité sociale des différentes méthodes de récoltes (Messier *et al.*, 2020). La TRIAD pourrait également favoriser une diversité d'habitats forestiers à l'échelle du paysage par le biais des différentes zones, permettant la survie d'espèces adaptées à des milieux différents : des forêts plus jeunes et équiennes en zones intensives, plus matures et inéquiennes en zones extensives, et plus vieilles et naturelles dans les réserves (Harris et Betts, 2023). Cette diversité des peuplements forestiers générée par la TRIAD serait alors, selon certaines études, plus importante que la présence d'une hétérogénéité d'âge au sein même des peuplements (Schall *et al.*, 2018). Enfin, la TRIAD permettrait de surveiller la performance des trois zones dans leurs différents rôles en comparant leur évolution respective, afin de pouvoir adapter l'aménagement au sein de chacune en retour (Messier, 2022).

Le problème du *sparing*/*sharing* en foresterie reste malgré tout relativement inexploré, et de nombreuses interrogations demeurent à son sujet. En particulier, le sujet des chemins forestiers nécessaires aux opérations forestières générés dans les deux stratégies n'a été qu'à peine effleuré dans la littérature. Ces chemins sont généralement différents des routes plus usuelles, avec une surface souvent faite de gravier ou de sol à nu, ainsi qu'un trafic routier réduit (Sessions *et al.*, 2016). Ils peuvent néanmoins causer certains problèmes comme au Québec, où la construction de chemins est passée de 150 km par an en 1970 à plus de 4000 km en 2015 (Bourgeois *et al.*, 2005). Aujourd'hui, les chemins forestiers y sont ainsi devenus le centre d'un débat important quant à leur quantité, leur manque d'entretien par le gouvernement, et leurs impacts sur les forêts (Deschênes, 2022 ; Plamondon Lalancette et Movilla, 2021 ; Radio Canada, 2021, 2022). La littérature existante dépeint l'influence des chemins forestiers comme globalement négative pour la conservation des forêts, mais avec certaines nuances importantes. Ils peuvent ainsi représenter une barrière au déplacement de certaines espèces, qui auraient tendance à les éviter (Marsh *et al.*, 2005 ; Witmer et deCalesta, 1985). Mais les chemins forestiers peuvent également aider au déplacement certaines espèces et ainsi en impacter d'autres, comme dans le cas du loup et du Caribou forestier (*Rangifer tarandus caribou*) au Québec (Commission indépendante sur les caribous forestiers et montagnards, 2022 ; Saint-Laurent *et al.*, 2014 ; Whittington *et al.*, 2011). Les chemins forestiers peuvent également aider à la dispersion d'espèces invasives (Mortensen *et al.*, 2009) et affecter l'habitat de différentes espèces dont certains oiseaux (Ortega et Capen, 1999), scarabées (Koivula, 2005) et d'autres macro-invertébrés (Haskell, 2000) en influençant la température, la végétation et les sols environnants (Avon *et al.*, 2010 ; St-Pierre *et al.*, 2021 ; Zhou *et al.*, 2020). Ils sont aussi une source importante de pollution de l'eau pour les cours d'eau qu'ils traversent ou qui se trouvent à proximité, en particulier car beaucoup de chemins sont dégradés et érodés (Akbarimehr et Naghdi, 2012 ; Girardin *et al.*, 2022 ; Zemke, 2016). Certaines études suggèrent même que les chemins forestiers auraient une influence notable sur les feux de forêt en augmentant les probabilités d'ignition (naturelles ou humaines) dans leur proximité, et en servant de coupe-feux physiques ainsi que de moyens de transport pour les pompiers (Narayanaraj et Wimberly, 2011, 2012 ; Yocom *et al.*, 2019).

Pour toutes ces raisons, mais aussi parce que la construction de chemins forestiers représente un coût substantiel pour les industries forestières (Epstein *et al.*, 2006 ; Groupe DDM et MFFP du Québec, 2020 ; Toscani *et al.*, 2020), réduire leur quantité peut être un objectif important de l'aménagement forestier. Toutefois, la nécessité de réseaux routiers forestiers plus permanents et plus étendus en utilisant des stratégies de *sharing* ou de *sparing* n'a été jusqu'ici seulement suggérée dans des ouvrages moins récents, mais pas observée ou testée (Alexander et Edminster, 1977 ; United States Department of Agriculture, 1997). Tittler *et al.* (2015) et Tittler *et al.* (2012) ont simulé une construction de routes dans un paysage avec différentes stratégies d'aménagement forestier, par le biais de lignes droites reliant le centre des zones de coupe avec le segment de route le plus proche. Leurs résultats suggèrent qu'agglomérer les coupes dans le paysage tendrait à réduire la quantité de routes nécessaires pour les coupes. À ma connaissance, ce sont les seules études ayant tenté un tel exercice à l'échelle du paysage, et dans le but d'explorer l'effet de différentes stratégies d'aménagement. La fragmentation du paysage générée par les chemins forestiers (Reed *et al.*, 1996) dans les stratégies d'aménagement intensives et extensives reste alors inexplorée.

Nous avons ainsi décrit le débat du *sparing*/*sharing* dans le cadre de l'aménagement forestier, qui s'exprime par l'utilisation de stratégies intensives ou extensives d'aménagement. Certaines méthodes, comme l'aménagement en TRIAD, proposent un mélange des deux stratégies pour optimiser les compromis qui en résultent. Néanmoins, le problème reste globalement peu exploré malgré son importance. Ceci est particulièrement le cas en ce qui concerne les chemins forestiers générés par l'utilisation de ces deux approches, comme nous l'avons souligné. Mais la menace grandissante des changements climatiques, qui impactent déjà fortement les forêts aujourd'hui, risque de rendre ces questions à la fois plus pressantes et plus complexes que jamais à résoudre. Nous aborderons donc à présent leurs impacts, et les conséquences de ceux-ci sur l'aménagement forestier et son étude.

0.3 L'aménagement conjugué au futur par les changements climatiques

Les changements climatiques sont un bouleversement pour les écosystèmes, mais aussi pour les priorités de la recherche en écologie. Depuis les travaux de Arrhenius (1896) jusqu'à la fondation du GIEC (Groupe International d'Experts sur le Climat) en 1988 (Programme des Nations unies pour l'environnement et Organisation météorologique mondiale, 1988), leur découverte est une histoire fascinante que je n'aborderais cependant pas ici. Leur cause est néanmoins identifiée aujourd'hui comme le résultat de plusieurs siècles de transformation de l'environnement par les sociétés humaines (Lee *et al.*, 2023). À l'échelle du globe, la température moyenne de la surface terrestre a augmenté de 1,1 °C depuis le début du 19e siècle (Lee *et al.*, 2023). En conséquence, la quasi-totalité des surfaces émergées de la planète ont subi une augmentation de leurs extrêmes de chaleur annuels (Lee *et al.*, 2023). Les vagues de chaleur et les périodes de sécheresse sont ainsi devenues plus fréquentes à travers le globe, y compris au Canada. Les années récentes y ont ainsi été très chaudes, alors que 2010 fut l'année la plus chaude au Canada depuis 1948. En cette année 2023, de nombreux feux de forêt se sont déclenchés au Québec à cause d'un mois de mai excessivement sec et chaud, défrayant la chronique (Barnes *et al.*, 2023). Dans le futur, il est prévu que la tendance de l'augmentation des températures globale continue malgré les mesures d'atténuation des émissions de gaz à effet de serre par l'humanité (GES) (Lee *et al.*, 2023). Le scénario climatique récent SSP2 (*Shared Socioeconomic Pathways*, ou Trajectoires Socioéconomiques Partagées), correspondant à une continuation des activités humaines actuelles, implique ainsi un réchauffement planétaire global de 2,7 °C à 3,5 °C (Lee *et al.*, 2023).

Les impacts des changements climatiques sur les forêts identifiés dans la littérature scientifique sont nombreux. À l'échelle globale, on observe une croissance accrue de certains types de forêts via l'effet fertilisant du CO2 ajouté dans l'atmosphère par l'humanité (Davis *et al.*, 2022 ; Haverd *et al.*, 2020). Des hivers plus doux et des printemps plus précoces en forêts boréales et tempérées y allongeraient également les saisons de croissance (Gauthier *et al.*, 2015 ; Reyer *et al.*, 2017). Les perturbations naturelles et humaines plus fréquentes pourraient également générer des quantités additionnelles de jeunes forêts, parfois plus productives en termes de biomasse (Kurz *et al.*, 2007 ; Mikoláš *et al.*, 2021). Ces effets positifs des changements climatiques sur la croissance des forêts pourraient cependant être contrebalancés par d'autres. Les perturbations naturelles plus fréquentes et intenses, ainsi qu'un climat plus chaud et des sécheresses plus extrêmes pourraient ainsi réduire la productivité des forêts tropicales (Lyra *et al.*, 2017 ; Mitchard, 2018), tempérées (Reyer *et al.*, 2017) et boréales (Gauthier *et al.*, 2015 ; Kurz *et al.*, 2007, 2008). Les perturbations naturelles et les températures changeantes pourraient également favoriser certaines espèces d'arbres au détriment d'autres. Il est ainsi prévu que les forêts boréales tendront à être progressivement remplacées par des forêts tempérées dans le futur, par le biais de feux de forêt plus fréquents et de températures plus clémentes en hiver et au printemps (D'Orangeville *et al.*, 2023 ; Gauthier *et al.*, 2015 ; Xu *et al.*, 2022). Ces changements de composition d'habitat entraîneraient en conséquence des changements à long terme dans les réseaux trophiques des forêts en altérant l'abondance et la survie de différentes espèces, ainsi que leurs interactions (D'Orangeville *et al.*, 2023 ; Queiroz *et al.*, 2022).

L'influence des changements climatiques sur les régimes des perturbations naturelles que subissent les forêts (à savoir leur fréquence, intensité et distribution) est très discutée dans la littérature (Seidl *et al.*, 2014, 2017). Une majorité d'études s'accorde ainsi sur une augmentation globale des risques de feux de forêt (Jain *et al.*, 2022), de sécheresses (Hammond *et al.*, 2022 ; Vicente-Serrano *et al.*, 2020), d'épidémies de pathogènes ou d'insectes (Sturrock *et al.*, 2011) et de chablis (Feng *et al.*, 2023) dans le futur, malgré des variations locales attendues (Lee *et al.*, 2023). Cependant, l'augmentation des risques liés aux vents, à la neige, au gel et aux insectes est moins certaine que pour ceux liés aux feux, à la sécheresse et aux pathogènes (Seidl *et al.*, 2017). Ces augmentations pourront causer des boucles de rétroaction positive par l'accentuation des émissions de carbone liées à la mortalité des arbres, renforçant en retour les changements climatiques (Zheng *et al.*, 2021). Prises ensemble, les augmentations prévues des perturbations naturelles risquent alors d'avoir des répercussions importantes sur les forêts et sur l'humanité. Les effets négatifs de ces perturbations sur différents services écosystémiques rendus par les forêts risquent en effet d'augmenter de par leur intensification (Thom et Seidl, 2016). Les changements dans le taux de croissance et de mortalité au sein des peuplements forestiers pourraient toutefois avoir plus d'influence sur les forêts que le changement des régimes de perturbations naturelles (Bouchard *et al.*, 2019 ; Boulanger *et al.*, 2018 ; Toledo *et al.*, 2011).

En conséquence, différentes études ont tenté de quantifier l'effet futur des changements climatiques sur les services écosystémiques rendus par les forêts. La plupart de ces effets seront très probablement négatifs (Albrich *et al.*, 2020a ; Elkin *et al.*, 2013 ; Mina *et al.*, 2017 ; Roces-Díaz *et al.*, 2021), en particulier pour les services de régulation comme le stockage de carbone (Albrich *et al.*, 2020a ; Giles-Hansen et Wei, 2022 ; Wang *et al.*, 2021). L'approvisionnement en services écosystémiques deviendrait plus instable alors que certaines forêts transitionneraient vers des états alternatifs, comme dans le cas des forêts alpines qui deviendraient plus feuillues (Albrich *et al.*, 2020a). Certaines forêts deviendraient également moins capables de fournir plusieurs services écosystémiques en même temps, comme dans le cas des forêts méditerranéennes dont l'approvisionnement en services dépendront de leur capacité à résister à des sécheresses plus fréquentes (Irauschek *et al.*, 2017 ; Morán-Ordóñez *et al.*, 2020 ; Schirpke *et al.*, 2020). Certaines forêts non aménagées pourraient alors, dans certains cas, mieux fournir certains services écosystémiques que des forêts exploitées (Seidl *et al.*, 2019). Les changements climatiques réduiraient également la quantité de bois disponible pour un rendement soutenu sur le long terme (Brecka *et al.*, 2020 ; Hiltner *et al.*, 2021 ; Zhao *et al.*, 2022), et donc la capacité de stockage des produits du bois (Paluš *et al.*, 2020). Les forêts ayant plus de mal à fournir différents services écosystémiques en même temps, il serait également plus difficile de conjuguer la récolte de bois avec d'autres services (p. ex. stockage de carbone, biodiversité, etc.) (Blattert *et al.*, 2023 ; Gutsch *et al.*, 2018).

Un aménagement forestier dit « durable » et « respectueux de l'environnement » deviendra ainsi plus difficile à mettre en place, car il risquera d'accentuer les effets des changements climatiques par différents procédés (Girona *et al.*, 2023). Les forêts récoltées peuvent ainsi devenir plus sensibles aux perturbations naturelles, dont les feux, le chablis, les épidémies d'insectes (Canelles *et al.*, 2021 ; Silvério *et al.*, 2019 ; Stritih *et al.*, 2021). Les chemins forestiers peuvent également augmenter les risques d'ignitions de feux (Narayanaraj et Wimberly, 2012), aider des espèces invasives à envahir le paysage (Mortensen *et al.*, 2009), et augmenter la mortalité des arbres par chablis (Abdi *et al.*, 2020). L'effet combiné des coupes et des perturbations peut également réduire les capacités de stockage de carbone par les forêts (Bradford *et al.*, 2013 ; Gough *et al.*, 2007 ; Russell *et al.*, 2018). Certaines pratiques pourraient cependant réduire le carbone émis par des forêts après une perturbation (p. ex., coupes de récupération) (Pilli *et al.*, 2021). Des études supplémentaires concernant l'effet des changements climatiques sur les services écosystémiques des forêts seront cependant nécessaires pour mieux anticiper ces problèmes. En effet, les études existantes concernent plus souvent les pays occidentaux, les forêts publiques, et en particulier des services de régulation (p. ex. stockage de carbone, régulation du climat) plutôt que d'autres (Acharya *et al.*, 2019). Il reste ainsi beaucoup d'incertitude autour des forêts gérées par des communautés et des pays en voie de développement.

Prendre en compte les impacts des changements climatiques en foresterie demandera ainsi différentes mesures. Principalement, on peut classer ces mesures en deux grandes catégories : mitigation et adaptation. Les stratégies de mitigation ont pour but de réduire les changements climatiques par le biais des forêts, en conservant ou en augmentant le carbone qui y est stocké, ou qui est stocké dans les produits du bois (Churkina *et al.*, 2020 ; Nambiar, 2019 ; Nunes *et al.*, 2020). Bien que ces différentes stratégies aient pour cible principale la gestion du carbone stocké par les forêts ou dans le bois, leurs impacts sont multiples. Elles peuvent alors affecter positivement ou négativement d'autres fonctions des forêts comme la biodiversité qu'elles contiennent où leurs qualités récréatives (Blattert *et al.*, 2023 ; Ellison *et al.*, 2017 ; Law *et al.*, 2018). Leur implémentation à de grandes échelles peut également représenter des coûts économiques importants (p. ex. en augmentant le prix de la récolte via des crédits carbone), et leurs effets seront différents selon les caractéristiques des forêts concernées (p. ex. densité de carbone stocké par surface, perturbations naturelles fréquentes, etc.) (Austin *et al.*, 2020).

À l'opposé, les stratégies d'adaptation ont pour but d'aider les forêts à préserver leur fonctionnement face aux impacts des changements climatiques (Bolte *et al.*, 2009 ; Cosofret et Bouriaud, 2019 ; Evans et

Perschel, 2009). Ces stratégies seraient essentielles, alors que les changements climatiques risquent de progresser plus rapidement que les capacités d'adaptation naturelles des forêts (Burrows *et al.*, 2011 ; Seidl *et al.*, 2016). Bolte *et al.* (2009) en définissent trois grandes catégories : adaptation active, adaptation passive, et conservation de la structure actuelle des forêts. Cette dernière approche est néanmoins difficile à justifier, car de nombreuses forêts présentent déjà des signes inquiétants d'altérations de leurs fonctions ou de leur structure (Baccini *et al.*, 2017 ; Giles-Hansen et Wei, 2022 ; Martin *et al.*, 2020). À l'inverse, les stratégies d'adaptations actives et passives tentent de modifier la structure des forêts en augmentant généralement leur diversité en essences et leur complexité structurelle (c.-à-d. arbres de classes d'âges différentes) (Jandl *et al.*, 2019). La diversité de structure et d'âge des forêts d'un paysage a en effet été proposée comme essentielle pour maintenir la résilience des écosystèmes (abordée plus en détail dans la section suivante), en augmentant leur diversité de réponses suite à des perturbations et en les rapprochant de leur variabilité naturelle (Messier *et al.*, 2013 ; Seidl *et al.*, 2016 ; Timpane-Padgham *et al.*, 2017). Pour se faire, les stratégies actives peuvent utiliser différentes pratiques de sylvicultures comme les éclaircies (pour induire une structure hétérogène à l'échelle du peuplement), ou bien un mélange de méthodes équiennes et inéquiennes (pour créer des forêts structurellement différentes à l'échelle du paysage) (Cosofret et Bouriaud, 2019 ; Evans et Perschel, 2009 ; Jandl *et al.*, 2019). En contraste, les stratégies passives ont pour objectif d'effectuer cette transition par le biais de processus naturels (Jandl *et al.*, 2019). Jandl *et al.* (2019) suggèrent que ces stratégies passives peuvent être efficaces seulement si les processus naturels peuvent mener ces forêts à des peuplements considérés comme plus résistants et résilients aux conditions futures. Néanmoins, ils notent que de nombreuses forêts ne respectent pas ce critère, et demanderaient alors des interventions actives pour être aidées.

Les stratégies de mitigation et d'adaptation liées aux forêts sont considérées dans la littérature comme prometteuses, mais elles sont aussi critiquées sur différents aspects (Drever *et al.*, 2021 ; Fargione *et al.*, 2018 ; Kaarakka *et al.*, 2021). Ces stratégies de mitigation et d'adaptation se sont alors rejointes au sein de nouvelles philosophies d'aménagement forestier comme la climate-smart forestry, qui intègrent explicitement les opportunités et les défis liés aux changements climatiques (Verkerk *et al.*, 2020). Mais malgré l'enthousiasme provoqué par ces nouvelles visions de l'aménagement dans la recherche ou dans le grand public, l'évidence de leur efficacité reste mitigée ou manquante (Popkin, 2019). Ceci est particulièrement le cas pour les stratégies de mitigation, dont l'efficacité est déjà impactée par les changements climatiques (Anderegg *et al.*, 2020). De nombreuses forêts du monde seraient ainsi déjà devenues des sources de carbone par les effets combinés du climat sur leur croissance et des perturbations naturelles et humaines, remettant en question leur capacité de conserver leurs stocks de carbone dans le temps (Baccini *et al.*, 2017 ; Giles-Hansen et Wei, 2022 ; Popkin, 2015 ; Wang *et al.*, 2021). Les programmes d'afforestation ou de reforestation pourraient également provoquer un réchauffement additionnel du climat si implémentés dans des zones nordiques à cause des changements d'albédo (Bernier *et al.*, 2011 ; Bright *et al.*, 2016 ; Kreidenweis *et al.*, 2016 ; Lawrence *et al.*, 2022), et entrer en conflit avec l'agriculture dans les zones plus au sud (Doelman *et al.*, 2020 ; Kreidenweis *et al.*, 2016). Ces nouvelles forêts présenteront également un risque élevé de disparaitre avant d'atteindre leur pleine maturité via des perturbations naturelles plus intensives et fréquentes (Mansuy *et al.*, 2013 ; Reyer *et al.*, 2009). Au final, les produits du bois représenteraient un puits de carbone relativement petit et à la nature transitoire, dont la comparaison avec les émissions liées aux coupes est parfois équivoque (Hudiburg *et al.*, 2019 ; Johnston et Radeloff, 2019 ; Malcolm *et al.*, 2020 ; Peng *et al.*, 2023). De plus, les stratégies de mitigation concentrées sur le carbone stocké par les forêts peuvent en retour réduire l'approvisionnement d'autres services écosystémiques importants (Blattert *et al.*, 2023 ; Gutsch *et al.*, 2018).

Malgré ces doutes concernant les stratégies de mitigations, les stratégies d'adaptation resteront essentielles dans le futur. L'évidence qui entoure ces stratégies reste toutefois elle aussi manquante ou incertaine. La littérature existante concerne majoritairement (comme souvent) les pays du nord, dont les sociétés plus riches qui subissent des climats moins chauds seront moins vulnérables aux changements climatiques que les pays du sud (Brunette *et al.*, 2018). Cette littérature concerne également plus souvent l'évaluation des risques liés à la perte de fonctions des forêts (en particulier leur productivité), ce qui n'aide pas toujours les décisions d'aménagement qui doivent aussi prendre en compte l'efficacité des pratiques d'adaptations (Keenan, 2015, 2017). Bien qu'il soit également recommandé de développer des stratégies d'adaptation qui augmentent la résistance et la résilience des fonctions des forêts (Hörl *et al.*, 2020), peu d'études les considèrent comme objectif ou comme sujet (Vilà-Cabrera *et al.*, 2018). De la même manière, la plupart des études existantes concernent des approches réactives (c. à d. mises en place quand un problème apparait) alors que la littérature tend à recommander des approches adaptées au futur (planned) (Bernier *et al.*, 2009 ; Brunette *et al.*, 2018). Empiriquement, certaines pratiques de sylviculture ont été associées à une résistance accrue des peuplements traités à des perturbations comme la sécheresse (D'Amato *et al.*, 2013 ; Keenan et Nitschke, 2016). Des études de modélisation ont également montré l'efficacité de stratégies de plantations d'espèces plus adaptées au climat futur pour augmenter la résilience des peuplements (Hof *et al.*, 2017 ; Lucash *et al.*, 2017). Les pratiques testées jusqu'à aujourd'hui tendent néanmoins à généralement augmenter la résistance ou la résilience de certains services écosystémiques précis (souvent production et stockage de carbone), souvent au détriment d'autres comme les services culturels ou de support (Vilà-Cabrera *et al.*, 2018).

Au-delà de l'évidence existante, de nombreuses théories concernant les possibilités d'adaptation des forêts par l'aménagement forestier restent alors largement inexplorées ou peu concluantes. De plus, les stratégies d'adaptation ne sont pas isolées des autres problèmes liés à la foresterie. Elles devront alors se faire en conjugaison avec les débats existants entre les stratégies d'aménagements intensives et extensives, ou de *land-sparing* et de *land-sharing* que nous avons mentionnées précédemment. L'adaptation est en effet une dimension supplémentaire, qui ne s'aligne pas toujours avec celle de la production ou de la conservation (Jandl *et al.*, 2019). Cette intégration entre adaptation et *sparing-sharing* est toutefois relativement absente de la littérature. Plus globalement, les stratégies d'adaptation sont rarement considérées à l'échelle du paysage ou sur le long terme (sauf pour certains processus comme la migration assistée), et plutôt à l'échelle du peuplement ou de quelques années (Vilà-Cabrera *et al.*, 2018).

Récemment, Messier *et al.* (2020) ont proposé une fusion de l'approche dite des « réseaux fonctionnels » et de la stratégie d'aménagement en TRIAD. Cette fusion aurait ainsi pour but d'adresser simultanément les trois priorités de conservation, de production et d'adaptation au sein des paysages forestiers. En plus de la TRIAD décrite précédemment, l'approche des réseaux fonctionnels se base sur la l'écologie des traits fonctionnels (Violle *et al.*, 2007) et de la connectivité fonctionnelle (Taylor et Fahrig, 2006). Les traits fonctionnels concernent les caractéristiques (morphologiques, physiologiques, comportementales, etc.) qui affectent la survie, la croissance ou la reproduction d'un individu d'une espèce donnée. On distingue plus particulièrement les traits « d'effet » qui concernent l'effet des espèces sur leur environnement, et les traits de « réponse » qui concernent leur réactions à leur environnement. En contraste, la connectivité fonctionnelle concerne la manière dont des entités spatiales (p. ex. parcelle d'habitat, configuration d'un paysage) facilitent le déplacement des individus d'une espèce donnée. Le but de l'approche des réseaux fonctionnels est alors d'augmenter la résilience des forêts face à des perturbations futures incertaines, en amplifiant la diversité et la redondance des traits de réponse présents dans les des peuplements d'un paysage (Messier *et al.*, 2019 ; Mori *et al.*, 2013). Ceci serait fait par le biais de plantations stratégiquement placées, afin que les espèces plantées puissent se diffuser dans d'autres peuplements et les enrichir à leur tour (Messier *et al.*, 2019). La théorie des réseaux fonctionnels propose ainsi de visualiser les forêts comme un réseau d'échange de diversité fonctionnelle par le biais de la dispersion d'espèces d'arbres entre peuplements (Craven *et al.*, 2016). Les plantations « fonctionnelles » propres à cette approche pourraient alors être intégrées au sein de la TRIAD, insérant une composante d'aide à l'adaptation des forêts dans la problématique du *sharing-sparing* (Messier *et al.*, 2020). À ma connaissance, cette stratégie est la seule qui tente d'adresser explicitement les priorités de conservation, de production et d'adaptation en même temps. Bien que la théorie des réseaux fonctionnels et son application ont été testées dans différentes études de modélisation (Aquilué *et al.*, 2021 ; Mina *et al.*, 2022), cette nouvelle version de la TRIAD reste à ce jour intouchée en dehors de sa formulation.

Dans cette section, nous avons ainsi vu que l'évidence à l'appui des stratégies d'aménagement face aux changements climatiques est généralement contrastée, et souvent manquante. Ceci contraste avec le fait que les personnes en lien avec la gestion des forêts semblent pourtant s'attendre à ce que les changements globaux impacterons très négativement celles-ci dans le futur (Himes *et al.*, 2023). La raison principale de l'état de la littérature est que l'étude des écosystèmes forestiers est difficile, surtout sur de grandes échelles spatiales et temporelles. Dans la section suivante, nous aborderons ainsi ces défis et les outils les plus adaptés pour les relever.

0.4 Le défi de l'étude de l'aménagement forestier

L'étude des impacts de l'aménagement forestier sur les forêts implique des obstacles techniques et physiques importants. Le premier concerne l'échelle temporelle : les forêts sont des systèmes dont la dynamique globale est relativement lente (Binkley, 2021 ; Maréchaux *et al.*, 2021). En contraste, la science est pratiquée par des humains aux vies relativement courtes et dont le financement des activités est souvent instable, bien que des programmes de suivis à long terme se sont développés avec le temps (Nature, 2022). Ce problème de la dynamique relativement lente des forêts est également renforcé par l'urgence des changements globaux. En 2023, il reste ainsi déjà moins de 80 ans avant la date de 2100 régulièrement utilisée par le GIEC comme horizon de modélisation des scénarios climatiques (Lee *et al.*, 2023). Ces 80 années ne laisseront en retour que peu de temps pour expérimenter sur les forêts, qui peuvent prendre au-delà de 80 ans pour atteindre des états particuliers (p. ex. vieilles forêts).

Le second problème lié à l'étude des écosystèmes forestiers concerne leur étendue spatiale. De nombreuses études se concentrent sur l'observation de ce qui est communément appelé un « peuplement » forestier. Ceux-ci concernent généralement une surface de forêt de quelques hectares dont les caractéristiques (âge, composition, structure, etc.) sont relativement homogènes (Nyland, 2016). En contraste, il est possible d'étudier les forêts à l'échelle du paysage, qui concerne généralement des milliers ou des millions d'hectares. À cette échelle, de nombreux autres processus et éléments principalement absents à l'échelle du peuplement, mais qui influencent les forêts se révèlent : la dispersion des arbres sur de grandes distances, les régimes de perturbations naturelles et humaines (comme les feux pouvant couvrir des milliers d'hectares), les interactions de forêts avec les caractéristiques géologiques et topographiques du paysage (bassins versants, élévation, sols, etc.), les chemins forestiers, et bien d'autres encore (Binkley, 2021). L'hétérogénéité des forêts observées à cette échelle est alors bien plus grande qu'à l'échelle du peuplement, qui est par définition homogène. La considération de l'échelle du paysage est ainsi essentielle pour prédire l'évolution des forêts, mais elle reste difficile (Green *et al.*, 2005). Les études à grande échelle spatiale peuvent ainsi rencontrer des problèmes similaires aux études à long terme, comme le manque de financement ou les difficultés techniques.

Mais au-delà des difficultés liées à leurs échelles, il est parfois même difficile de définir et mesurer les processus étudiés au sein des forêts. Un exemple connu concerne la notion de résilience des forêts, aujourd'hui présentée comme une priorité de l'aménagement forestier, et mentionnée plusieurs fois dans les sections précédentes. Lorsqu'appliquée à des écosystèmes, elle concerne leur capacité à absorber des changements environnementaux lents ou brusques sans pour autant changer leurs fonctions (Folke, 2016 ; Reyer *et al.*, 2015). De nombreuses mesures ont été proposées à travers le temps pour la mesurer, y compris dans les forêts; mais aucune de ces mesures individuelles ne semble avoir véritablement créé un consensus. Au contraire, il est aujourd'hui accepté que la résilience d'un écosystème doit toujours être mesurée en relation à une variable donnée (p. ex., la biomasse d'une forêt) et à une perturbation donnée (p. ex., un feu de forêt) (Carpenter *et al.*, 2001 ; Folke, 2016). Il est aussi admis que différentes mesures doivent être utilisées pour capturer les différents « aspects » de la résilience (p. ex. vitesse de récupération, risque de transition vers des états alternatifs, etc.) (Folke, 2016). Bien qu'elle représente un concept important face aux changements climatiques, la résilience des forêts est ainsi difficile à mesurer. En particulier, elle demande souvent de mesurer les variables d'intérêt avant et après d'une perturbation, ce qui est particulièrement difficile pour des perturbations naturelles dont l'occurrence n'est pas contrôlable. On y retrouve également un problème cité plus haut : comment étudier les possibilités d'augmenter la résilience des forêts face aux changements climatiques, si leur étude demande des décennies de données pour conclure ?

Face à ces nombreux défis liés à l'étude des forêts (et des écosystèmes en général), la discipline de la modélisation semble avoir gagné en popularité en sciences de l'environnement depuis la fin du 20e siècle

(Ríos-Saldaña *et al.*, 2018). De nombreux modèles ont ainsi été développés pour étudier le fonctionnement et prédire l'évolution des écosystèmes forestiers. Dans le passé, ces modèles étaient plus généralement des systèmes d'équations différentielles pouvant être résolus par des méthodes analytiques , comme le modèle de Lotka-Volterra (Breckling *et al.*, 2011). Les interactions spatiales parfois présentes dans ces modèles étaient alors modélisées implicitement, dans le sens ou les entités modélisées n'avaient pas de position donnée dans l'espace (p. ex. l'augmentation de la surface d'un type de forêt diminue celle d'un autre type, sans que ces forêts soient placées sur une carte). Avec l'évolution des technologies — et en particulier des ordinateurs —, de nombreux modèles de simulation sont apparus, basés sur des interactions entre des individus (Judson, 1994), des entités ou des processus (Cuddington *et al.*, 2013). Un grand nombre de ces nouveaux modèles étaient également spatialement explicites, permettant ainsi de placer ces entités ou processus simulés sur des cartes (Breckling *et al.*, 2011). Ainsi, dans ces modèles, de nombreuses entités — comme des peuplements forestiers ou des arbres — peuvent évoluer simultanément, et en interaction spatiale les unes avec les autres. Elles peuvent alors générer des formes d'auto-organisation spatiale qui ne sont pas forcément visibles dans des modèles analytiques, comme dans le cas de boucles de rétroactions où les patterns spatiaux formés par un processus (p. ex. les parcelles de jeunes forêts crées par des feux) influencent en retour le processus (p. ex. les feux sont moins fréquents ou se diffusent moins à cause des jeunes forêts). Bien que la complexité de ces modèles est alors devenue trop importante pour déduire leur comportement de manière analytique, ils peuvent être explorés par le biais de simulations. Durant celles-ci, un algorithme représentant la structure du modèle modifie ses variables internes de manière itérative à travers les dimensions temporelles et spatiales présentent au sein du modèle. Ces modèles peuvent alors être particulièrement intéressants pour des applications de prédictions (Evans, 2012).

Aujourd'hui, le domaine de l'écologie forestière est ainsi équipé d'une diversité de modèles toujours plus complexes. On y trouve en particulier les *Forest Landscape Models* (FLMs) comme LANDIS, iLand, LANDIS-PRO ou encore FATE-HD, qui sont particulièrement adaptés pour explorer les dynamiques des forêts sur le long terme et à de grandes échelles spatiales. Les FLMs sont des modèles spatialement explicites simulant des paysages entiers comprenant des forêts. Leur développement a suivi l'apparition de l'écologie du paysage (landscape ecology) en tant que discipline, mais aussi l'apparition de langages de programmation plus accessibles (Scheller et Mladenoff, 2007). On trouve des FLMs plus ou moins complexes, alors que les plus avancés se distinguent par trois grandes composantes (Scheller et Mladenoff, 2007) : la présence d'interactions spatiales (p. ex., des perturbations naturelles ou des espèces arbres qui se diffusent par zones de forêts voisines); la simulation de la dynamique de communautés d'arbres (c.-à-d. de l'évolution de la végétation à l'échelle plus basse du peuplement forestier); et la simulation de différents processus au sein des écosystèmes forestiers (p. ex., la productivité primaire nette des peuplements simulés, leur stockage de carbone, leur recyclage de nutriments, etc.). L'un des FLMs les plus connus aujourd'hui est LANDIS-II, un modèle au développement ouvert (*open-source*) qui comprend les trois composantes citées plus haut (Scheller *et al.*, 2007). Depuis sa création en 2007, LANDIS-II a été utilisé dans plus de 200 études différentes (LANDIS-II Foundation, s. d.). Programmé dans le langage C#, sa structure permet la programmation d'extensions modulaires qui peuvent être installées séparément. Ces extensions peuvent alors être choisies librement par toute équipe de recherche afin de s'adapter à des questions ou des contextes particuliers. De plus, plusieurs extensions peuvent représenter le même processus, mais avec différents niveaux de détails. Le développement de nouveaux modules peut également toucher de nouvelles questions de recherches sans avoir à recréer un modèle entier, tout en continuant de profiter de l'expertise représentée par les modules existants.

La modélisation présente ainsi de grands avantages pour étudier les questions qui concernent les impacts de l'aménagement forestier sur de grands espaces (*land-sharing* et *land-sparing*) et dans le temps (changements climatiques) surlignées dans les sections précédentes. Les modèles représentent néanmoins des abstractions de la réalité, et en sont donc par définition une représentation incomplète. Cependant, ces abstractions permettent en retour d'explorer la dynamique des écosystèmes forestiers au-delà des possibilités des études empiriques. Les modèles permettent en particulier de tenter de prédire l'évolution des forêts dans des conditions climatiques et sociales changeantes, même si ces conditions ne sont pas encore présentes aujourd'hui. Le domaine de la modélisation en écologie est également en constante évolution, avec l'apparition constante de nouveaux besoins (Albrich *et al.*, 2020b). À ce sujet, on peut rappeler la question des chemins forestiers, importante pour explorer les impacts des aménagements forestiers plus extensifs ou intensifs. De nombreux modèles furent développés à travers le temps pour déterminer les tracés de chemins forestiers nécessaires aux opérations de foresterie sur des cartes, comme PLANEX (Heinimann, 2017). Mais ces modèles sont plus souvent des modèles d'optimisation destinés aux ingénieurs forestiers, et fonctionnent sur de petits territoires de l'ordre de la centaine d'hectares. Ils nécessitent alors des temps de calcul importants pour générer des tracés aussi peu coûteux que possible, et demandent d'importantes quantités de données pour être paramétrés (p. ex., coûts de débardage par câble sur différentes distances) (Bont *et al.*, 2015 ; Chung *et al.*, 2004). Ce fonctionnement contraste avec le besoin d'explorer les impacts de l'aménagement forestier à l'échelle du paysage et sur de longues périodes. De plus, l'aménagement forestier est influencé à ces échelles par d'autres processus comme les perturbations naturelles qui ne sont pas représentées dans ces modèles d'optimisation. La construction de chemins forestiers devrait alors idéalement être intégrée au sein d'un FLM pour qu'elle soit considérée en interaction, et non séparément des autres facteurs qui affectent les forêts.

Cette section conclut ainsi le tour d'horizon de la littérature scientifique qui entoure le sujet de cette thèse. Dans les sections suivantes, nous préciserons alors les questions de recherches qui y seront abordées, ainsi que les méthodes et approches utilisées pour y répondre.

0.5 Question de recherche et objectifs

Le thème global de cette thèse, dessiné à travers les sections précédentes, concerne l'union des nombreuses missions actuelles de l'aménagement forestier. Comme nous l'avons vu, les forêts sont la source d'un grand nombre de ressources pour l'humanité et de processus naturels essentiels à l'échelle planétaire. Leur aménagement doit alors conjuguer les différents besoins humains qui en dérivent, sans compromettre les autres fonctions des forêts. Cette aménagement doit également se faire malgré les bouleversements de l'ère de l'anthropocène (Steffen *et al.*, 2015). Deux grandes questions ont alors émergé de ces discussions. La première concerne l'exploration des stratégies intensives ou extensives, ou de *land-sparing* ou *land-sharing* en foresterie :

Question 1. Quels sont les impacts des stratégies de *land-sparing* et de *land-sharing* en foresterie à l'échelle du paysage, et en particulier lorsque les chemins forestiers nécessaires aux coupes sont pris en compte ?

La deuxième concerne l'intégration des stratégies d'adaptation des forêts aux changements climatiques (e.g. augmentation de leur résilience aux perturbations futures) au sein de la problématique du *sharing-sparing* :

Question 2. Comment aider les forêts à s'adapter aux changements climatiques, tout en maintenant l'approvisionnement de leurs différents services sur le long terme ?

Ces deux questions représentant des sujets très vastes, nous tâcherons à présent de les préciser en fonction des connaissances surlignées dans les sections précédentes.

Concernant cette première question, nous avons vu que relativement peu d'études avaient exploré les impacts du *land-sharing* et du *land-sparing* en foresterie. La plupart de ces études concernaient la biodiversité de la faune ou de la flore au sein des paysages étudiés, ou bien l'abondance de certains taxa. Les études existantes n'ont également pas pu explorer un grand nombre de stratégies d'aménagement. En particulier, l'exploration d'un gradient de stratégies allant de plus intensives vers plus extensives permettrait de mieux capturer les relations (linéaires ou non) entre l'intensité de l'aménagement et ses impacts sur différentes variables ou fonctions des forêts. Nous avons également vu qu'en pratique, les stratégies d'aménagement intensives ou extensives se traduisent souvent par une utilisation prédominante de méthodes de sylviculture équiennes ou inéquiennes respectivement. Quelles que soient les méthodes employées, elles nécessitent cependant un réseau de chemins forestiers construit pour accéder aux forêts. Cette conséquence de l'aménagement n'a jamais été étudiée directement jusqu'ici en comparant des stratégies de *land-sparing*/*sharing*. Ces chemins peuvent pourtant causer en une perte de surface forestière, mais aussi en une fragmentation importante du paysage. Nous avons également mentionné que la structure et la composition des forêts tendaient à beaucoup influencer les services écosystémiques qu'elles fournissent. Pour cette raison, différentes études utilisent les vieilles forêts comme un proxy de biodiversité et de différents services qui leur sont propres, et qui sont moins fournis par des forêts plus jeunes (Balmford, 2021 ; Côté *et al.*, 2010 ; Tittler *et al.*, 2015). En prenant tous ces éléments en compte, nous pouvons alors préciser cette première question sous la forme suivante :

Question 1 (précisée). Comment est-ce que l'utilisation variable de stratégies de *land-sharing* (représentées par l'utilisation accrue de méthodes inéquiennes) en aménagement forestier affecte la composition et la fragmentation des paysages forestiers par le biais des coupes et des chemins qui leur sont nécessaires ?

Les principales hypothèses liées à cette première question seront alors :

- Les stratégies de *land-sharing* (représentées par l'utilisation accrue de méthodes inéquiennes) protègent une plus grande surface de forêts plus âgées en comparaison aux stratégies de *land-sparing*

• Les stratégies de *land-sharing* (et l'aménagement inéquien) nécessitent plus de chemins forestiers dans le paysage que les stratégies de *land-sparing*, et génèrent ainsi plus de fragmentation

Cette question implique alors d'atteindre les objectifs suivants :

• Développer des scénarios d'aménagement qui varieront la proportion d'utilisation de méthode équiennes ou inéquiennes

• Mesurer la dynamique des forêts concernées par le biais de leur âge et de leur fragmentation sur un temps suffisamment long pour observer les impacts de ces méthodes d'aménagements

• Déterminer des mesures de fragmentation, de composition et de fonctions des écosystèmes de forêts adaptés à la question

• Modéliser un paysage forestier réel avec les différents processus qui y ont lieu, en particulier les perturbations naturelles et humaines

Ce dernier objectif présente toutefois un problème important. Comme nous l'avons noté à plusieurs reprises, il n'existe pas (à ma connaissance) de modèle permettant de simuler à la fois les perturbations naturelles d'un paysage, les perturbations humaines, et la construction de chemins forestiers qui découle de ces dernières. Ce problème est d'ailleurs l'une des raisons pour lesquelles la question des chemins en aménagement forestier reste globalement intouchée. Pour le résoudre, il nous faudra ainsi développer un nouvel outil pour simuler la construction de chemins forestiers en parallèle des autres processus ayant lieu à l'échelle du paysage. Comme nous l'aborderons plus en détail dans la section suivante, cet outil sera développé au sein du modèle LANDIS-II, en tant que nouvelle extension.

Le développement de cet outil nous amène ainsi à une autre question de recherche précédant la première :

Question 0. Comment simuler la construction des chemins forestiers nécessaires à l'aménagement à l'échelle du paysage et sur de longue période ?

Cette question étant largement exploratoire, et liée à un contexte de modélisation particulier, il ne me parait pas nécessaire de la lier à des hypothèses précises. Elle nous amène néanmoins aux objectifs suivants :

- Identifier une approche de simulation dans la littérature permettant de répondre aux critères suivants :
  - Paramétrisable à l'échelle du paysage avec des données existantes
  - Capable de répliquer les caractéristiques d'un véritable réseau routier (densité de routes, distribution spatiale des routes, fragmentation de quoi, coûts de construction, etc.)
  - Qui évite l'utilisation de données spatiales trop fines qui sont très difficiles ou impossibles à obtenir ou à simuler à l'échelle du paysage (p. ex. position des zones de débarquement de bois, etc.)
  - Possède une performance satisfaisante en termes de temps de calcul (c.-à.-d. similaire aux autres extensions de LANDIS-II pour un pas de temps donné)
  - Se combine au fonctionnement des autres processus présents au sein d'un FLM (ici, LANDIS-II)
- Coder l'algorithme au sein d'un modèle existant (ici, LANDIS-II)
- Tester la capacité de l'approche/de l'algorithme déterminé à reproduire les caractéristiques les plus importantes d'un réseau routier réel (voir plus haut)

Concernant notre deuxième question de recherche, nous avons vu que quelques approches permettant de concilier *land-sharing*, *sparing* et adaptation au sein d'un même paysage ont déjà été proposées. On pensera en particulier à la stratégie TRIAD améliorée proposée par Messier *et al.* (2020). Celle-ci combine des zones extensives, intensives et de conservation avec des plantations ayant pour but d'augmenter la diversité fonctionnelle des peuplements du paysage. Le but principal de cette nouvelle TRIAD (qui sera nommée TRIAD+ dans le reste de cette thèse) est alors d'augmenter la résilience des forêts face à des événements perturbateurs extrêmes futurs (p. ex. de nouvelles épidémies d'insectes ou de pathogènes, de longues sécheresses, etc.). La fréquence, l'intensité et les impacts de ces événements étant incertains à cause de la complexité des changements globaux, ces plantations serviraient à augmenter la diversité de réponses fonctionnelles des peuplements en cas de perturbations. Cette diversité de réponses accrue augmenterait en retour la résilience de certaines caractéristiques importantes des peuplements, comme leur biomasse (Mori *et al.*, 2013). Il reste néanmoins à savoir si cette approche de TRIAD+ est réellement capable d'augmenter la résilience des peuplements. Il est aussi important de voir si cela ne se fait pas au prix d'autres fonctions, caractéristiques ou services importants des forêts, comme leur production de ressource ligneuse. En prenant tout cela en considération, nous pouvons ainsi préciser la question sous la forme suivante :

Question 2 (précisée). Est-ce que l'approche de la TRIAD+ est capable de préserver les caractéristiques importantes des forêts telles que leur biomasse face à des perturbations extrêmes incertaines, sans créer des compromis trop grands entre ces caractéristiques ?

Cette question testera ainsi les hypothèses suivantes :

- La TRIAD+ augmente la résilience de la biomasse des peuplements du paysage a différentes perturbations sur le long terme, en comparaison a d'autres stratégies d'aménagement

- La TRIAD+ réussit cette augmentation de résilience sans impliquer des compromis importants avec le volume de bois disponible à la récolte (représenté par la biomasse mature des peuplements)

Les objectifs résultants de cette question seront alors :

- Formuler précisément l'élaboration d'une stratégie TRIAD+ au sein d'un paysage donné

- Formuler d'autres scénarios d'aménagement (p. ex. Business as Usual) avec lesquels comparer la performance de la TRIAD+

- Simuler l'occurrence de plusieurs perturbations extrêmes des forêts du paysage modélisé

- Mesurer la résilience de la biomasse mature des forêts touchées par ces perturbations extrêmes

- Explorer la présence de compromis entre plusieurs aspects des forêts liées à l'utilisation de la TRIAD+ (volume de bois disponible à la récolte, surface des aires protégées, résilience de la biomasse mature des peuplements)

En considérant ces différentes questions, nous aurons ainsi l'opportunité d'apporter des contributions importantes au domaine de l'écologie forestière. Elles permettront, je l'espère, d'améliorer la manière dont les humains aménageront les forêts dans le futur. Notre première question en ordre chronologique — la question 0 — permettra ainsi le développement d'un outil qui pourra être utilisé dans de nombreuses recherches par la suite. Étant moi-même un grand admirateur du concept des logiciels gratuits et ouverts (*Free and Open Source Software*, ou FOSS), je ne peux m'imaginer programmer un tel outil sans le mettre à disposition de tout le monde. Les études futures en écologie forestière auront donc ainsi le pouvoir de simuler gratuitement la construction des chemins forestiers en parallèle des autres perturbations naturelles et humaines qui influencent les forêts. La prise en compte de ceux-ci permettra alors de révéler des compromis jusqu'ici inexplorés au sein des différentes stratégies d'aménagements. Un tel outil pourra également révéler les potentiels impacts d'une pression additionnelle que les forêts subissent sur le long terme, mais qui est rarement prise en compte.

Ma deuxième question — la question 1 — amènera des informations cruciales pour élucider le débat du *sharing*/*sparing* en foresterie. Elle permettra en particulier de comprendre comment ces stratégies influencent la quantité de chemins forestiers et la fragmentation du paysage. Loin d'être anodin, nous avons vu que le problème des chemins forestiers est aujourd'hui très discuté dans des régions comme le Québec où ils sont très prévalents. Leurs impacts sur certaines espèces peuvent ainsi y être importants, ce qui doit être pris en compte dans l'exploration de ces différentes stratégies d'aménagement. Enfin, notre dernière question — la question 2 — permettra d'explorer une stratégie d'aménagement qui cherche à jongler entre les objectifs de conservation, de production et d'adaptation : la TRIAD+. Ce numéro de jonglage risque malheureusement de devenir essentiel pour toutes les stratégies d'aménagement dans le futur, alors que les impacts des changements climatiques s'intensifieront. La TRIAD+ n'est peut-être pas la stratégie qui réussira le mieux ce numéro, et elle ne sera sûrement pas non plus la seule stratégie qui en sera capable. Néanmoins, l'étude de sa performance révélera des pistes pour l'amélioration de stratégies existantes, ou l'élaboration de nouvelles stratégies.

Pris tous ensemble, j'espère que ces travaux amèneront un changement de perspective sur les impacts de l'aménagement forestier. Ils permettront d'observer ces impacts avec un regard plus vaste et plus à long terme que les études de terrains qui sont réalisées en écologie. Je ne cherche néanmoins pas à renier les contributions des études de terrains qui, comme nous allons le voir, seront absolument vitales pour les travaux de cette thèse. Plutôt que de les remplacer, ces travaux chercheront ainsi à les compléter. Les forêts étant des systèmes fonctionnant à des échelles spatiales et temporelles dépassant souvent notre imagination, c'est en effet en les explorant à leurs échelles que nous comprendrons comment mieux les respecter et les traiter. Alors que notre relation avec elles continue de se transformer lentement depuis l'exploiteur jusqu'à l'intendant (Messier *et al.*, 2015), j'espère que cette thèse continuera d'améliorer et de réparer cette relation avec ce qui a été notre environnement ancestral, et ce qui sera nécessairement une partie vitale de notre futur.

0.6 Structure de la thèse

Cette thèse de doctorat se divise en 3 chapitres, précédés de la présente introduction. Le premier chapitre est un article publié dans le *Canadian Journal of Forest Research* en mars 2023. Il décrit le développement et le test de la nouvelle extension de LANDIS-II pour la simulation de la construction des chemins forestiers (Question 0). Le deuxième chapitre est un article publié dans le journal *Landscape Ecology* en septembre 2023. Il concerne l'exploration des impacts de stratégies extensives et intensives d'aménagement forestier à l'échelle du paysage (Question 1). Le troisième chapitre est un article en cours de publication. Il concerne l'exploration de la performance de la stratégie TRIAD+ à l'échelle du paysage pour augmenter la résilience des forêts face aux changements climatiques (Question 2). La dernière partie de cette thèse consiste en une conclusion globale quant à son sujet, alimentée par les résultats des 3 chapitres.

0.7 Méthodes et démarches

0.7.1 Zone d'étude

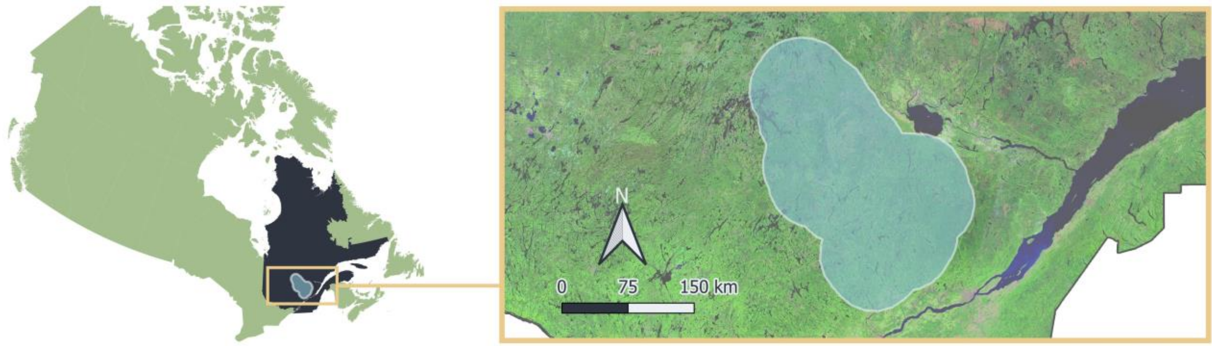

Figure 0-1 : Localisation de la zone d'étude utilisée durant les 3 chapitres de cette thèse dans la province de Québec (noir), au Canada (vert). Données d'imagerie du satellite Sentinel.

Nous décrirons ici brièvement les points de méthodes communes aux 3 chapitres de cette thèse, ainsi que la démarche globale qui y est réalisée. Les 3 chapitres utilisent tous une zone d'étude située dans la région administrative de la Mauricie, au sein de la province du Québec, au Canada (Figure 0-1). Cette zone d'environ 4 millions d'hectares contient différentes unités d'aménagement définies par le Ministère des Ressources Naturelles et des Forêts du Québec (MRNF), et est composée de forêts mixtes et boréales. Cette zone fut choisie pour différentes raisons, dont le fait que l'aménagement en TRIAD a été expérimenté dans la région de la Mauricie dans laquelle se situe la zone (Messier *et al.*, 2009). Plus globalement, le Québec propose des données publiques d'inventaire forestier très précises et couvrant toute la zone, facilitant la paramétrisation de LANDIS-II que nous avons utilisé (MFFP, 2015).

Cette zone d'étude contient très peu de communautés rurales, et un réseau routier principal peu dense en conséquence. Elle est alors un terrain idéal pour étudier les impacts de scénarios d'aménagement forestier sur de grands espaces de forêts, dans un milieu où les chemins forestiers seront plus nombreux que les routes principales. Les forêts de la zone sont publiques et gérées par le gouvernement québécois. Il est ainsi aisé d'y simuler des stratégies d'aménagement à grande échelle sans rencontrer la difficulté de la gestion des nombreuses parcelles de forêt privées. Le contexte de cette zone est ainsi particulier, et contraste avec des territoires forestiers plus anthropomorphisés comme en Europe. Cette particularité sera prise en considération pour interpréter correctement les résultats des recherches menées au sein de cette thèse.

La zone est également séparée en deux zones de végétation (domaines bioclimatiques), caractérisée par des régimes de perturbations naturelles différentes : les forêts boréales du nord de la zone tendent à subir de nombreux feux (Couillard *et al.*, 2022), alors que les forêts mixtes du sud tendent à subir des épidémies de tordeuse de bourgeon d'épinette (*Choristoneura fumiferana*) (Bergeron et Fenton, 2012). Ces deux régimes de perturbations naturelles différentes représentent ainsi en elles-mêmes un facteur qui peut influencer les effets de l'aménagement forestier.

0.7.2 Modèle utilisé

Les forêts de notre zone d'étude ont été simulées avec le modèle LANDIS-II, décrit dans les sections précédentes. Le chapitre 1 concerne sur le développement d'une nouvelle extension pour LANDIS-II, avec l'aide de Osvaldo Valeria de l'Université du Québec en Abitibi-Témiscamingue (UQAT). Les paramètres de cette nouvelle extension ont été dérivés d'avis d'experts et de données partagées par l'ancien Ministère des Forêts, de la Faune et des Parcs du Québec (MFFP) lors de mon stage doctoral au sein de celui-ci en 2019. Les extensions utilisées par la suite dans les chapitres 2 et 3 ne seront pas les mêmes, afin de s'adapter aux questions de recherches spécifiques des deux chapitres. Les deux chapitres utiliseront cependant les trois extensions Biomass Succession, Base Fire et Biomass Harvest de LANDIS-II. Biomass Succession est une extension simulant l'évolution de la biomasse des arbres modélisés. Ces arbres sont simulés au sein de chaque cellule dans des « cohortes d'âges », à savoir des groupes d'arbre du même âge et de la même espèce. Une valeur de biomasse pour chaque cohorte d'âge est alors calculée par l'extension. Cette biomasse évolue en fonction de différents facteurs : les pas de temps, la lumière disponible au sein de la cellule (simulant la compétition entre arbres au sein du peuplement), l'écorégion dans laquelle la cellule se trouve, l'espèce considérée, l'âge de la cohorte, etc.

Les paramètres de Biomass Succession furent fournis par l'équipe de Yan Boulanger (chercheur à Ressources Naturelles Canada) et Dominic Cyr (chercheur à Environnement et Changement Climatique Canada), tous deux co-auteurs des chapitres 2 et 3. Leur équipe a développé une méthodologie complexe utilisant le modèle physiologique PICUS pour simuler la croissance de différentes espèces arbres dans des conditions climatiques et sur des sols différents (Boulanger *et al.*, 2017). Ces simulations sous PICUS permettent alors de dériver les nombreux paramètres de croissances nécessaires à l'utilisation de l'extension Biomass Succession. Il est ainsi possible de simuler l'impact des changements climatiques sur la croissance des arbres sous LANDIS-II. L'extension Base Fire permet quant à elle de simuler l'occurrence de feux de forêt de manière stochastique dans LANDIS-II tout en contrôlant l'aire brûlée à chaque pas de temps dans différentes zones de feu. J'ai alors paramétré Base Fire pour notre zone d'étude par le biais de simulations de calibration afin de reproduire les aires annuelles brûlées prévues dans les zones de feu homogènes de Boulanger *et al.* (2014). Enfin, l'extension Biomasse Harvest permet de simuler la récolte de bois dans le paysage en appliquant sur des surfaces de forêts données une série de prescriptions pré-définie qui peuvent moduler la biomasse récoltée pour différentes cohortes d'âges.

0.7.3 Protocole expérimental

Le protocole expérimental des trois chapitres est similaire, et se base sur l'élaboration de différents « scénarios » de simulation pour LANDIS-II. Chacun de ces scénarios varie alors différents facteurs, comme l'utilisation de méthode équienne ou inéquiennes via l'extension Biomass Harvest, mais aussi la présence de changements climatiques selon différents futurs possibles (scénarios RCP) ou l'occurrence de perturbations extrêmes (dans le chapitre 3). Les stratégies d'aménagement forestier implémentées ont été comparées en faisant en sorte que la même « cible » de biomasse ou de volume de bois devait être récoltée à chaque pas de temps dans tous les scénarios. Les extensions de récoltes de LANDIS-II ne sont cependant pas capables de récolter une valeur « cible » de biomasse ou de volume de bois, mais seulement une cible de surface de forêt récoltée. Pour compenser à ce problème important, j'ai développé une autre extension pour LANDIS-II nommée « Magic Harvest », permettant de contrôler plus finement le comportement de Biomass Harvest en permettant de récolter une cible de biomasse à chaque pas de temps, et de selectionner les prescriptions à appliquer avec plus de fléxibilité.

Chaque scénario unique utilisé dans LANDIS-II fut reproduit en 5 répliquas, afin de prendre en compte la variabilité stochastique présente dans différents processus de LANDIS-II (p. ex. le déclenchement des feux, la dispersion des arbres, etc.). Je n'ai néanmoins pas utilisé de test statistique au cours des 3 chapitres, car leur utilisation n'est pas nécessaire dans le cadre de l'utilisation de modèles par simulation (White *et al.*, 2014). Mon approche a ainsi pour but de décrire et interpréter la dynamique des différentes variables mesurées entre scénarios au cours des simulations, tout en prenant en compte la variabilité liée aux processus stochastiques. L'ensemble de ces variables réponses utilisées dans cette thèse, ainsi que les facteurs variants entre scénarios, sont décrits dans le Tableau 0-1. Les changements climatiques ne sont pas simulés dans le chapitre 2 pour réduire la complexité des simulations et faciliter l'interprétation des résultats, car le débat du *sharing*/*sparing* peut être étudié sans leur présence. Néanmoins, les scénarios du chapitre 2 font varier l'agrégation des coupes dans le paysage afin de voir si celle-ci peut atténuer la fragmentation causée par les chemins forestiers. J'y fais aussi varier la présence d'un réseau de chemin forestier au début de la simulation, pour voir si le contexte initial du paysage tend à influencer la dynamique des chemins sur le long terme.

Tableau 0-1 : Description des variables explicatives et des variables réponses utilisées durant les 3 chapitres de cette thèse

Chapitre	Variables explicatives (Facteurs variants entre scénarios)	Variables réponses
Chapitre 1	• Réseau routier forestier réel ou réseau routier simulé avec le module FRS (en 2020 dans la zone d'étude)	• Densité de routes dans le paysage • Aire de forêts dans le paysage • Nombre de parcelles de forêts (formées par les chemins forestiers) • *Clumpy* (indice de fragmentation) • Total Core Area (indice de fragmentation) • PAFRAC (indice de fragmentation)
Chapitre 2	• % de la cible de biomasse à récolter par pas de temps récoltée par des méthodes équienne ou inéquienne • Agrégation des coupes dans le paysage • Présence d'un réseau routier forestier initial en début de simulation	• Densité de routes dans le paysage à travers le temps • Coûts de construction et de réparation des routes à travers le temps • Quantité de vieilles forêts dans le paysage à travers le temps (> 90 ans) • Fragmentation des vieilles forêts dans le paysage à travers le temps (via l'indice *Clumpy*)
Chapitre 3	• Scénarios d'aménagement forestier : TRIAD+, TRIAD normale, BAU avec ou sans plantations fonctionnelles • Changements climatiques : stable (climat actuel), RCP 4.5, RCP 8.5 • Perturbation extrême à t = 100 : grand feu de forêt, sécheresse sévère, invasion de Dendroctone du Pin	• Biomasse mature totale dans le paysage à travers le temps • Diversité fonctionnelle moyenne des peuplements du paysage à travers le temps • Résilience de la biomasse mature des peuplements touchés par les perturbations extrêmes via : ○ Leur résistance (perte de valeur par rapport au pas de temps avant perturbation) ○ Leur changement net (différence entre avant la perturbation et après 100 ans de récupération) ○ Leur taux de récupération (inverse du temps nécessaire pour atteindre un changement net de 0 si atteint)

CHAPITRE 1

A LANDIS-II extension for simulating forest road networks

Ce chapitre a été publié en anglais dans la revue scientifique Canadian Journal of Forest Research en Mars 2023. **Le texte ici présent contient cependant quelques modifications mineures recommandées par le jury**. Le text original a été rédigé par les personnes suivantes :

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- Osvaldo Valeria[4,5]
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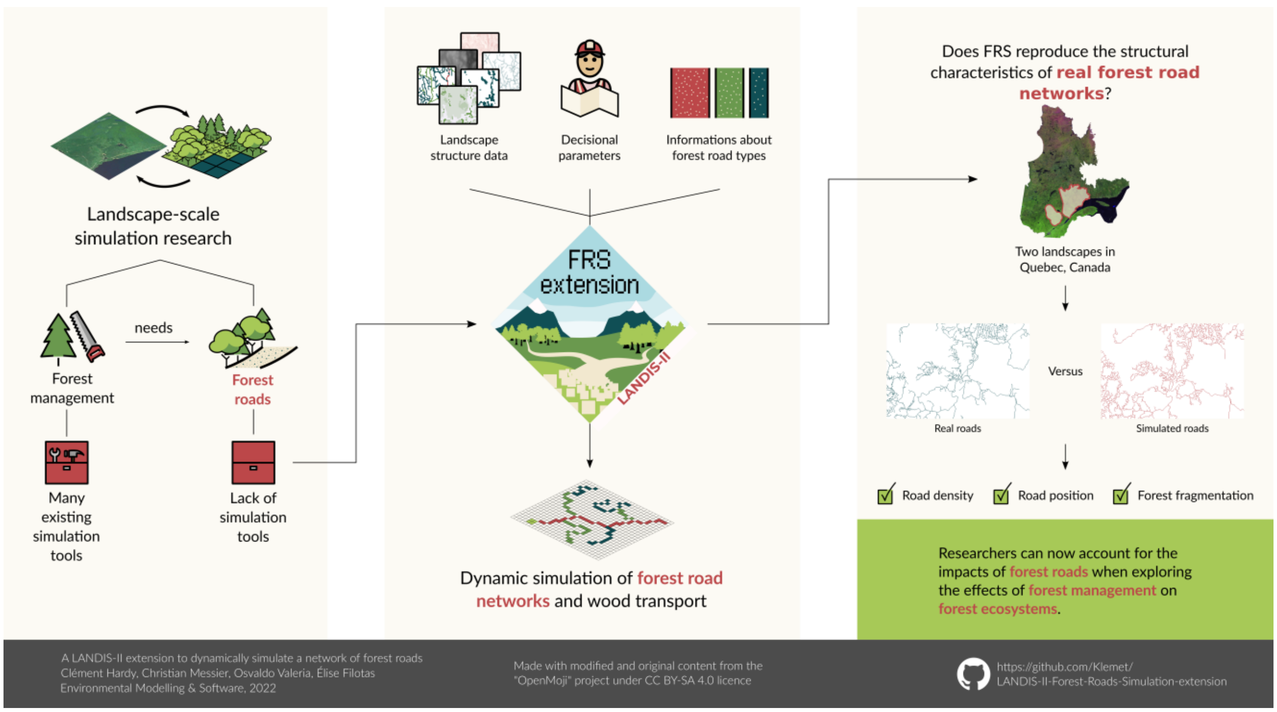
Mots-clés : LANDIS-II; forest roads; road networks; forestry; spatially-explicit modelling; forest ecology

Citation complète : Hardy, C., Messier, C., Valeria, O. et Filotas, E. (2023). A LANDIS-II extension for simulating forest road networks. Canadian Journal of Forest Research. https://doi.org/10.1139/cjfr-2022-0306.

Résumé : Dans ce chapitre, nous décrivons le dévellopement d'une nouvelle extension pour le modèle LANDIS-II, nommée « *Forest Roads Simulation* » (FRS). L'extension FRS à pour but de simuler la construction des chemins forestiers nécéssaires aux opérations forestières simulées dans LANDIS-II. Nous présentons la structure de l'algorithme suivi par l'extension FRS, ainsi que les paramètres et données entrantes nécéssaires à son fonctionnement. Nous démontrons ensuite la capacité de l'extension FRS à

reproduire différentes caractéristiques clés de deux réseaux routier existants : la densité de routes, la position des routes, et la fragmentation du paysage causée par les routes. Nous concluons sur la capacité de cette nouvelle extension à répondre aux besoins existants dans la recherche en écologie forestière en soulignant les nouvelles questions de recherche qu'elle pourra aider à élucider.

Résumé graphique :



1.1 Introduction

For several decades now, roads have been known to have important impacts on ecosystems such as increasing the loss and fragmentation of natural terrestrial and aquatic habitat (Bennett, 2017 ; Li *et al.*, 2003 ; Trombulak et Frissell, 2000 ; van der Ree *et al.*, 2011). Efforts to quantify the ecological effects of roads and reduce their negative impacts have even led to the creation of a separate field of study known as "Road ecology" (van der Ree *et al.*, 2011). Forest roads are a particular type of road characterized by a low traffic volume, traffic mostly in one direction of long and heavy trucks, and often a surface of gravel or local soil (Sessions *et al.*, 2016). Such roads are often abandoned or maintained for other purposes (fire-fighting, recreational activities, etc.) (Hunt *et al.*, 2009 ; Thompson *et al.*, 2021 ; Zhang *et al.*, 2020). Despite having less traffic than main paved roads in more populated areas, forest roads can have many negative impacts on forest ecosystems (Boston, 2016). Indeed, they have been shown to reduce the surrounding macroinvertebrate soil fauna (Haskell, 2000) and the abundance of some beetles (Koivula, 2005), to facilitate the spread of invasive plants (Mortensen *et al.*, 2009), to alter the quality of habitat for some species of birds (Ortega et Capen, 1999) and to fragment the landscape more than clearcuts (Reed *et al.*, 1996). Forest roads also change the behaviour of some large mammals: elk tend to avoid roads (Witmer et deCalesta, 1985), whereas wolves tend to use them, increasing predation pressure on their prey (James et Stuart-Smith, 2000 ; Whittington *et al.*, 2011). It has even been suggested that forest roads could influence the spatial boundaries of forest fires, a key natural disturbance in boreal regions (Narayanaraj et Wimberly, 2011 ; Yocom *et al.*, 2019). In addition, forest roads can be an important source of water pollution and have impacts on the surrounding vegetation and soil conditions (Avon *et al.*, 2010 ; St-Pierre *et al.*, 2021 ; Zhou *et al.*, 2020).

Forest roads are also a substantial operational expense for the forest industry, costing as much as half the harvest operations themselves (Epstein *et al.*, 2006). This cost can, however, vary greatly among regions. For example, in several countries of Europe, costs related to forest roads represent 5% to 10% of the total cost of forestry operations (Toscani *et al.*, 2020), while in Chile these costs can reach 55% of the total operational cost (Epstein *et al.*, 2006). In Quebec (Canada), forest roads represented between 10% and 18% of the total operational cost for the forest industry in 2019, depending on the type of forest harvested (Groupe DDM et MFFP du Québec, 2020). Locally, cost can vary greatly due to soils, sub-soils and slopes (Stückelberger *et al.*, 2006). Approximately half of these costs is allocated to road construction themselves, while the other half is allocated to road repair and maintenance. Forest roads present several advantages, such as easier fire management and protection, and increased access to forests for economic or recreational purposes. Notably, forest roads provide greater access to several essential services to remote communities (e.g., indigenous nations) (Adam *et al.*, 2012). However, because forest roads increase public accessibility to distant areas, their construction, maintenance, and removal is associated with land-use conflicts. Examples of socio-economic tensions related to forest roads include increased road traffic and recreational activities (hiking, fishing, motorized vehicles, etc.) in proximity to or within ancestral forests of indigenous communities, poor consultation with local communities in the planning of forest roads, and loss of access for recreational users following road decommissioning (Adam *et al.*, 2012 ; Bourgeois *et al.*, 2005 ; Hunt *et al.*, 2009 ; Kneeshaw et Gauthier, 2006).

Due to the diversity of economic, social, and ecological impacts associated with forest roads, there is an increased need to develop decision-making tools able to generate different scenarios of forest road network delineation and evaluate their trade-offs at a strategic level. Forest engineers rely on efficient software to plan forest road networks, such as Woodstock Road Optimizer (Remsoft, 2019) or PLANEX (Epstein *et al.*, 2006). These tools are used at the tactical and operational levels and focus on optimizing road design and harvest scheduling to reduce supply cost. However, these tools account for fine-scale constraints such as the skidding methods used or the position of timber landings, and therefore require detailed parametrization (Bont *et al.*, 2015 ; Chung *et al.*, 2004). Moreover, some of these tools are built with expensive proprietary software, making their use costly. In addition, these tools are not designed to explore the long-term and large-scale impacts of forest road networks. Finally, their stand-alone nature makes them inadequate to simulate forest roads along with forest succession, natural disturbances and management options.

Here, we present the Forest Roads Simulation (FRS) extension, an extension for the LANDIS-II model. LANDIS-II (Scheller *et al.*, 2007) is a spatially-explicit and raster-based Forest Landscape Model (FLM) that has gained recognition in past years for its ability to study, through simulations, the interactions between management, natural disturbances and climate change in North American forests (Mina *et al.*, 2020 ; Molina *et al.*, 2021 ; Tremblay *et al.*, 2018). Hence, the FRS extension adds to the ability of LANDIS-II to simulate several key ecological (forest growth, succession, forest fire, etc.) and anthropogenic processes (harvesting or land-use changes), on large spatial and temporal scales. The FRS extension simulates the construction of forest roads at the landscape scale and interaction with the other processes simulated by LANDIS-II (harvesting, succession, natural disturbances). The FRS extension thus differs from previous models simulating optimized forest road networks at the operational and tactical levels to focus, instead, on simulating forest roads at the strategic planning levels. Indeed, forest landscape models, like LANDIS-II and its other extensions, are not aimed at maximizing forestry or economic outputs but rather at determining the interactions between planned forestry strategies (harvest scheduling and operations) and ecological processes (succession and natural disturbances). Consequently, the FRS extension offers the possibility to explore research topics related to forest roads at larger spatial and temporal scales including their effects on landscape fragmentation, carbon balance, animal movement, recreational access, fire management, timber harvesting cost, and social acceptability.

1.2 Methods

1.2.1 Model description

1.2.1.1 Design considerations

In developing the FRS extension our main design objectives were to propose a model for simulating forest road networks at a strategic scale that 1) requires limited and available parametrization; 2) is able to replicate characteristics of real forest road networks (e.g., density of roads, extent and distribution, fragmentation level, and costs); 3) does not rely on low-level features of forest roads that have limited influence on the landscape-scale network (e.g., skidding methods, spacing and positioning of landings where the timber is stacked, etc.; see Chung *et al.* 2004 for examples); and 4) has a competitive performance (in run time) regarding other LANDIS-II extensions (i.e., under 30 min per time step; Sturtevant *et al.* 2004).

1.2.1.2 Goal of the extension

Forest road networks are built while attempting to minimize expenses (such as salaries of construction workers, equipment rental, cost of surface material, as well as costs related to the protection of the environment; Heinimann, 2017), with lower standards than regular roads despite supporting heavy trucks (Légère, 2001 ; Ryan *et al.*, 2004). They are also developed sequentially rather than planned over a long-term horizon in an optimal manner, as forest industries construct new roads depending on available budget. Therefore, to capture the characteristics of existing forest road networks, the FRS extension focuses on designing roads to access harvested areas with a minimal cost of construction, but without necessarily minimizing the costs of construction across the entire network developed during the full planning horizon. Consequently, the main task of the FRS extension is to compute least-cost paths between a set of scheduled harvested areas to specific locations on the landscape where timber needs to be transported (e.g., main road network, sawmills). Note that the FRS extension does not solve what is called the “integrated forest harvest-scheduling model”. This refers to the more complex problem of optimizing the forest road network design together with the choice of areas to be harvested (Heinimann, 2017 ; Naderializadeh et Crowe, 2020) – a problem solved by operational software such as Woodstock or PLANEX (Epstein *et al.*, 2006).

Hence, the goal of the FRS extension is to solve the problem of connecting multiple known areas – a problem known as the *Multiple Target Access Problem* (MTAP) (Heinimann, 2017). The MTAP is a predecessor to the larger problem of integrated forest harvest-scheduling and has been used several times in the context of forest roads (Heinimann, 2017 ; Shirasawa et Hasegawa, 2014). In essence, it can correspond to a minimum spanning tree (MST) problem (Shirasawa et Hasegawa, 2014). While the MTAP is solvable, finding the optimal solution becomes extremely difficult for many targets (i.e., places that must be connected by roads such as harvested areas, sawmills, and the main road network) (Shirasawa et Hasegawa, 2014). For a landscape covering millions of hectares, solving the MTAP becomes impossible, or requires an impractical amount of time. As an example, a fine-scale operational model can find optimal solutions to problems similar to the MTAP in up to 8 hours of computation for an area of 4.3 km^2, making it inappropriate for landscapes used in simulation research, which can easily reach 10 000 to 100 000 km^2 (Bont *et al.*, 2015).

The complexity of the MTAP in the context of forest road network design has led to the development of several heuristics (i.e., assumptions or simplifications) to reduce solving time (Shirasawa et Hasegawa, 2014). However, using a heuristic implies that solutions are often approximations departing from the optimal solution to the MTAP. Hence, the choice of a particular heuristic can remarkably alter the resulting road network compared to the optimal MTAP solution (Anderson et Nelson, 2004). Here, we developed the FRS extension by selecting three heuristics that are intuitive, reduce the extension's complexity for users, and improve execution time. These heuristics are also used in professional software like Remsoft Road Optimizer (Remsoft, 2019). The three proposed heuristics rely on the same concept of "breaking" the MTAP into a set of Single Target Access Problems (STAP), i.e., the problem of optimizing the path between two points. STAPs are in turn easy to solve via simple path-finding algorithms such as the Dijkstra algorithm (Anderson et Nelson, 2004). Breaking down the MTAP into STAPs also replicates the sequential process used in the forest industry when constructing forest road networks. We propose two heuristics ("closest first" and "farthest first") used by Anderson et Nelson (2004) and tested by Shirasawa et Hasegawa (2014), as well as a "random" heuristic. Heuristics are described in section 1.2.1.5.2.

1.2.1.3 Structure

The FRS extension is an extension of the LANDIS-II model. LANDIS-II is itself an open-source project coded in C# made up of a core and a catalog of different extensions (Figure 1-1) (Scheller *et al.*, 2007). The FRS extension comes with its own documentation and Github repository, allowing any user to learn how to use it or to modify it if needed, with detailed information available in Hardy (2021).

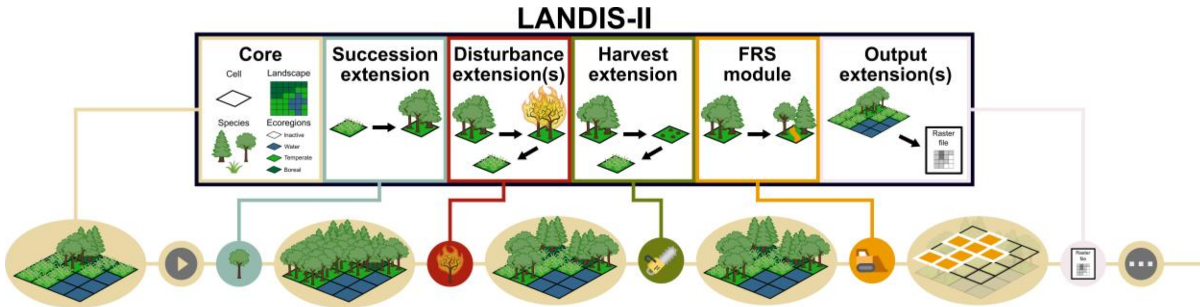


Figure 1-1 - The LANDIS-II modelling structure. The forest landscape is simulated by the periodic and successive intervention of different extensions: succession and disturbances (e.g. harvesting, forest fire, insect outbreaks). The Forest road simulation extension follows the harvesting extension.

The FRS extension operates after the harvest extension in LANDIS-II. Note that the FRS extension is compatible with any other extension of LANDIS-II, as it edits and reads a distinct spatial dataset that we name the "road landscape", without editing the forest landscape itself. The road landscape consists in a matrix of cells (or pixels): each cell is either empty or occupied by a road. In the latter case, the road category is also provided (primary, secondary, tertiary; see section 1.2.1.5.6). Moreover, the road landscape contains locations named "exit points" for the timber, which are defined by the user. These cells correspond to transit locations from which timber is further transported (e.g., main paved road network, train station), or final destinations where timber is processed (e.g., sawmill). Note that a cell containing a road remains available for harvesting. This assumption was made given that the widest forest roads rarely exceed 30 m and therefore remains smaller than the usual grain size used in LANDIS-II models (e.g., 1 ha).

1.2.1.4 Creation of the cost map

During the initialization phase of LANDIS-II, before the simulation starts, the FRS extension prepares the "cost map" that will be used by the least-cost path algorithm. The "cost map" is a matrix with the same dimensions as the simulated landscape that provides the cost for road construction within each cell. The cost map can take into account six types of landscape features known to influence the cost of forest road construction: existing roads (prevents the construction of new ones), elevation (slows down construction or requires adaptations), topographic obstacles (requires detours, e.g. cliffs or breaks), lakes and rivers (requires the construction of bridges), streams (requires the construction of culverts), and soils (influences the material needed, the amount of work required, and cost) (Figure 1-2). These features need to be provided by the user in the form of raster files, and associated cost parameter values in a .txt file. However, the user must provide information about existing roads and elevation for the extension to function; the other features are optional but recommended to improve the extension predictions (Figure 1-2). All raster files must have the same resolution and extent as the other raster files employed by LANDIS-II's core extension. The user must also input the "basal construction cost", corresponding to the minimal cost for building a forest road across a distance equal to the size of a cell and under ideal conditions (e.g., flat terrain, no water body, and good structural soil characteristic) in units of currency.

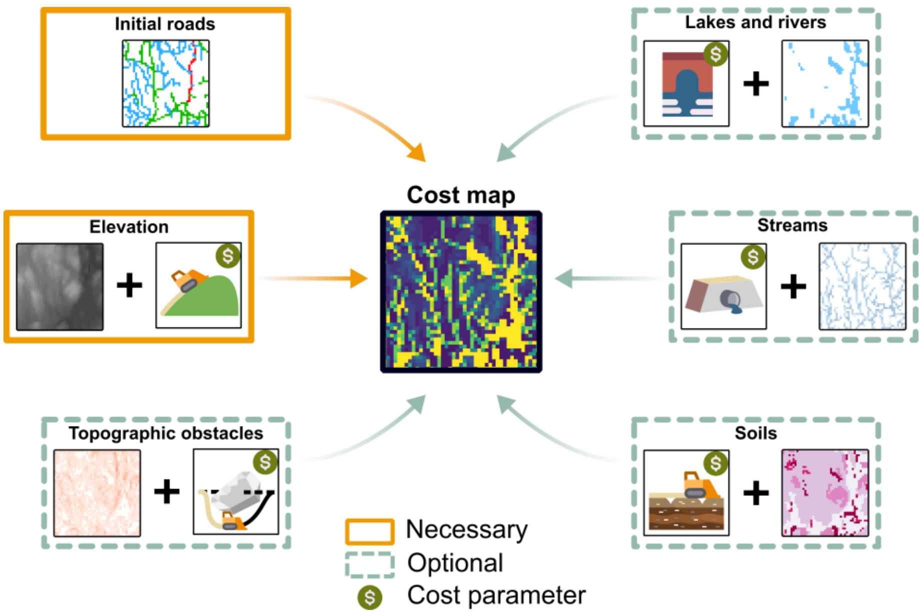


Figure 1-2 - The cost map constructed by the FRS extension during the initialization of LANDIS-II depends on six landscape features: the initial road network, the elevation (slope), the topographic obstacles, lakes and rivers, streams and soils. Features and their associated costs are fed to LANDIS-II in the form of raster and parameter text files at initialization step.

The total cost of construction of a forest road (Equation 1-1) on a single cell, C_{total}, is zero when a road already exists on the cell, otherwise it is calculated by adding all considered costs together:

$$C_{total} = R_{category} D_{obstacles} \left(C_{basal} + C_{slope} + C_{bridge} + C_{culvert} + C_{soil} \right) \quad \text{Equation 1-1}$$

where C_{basal} is the basal construction cost. C_{slope} is the additional cost due to the elevation. It is based on the average slope between the elevation of a cell and that of its immediate neighbors. C_{slope} is provided in a user-defined lookup table where an additional cost is associated to each class of slopes. C_{bridge} is the cost of constructing a bridge section across a cell that contains a water body. This cost does not depend on the presence of water bodies in surrounding cells. Hence, the total cost of building an entire bridge across a water body corresponds to the sum of costs for all cells along the bridge. $C_{culvert}$ is equal to the average cost of installing a culvert multiplied by the estimated number of culverts that would need to be installed within a cell. This estimation is made by using the total length of streams in a cell since several meandering streams may be present. We then assume that the number of needed culverts equals the ratio between this total length and the cell's diagonal length rounded to the next higher integer. For example, if 150 m of streams is present in a 1ha resolution cell (100m x 100m), we estimated that two culverts would be needed. C_{soil} corresponds to the additional construction cost due to the local soil conditions. For example, C_{soil} can represent the cost of transporting gravel from afar or the cost of rock blasting. Finally, $D_{obstacles}$ is a multiplicative factor representing the increase in the overall cost of road construction due to topographic obstacles that require road detours within a cell. Whereas C_{slope} represents costs due to a change in the average elevation with the surrounding cells, $D_{obstacles}$ encompasses fine-scale topographic variabilities within the cell (holes, cliffs, breaks, etc.). In our case, we defined $D_{obstacles}$ using the number of contour lines in a cell representing a 10m change in elevation, and derived from a digital terrain model. A high number of contour lines would imply either a steep but homogeneous terrain, a flat but heterogeneous terrain, or a mix of both. A lookup table parameterized by the user assigned a multiplicative factor to different classes of contour line quantity (e.g., $D_{obstacles}$ = 1, 2 or 3 for 0 – 1, 2 – 4, and 5 and more contour lines per cell, respectively.). The precise way in which these costs are calculated by taking into account both the topographical maps and the user-defined parameters is described in the user manual of the model (Hardy, 2021).

These costs must be parameterized relative to a "reference" road category (e.g., primary roads). The estimation of the cost of construction for other road categories are then computed using the multiplicative value $R_{category}$ ($R_{category} = 1$ for the reference road category, $R_{category} < 1$ for less costly categories, and $R_{category} >$

1 for more costly categories). Hence, the multiplicative value $R_{category}$ adjusts the total cost as would be expected when building a road of larger or smaller width.

The values for these cost parameters can be obtained in different ways depending on data availability in the study region. The best estimates can be found from databases containing the measured costs of construction of existing forest road sections, from which prices per unit of distance can be extracted. In regions where such databases are not easily accessible to the public, parameterization could require expert opinion from forest engineers in logging companies or governmental agencies (Hardy, 2021). However, irrespective of the methodology used, soil types, elevation and construction costs are likely to be heterogeneous within a cell for higher spatial resolution (Stückelberger *et al.*, 2006).

Following the creation of the cost map, the FRS extension checks whether the initial road network provided by the user contains road cells that are not connected to any exit point cells (either directly or via other roads). The presence of unconnected roads can be attributed to errors in the detection or registrations of roads, or by the destruction or deactivation of roads. The FRS extension then proceeds by creating new roads to connect those isolated road cells to exit point cells. This step completes the activation phase. During the simulation of the LANDIS-II model, the cost map is updated every time a road is built or destroyed, making it dynamic.

1.2.1.5 During the simulation

During a simulation of the model, LANDIS-II activates each extension selected by the user successively according to their respective return time (Figure 1-1). As LANDIS-II can simulate disturbances that vary in their recurrence (e.g., harvesting every 5 years and insect outbreaks every 40 years), we recommend parameterizing LANDIS-II so that the FRS extension operates immediately following the selected harvest extension, and with the same recurrence.

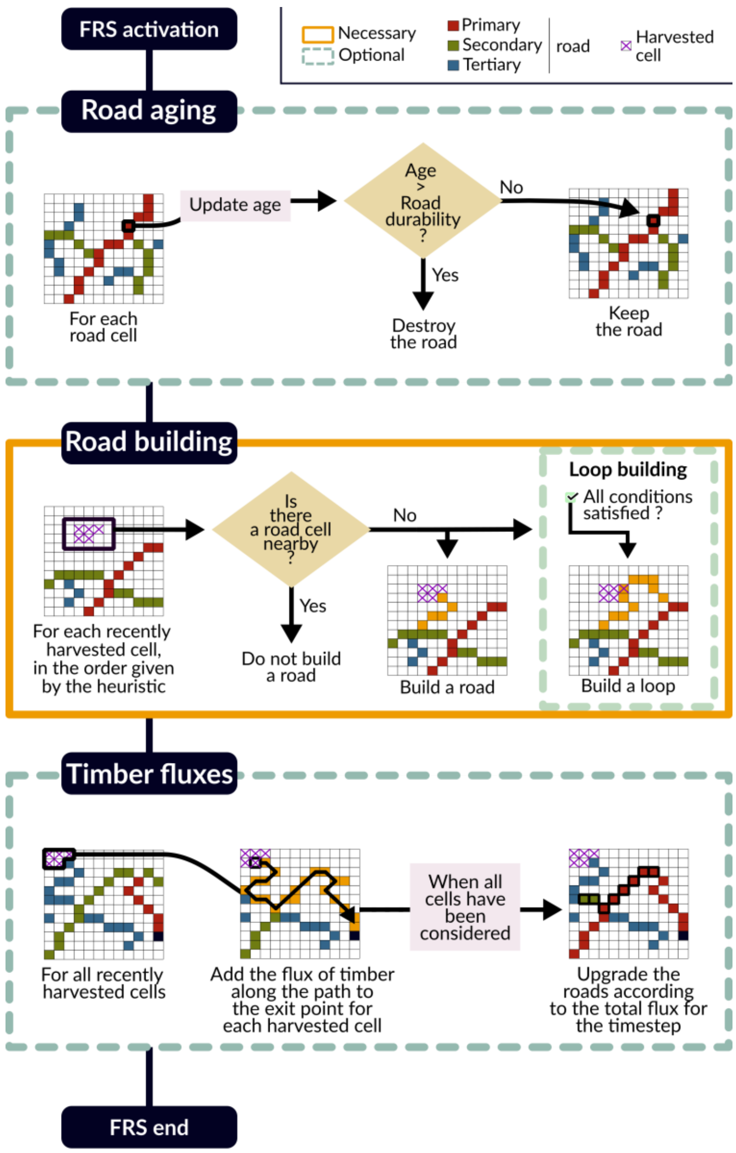


Figure 1-3 : Flow chart of the different stages followed by the FRS extension at time step t during a simulation in LANDIS-II: road aging (top), road building (middle), simulation of timber fluxes (bottom).

1.2.1.5.1 Road aging

When the FRS extension is activated at a time step *t*, it starts by increasing the age of the forest roads in the landscape following the selected recurrence (if the option has been selected by the user) (Figure 1-3, top). Indeed, each time a new road is "built" on a cell, the cell is given the "age" value 0. At the following activation of the FRS extension, the age of cells containing a road is updated by adding the time since its previous activation. If the age of a cell exceeds the maximum age of the road (determined by its road type;

see below), then the road is considered unusable, and disappears from the landscape. This aging mechanism allows the landscape to escape the legacy of its initial state, as forest roads dynamically appear and disappear with time. However, crucial road cells can be considered repaired if they are re-created by the extension during the same time step. Note that in this initial version of the model, repairing roads in this way costs the same as entirely rebuilding them. Moreover, permanent forest roads, such as main arteries, can be simulated by assigning them a high longevity. The age of all roads in the initial road network is currently initialized at 0. Initialization to real age values is nonetheless possible if these data are available.

1.2.1.5.2 Ordering of the recently harvested cells

Following road aging, the FRS extension identifies all cells harvested by the harvest extension since its last activation and orders them in a list according to the heuristic chosen by the user (Figure 1-3, middle). Three heuristics are currently available: closest first, farthest first, or random. The closest first and farthest first heuristics order the recently harvested cells according to their Euclidean distance to the nearest existing road, while the random heuristic orders them randomly. The selected heuristic influences the order in which the forest roads will be created, and thus the shape of the resulting road network. While the three heuristics are available to the user, previous studies have shown that the closest first heuristic replicates the progression of a forest road network through a landscape and creates a network that is closest to an optimal one (Shirasawa et Hasegawa, 2014).

1.2.1.5.3 Construction of new roads

Road construction proceeds by considering each harvested cell one by one according to their ordering. First, the distance between the harvested cell and the closest existing road is computed. If this distance is less than the skidding distance parameter (Table 1-1), no new road is constructed as we consider that timber will simply be skidded toward the nearest road (Figure 1-3, middle). If the distance is greater than the skidding distance, the Dijkstra algorithm (Dijkstra, 1959) is used to solve the STAP. More precisely, the Dijkstra algorithm solves the “single-source many-targets shortest path problem” (SSMTSP), a special case of STAPs that finds the path between the harvested cell and one of multiple potential arrival points on the road network (Bast *et al.*, 2003). The cost of the path is the sum of the construction costs, derived from the cost map, for each cell along the path. This cost is recorded by the extension for each road segment, and the sum of these costs is made available to the user in an output file.

Table 1-1 : Description of the most important parameters of the FRS extension.

Parameter name	Optional	Units	Description
Time step	No	Years	Number of years between two activations of the FRS extension during a LANDIS-II simulation.
Skidding Distance	No	Meters	Maximum distance from a recently harvested cell to the nearest existing road across which timber is skidded to the existing road.
Looping Behavior	No	Boolean	Activates the looping algorithm
Looping – Minimum Distance	Yes	Meters	Size of the neighborhood around a recently harvested cell that must be free of road cells for a loop to be constructed.
Looping – Maximum Distance	Yes	Meters	Size of the neighborhood around a recently harvested cell that must contain at least two road cells for a loop to be constructed.
Looping – Maximum Road Density	Yes	Percentage	Maximum number of road cells in the neighborhood of a recently harvested cell for a loop to be constructed. The size of the neighborhood is defined by the Maximum Looping Distance.
Looping – Maximum Cost Ratio	Yes	No unit	Maximum ratio between the cost of the second segment and the cost of the first segment of the loop.
Looping – Probability	Yes	No unit	Probability that a loop is constructed if all the other loop conditions are respected.
Basal Distance Cost	No	Monetary units	Minimum cost to build a road across a single cell.
Coarse Elevation costs	No	Monetary units	Table of additional construction costs due to the elevation for different ranges of slope value.
Fine Elevation Costs	Yes	Monetary units	Table of multiplicative construction costs due to detours needed to avoid topographic obstacles for different ranges of fine elevation values in a cell.
Bridge Cost	Yes	Monetary units	Average cost to build a bridge across a single cell.
Culvert Cost	Yes	Monetary units	Average cost to build a culvert across a single cell
Soil Costs	Yes	Monetary units	Additional construction costs due to the type of soil present in the cell.
Simulation of road aging	No	Boolean	Enables road aging.
Simulation of the wood flux	No	Boolean	Enables the simulation of the wood flux through the roads.
Lower and Upper Wood Flux Thresholds	Yes	Age cohorts transported by year	Associates ranges of wood flux values to types of forest road (e.g. primary, secondary, etc.)
Multiplicative Cost Values	No	No unit	Indicates how the cost of construction of a given road type is increased or decreased compared to a reference type.
Maximum Ages Before Destruction	Yes	Years	Indicates how long a road of a given type can last without any repairs or upgrades before it gets destroyed by wear.

In our model, the Dijkstra algorithm is allowed to move in a Moore neighborhood (nearest and next-nearest neighbors), with the cost of movement being weighted by the distance between cell centroids to avoid directional bias (Holland *et al.*, 2007). We kept this simpler philosophy since approaches that consider more angles of movement (Stückelberger *et al.*, 2007) generally result in increased computation time and complex interpretation of road configuration and costs.

1.2.1.5.4 Option to create road loops

The FRS extension also includes a "loop" algorithm to better replicate the occurrence of loop structures observed in existing road networks. A loop appears when more than one road connects two points of the landscape and surrounds a patch of forest. Loops occur for a variety of reasons and can be planned or not. For example, the harvested wood may need to travel to different mills along different roads. In addition, temporary disturbances such as a forest fire or flood that obstructs an existing road may require the construction of a new road leading to an eventual loop. Once constructed, they can increase the accessibility of certain forest areas for forestry vehicles or timber trucks, or increase the resilience of the network (i.e., the existence of multiple paths to a given place, increases the chances that one path will remain available if disturbances such as fire or flood obstruct other paths). However, it is hard to isolate a particular set of rules or probabilities that replicates the decision process by forest road engineers to create loops in the network.

The loop algorithm of the FRS extension creates a loop stochastically by building a second road segment, with a given probability, from a recently harvested cell to the rest of the network, in contrast to building a single road by default. Four conditions must be met for a loop to be constructed (Table 1-1): 1) no existing road cell should be too close to the harvested cell of interest (Minimum looping distance); 2) at least two existing road cells should be present in the vicinity of the harvested cell (Maximum looping distance); 3) a maximum number of existing road cells in the vicinity of the harvested cell should not be exceeded (Maximum road density); and 4) the second road to be built should not be too costly (Maximum cost). The looping algorithm therefore controls the size, distribution and quantity of loops without having to explicitly simulate the complex decisions of forest engineers. More details on the loop algorithm can be found in the user guide of the FRS extension (Hardy, 2021).

1.2.1.5.5 Planned return to harvested cells

Some harvest prescriptions require a return to harvested cells for a second cut (e.g., shelterwood), or for repeated, periodic cuts (e.g., selection system). However, if road aging is activated, a road constructed to reach a particular cell may have disappeared at the time of the next prescribed harvesting. Hence, when a new road is built to a recently harvested cell, the FRS extension computes and selects the least expensive option between 1) building a low-cost road that may need to be re-built when returning to the cell, or 2) building a higher-grade road that will last for the next access. To that end, the FRS extension takes advantage of the fact that the user can input different road categories associated with a different longevity (e.g., primary, secondary, or tertiary road).

1.2.1.5.6 Computation of the wood flux

Once all forest roads necessary to access recently harvested cells have been created, the FRS extension simulates the timber flux transported through the road network (if the option has been selected by the user) (Figure 1-3, bottom). The timber from each harvested cell is transported across all road cells that connect its cell of origin to an exit point in the landscape. The timber flux thus “flows” across the road network (as water does in a watershed) and increases upon converging with other roads. As a result, the timber flux in a given road cell at a particular time step is simply the sum of the amount of timber going through this road cell. The amount of timber leaving a recently harvested cell is estimated by the number of age cohorts (i.e., groups of trees of the same age) harvested in the cell at the current time step. We used age cohorts as they represent a unit shared between different extensions of LANDIS-II, allowing for more compatibility. Moreover, since an increasing amount of harvested age cohorts necessarily implies an increasing amount of harvested biomass, age cohorts constitute a good proxy for the amount of harvested timber.

The FRS extension can then use the timber flux to inform of potential “upgrade” to a higher category of a forest road. Each road category can accommodate a timber flux up to a maximum level above which a higher category is required (primary > secondary > tertiary, etc.) (Nevečerel *et al.*, 2007). Integration of road category into FRS is important for studies that focus on the impact of road traffic on fauna or estimate degradation over time (Girardin *et al.*, 2022), for example. Updating the category of a road also involves a higher cost. This cost of upgrade is computed for every road cell, adding to the costs of maintenance of the forest road network ($R_{category}$ in Equation 1-1). Additionally, when a road cell is upgraded, its age is reset to 0. An example of how to obtain the threshold of wood flux (in number of age cohorts) for the different road types through a calibration procedures is presented in Appendix B.6.3.4.

1.2.1.6 Computing performance

We improved the computing performance of the extension by using two open-source C# NuGet packages. More details about these packages, as well as how they improved the computing performances of the extension are detailed in Appendix A.1. The employed packages reduced the computing time of the FRS extension to less than a minute when executing a single iteration on a landscape of 4 million active cells and about 5% harvested cells, with a 2.60GHz Intel i7 4 cores-CPU and 16GB of RAM.

1.2.2 Model testing

We tested the ability of the FRS extension to reproduce key structural characteristics of forest road networks: the density of roads in the landscape, the fragmentation level of the landscape, and their approximate location. These road network properties are important indicators of the impacts that roads have on forest landscapes (Bennett, 2017 ; Forman, 2005 ; Karlson et Mörtberg, 2015). We limited our tests to these main properties due to the difficulty of finding data regarding existing forest roads, their construction date, their precise construction cost, and their original destination. As such, other forest road properties simulated by the FRS extension (e.g., road age, road category, road costs and the precise location of roads) could not be validated, as reliable estimates for these measures could not be obtained from existing data in our study areas.

1.2.2.1 Study area

We tested the forest road network simulated by the FRS extension in two different forested regions in the province of Quebec, Canada (Figure 1-4), the Mauricie and the Côte-Nord regions. The region in Mauricie covers 5 million hectares and extends above the 46° parallel north from the mixed to the boreal forests. Only a small proportion of its surface (< 2%) is non-forested – mainly agricultural areas in the south – and there is a high quantity of "main" paved roads (e.g., highways). It also contains many lakes, rivers and streams, with the Saint-Maurice River being the largest and dividing the landscape into two parts. The region in Côte-Nord covers 13 million hectares going from N48° to N52°, and extends across the boreal forest. Whereas multiple paved roads are present in its southern part, only a single paved road (route 389) reaches the northern limit of this region, representing a vital artery of transport through the area. The Manicouagan River and the Outardes River run north to south in the Côte-Nord region. This region is more hilly and mountainouswith around 15% of its area at an elevation above 1,000 m, compared to 5% for the

Mauricie region. Both regions contain a high density of forest roads resulting from decades of forest harvesting, with 13.5 and 6.73 meters of roads per hectare in the Mauricie and Côte-Nord regions, respectively.

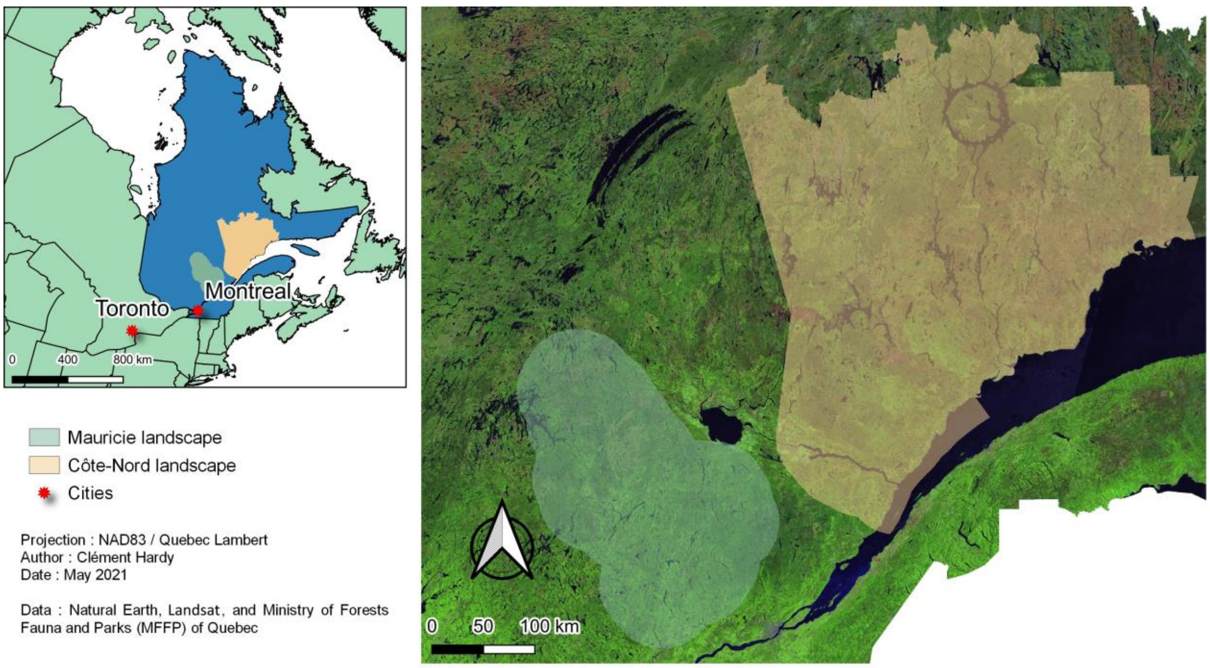


Figure 1-4 : Map of the two landscapes used to test the FRS extension, in Quebec province, Canada. Polygons of countries (top left) and cities locations are both from the Natural Earth dataset (https://www.naturalearthdata.com/). Satellite imagery is from the Landsat project, and accessed through the data portal of the Ministry of Forests, Fauna and Parcs of Quebec (https://geoegl.msp.gouv.qc.ca/ws/mffpecofor.fcgi?request=GetMetadata&layer=lsat_mos2020). Study area extent for the Côte-Nord region (orange) comes from Labadie et al (2023).

The road networks were simulated to connect existing areas of forestry operations to the main road network in their respective regions. Areas of operations were identified from Quebec's 5th forest inventory which includes forest operations at times dating back as far as the beginning of the 20th century (MFFP, 2018b). We compared the simulated networks with those contained in the governmental database, "AQReseau+", of all terrestrial transport routes in Quebec, including forest roads (Gouvernement du Québec, 2015). The LANDIS-II model and the FRS extension were parameterized for both regions. The Mauricie region, was modeled using a 100 m x 100 m cell resolution and the Côte-Nord region was modeled using a 250 m x 250 m cell resolution.

1.2.2.2 Parameterization

For both case studies, the FRS extension was parameterized with data provided by the Ministry of Forest, Fauna and Parks (MFFP) of Quebec. Terrain data (elevation, hydrology and soils) are publicly available through "Données Québec", and data related to cost parameters (e.g., influence of elevation on construction cost, cost of culverts, etc.) were acquired from MFFP internal studies and expert opinions. Cost parameters obtained from expert opinions were then adjusted, within the range of existing values, to obtain the best concordance in replicating the characteristics of the existing forest road network.

1.2.2.3 Rasterization and data cleaning

The FRS extension generates the road network in raster format. Therefore, the first step in our comparison protocol was to convert the AQReseau+ data, consisting of shapefile objects (Figure 1-5a), to a raster layer. This rasterization process was performed using an 8-neighbors rule. More precisely, we rasterized the vector roads going from any point A to a point B by keeping only the raster cells on the shortest path from A to B. This path was based on a movement in 8 directions through cells that intersected the vector road (Figure 1-5b). This effectively kept the location of the existing roads. While other rules can be used to rasterize line strings, such as "keep all cells intersecting the line" or "keep all cells with a centroid close enough to the line", they may over- or under-estimate the presence of roads. The chosen rule confers an adequate compromise between insuring that rasterized road segments are free of discontinuities while limiting the number of redundant cells. This, in turn, reduces potential biases coming from the rasterization process.

Figure 1-5 : The 5-step process by which the vector data from the AQReseau+ database were transformed in order to be compared with the outputs of the FRS extension: a) the initial vector data, b) the rasterization of the road network, c) the removal of forest roads that did not lead to forestry operation areas, d) the removal of forestry operation areas not reached by roads in a radius of 250m, and e) the final, transformed data. Features that were removed during the transformation process appear in red in the illustration.

Following the rasterization process, we cleaned the obtained raster map by eliminating existing roads that were irrelevant for the validation protocol (Figure 1-5c). Indeed, the FRS extension can only simulate forest roads leading to areas of forestry operations, as it is currently not designed to create roads for purposes other than timber transport. On the other hand, many forest roads reported in AQReseau+ did not lead to any registered harvested areas. The construction date of some forest roads and their associated harvested areas were also unreliable or nonexistent. Therefore, in the raster map of existing roads, we only kept roads connecting harvesting zones to the main paved road network and removed all other roads. To that end, we first gathered the areas of all the forest operations ever recorded by the MFFP (e.g., cuts, thinning, plantations, etc.) in the two landscapes. We then retained all road cells within these areas as well as the road cells needed to connect them to the main paved road network through the shortest path (Figure 1-5c). This procedure eliminated 9.6% and 9.8% of all road cells in the Mauricie and the Côte-Nord regions respectively.

Moreover, the FRS extension simulates roads to every harvested pixel associated with forestry operation areas. As such, harvested cells will always be accessible by roads following a simulation of the FRS extension. However, the AQReseau+ database is not entirely complete, as some forest roads are unregistered and undetected, and hence absent from the record even if the presence of a forestry operation areas implies their existence. Therefore, we removed harvested cells that were unconnected by a road or at a distance greater than 250 m away from an existing road, which corresponds to a selected maximum skidding distance for the harvest timber (Figure 1-5d). This procedure eliminated 24% and 15% of the pixels where a forest operation was recorded in the Mauricie and the Côte-Nord regions respectively. From this final step, we obtained a rasterized road network composed of roads leading to forestry operation areas, and forestry operation areas connected to roads (Figure 1-5e).

1.2.2.4 Measurements

We simulated forest roads with the FRS extension ten times for each study region to account for the stochasticity present in the loop algorithm, and compared the resulting road networks with the existing road network in each region. While LANDIS-II is a dynamic model which can simulate the evolution of a landscape on hundred of years (see section 1.2.1.3), these simulations were done on a single time step where harvesting and road building was done in tandem. Our results thus only concern the results of this single time step, as we wanted to only compare snapshots of real and simulated road networks rather than their dynamic through time. We measured key structural characteristics of road networks using standard metrics from landscape ecology that capture their impacts on forested landscapes: road density, number of forest patches, and level of fragmentation generated by roads. These metrics provide a quantitative assessment of the extent, density, and spatial distribution of road networks, allowing for a reliable discrimination between the real and simulated networks. We computed the total area of forest in the landscape as well as four indices of fragmentation of forest area: the number of forest patches delimited by non-forest areas (N), the *Clumpy* index (*Clumpy*), the Total Core Area index (TCA), and the perimeter-area fractal dimension index (PAFRAC) (McGarigal *et al.*, 2012) (equations for the indices appear in Appendix A.2). These distinct indices were chosen because they represent different aspects of fragmentation. The number of forest patches indicates the degree of subdivision of a forest landscape. Inversely, *Clumpy* captures the level of aggregation of habitat patches. TCA quantifies the total amount of core habitat area in the landscape where the core of a patch is the interior area at a given distance from its edge; here, we used a distance of 1 pixel (i.e., 100 m for the Mauricie region and 250 m for the Côte-Nord region). Finally, PAFRAC represents the fractal dimension of the habitat patches and is used to measure the complexity of their shape (Wang *et al.*, 2014). Moreover, TCA is highly correlated to habitat amount, while *Clumpy* and PAFRAC are not (Wang *et al.*, 2014). All indices were computed using the R package *landscapemetrics* (Hesselbarth *et al.*, 2019). We assumed that all forest pixels in our landscape, regardless of their age and species composition, consisted of habitat.

In addition, we evaluated the superposition of simulated and existing roads at a local scale. This approach allowed us to assess the capacity of the FRS extension to simulate the presence of existing roads without requiring them to occupy the exact same location. To that end, we computed the percentage of simulated road pixels that was located 500 m or less from existing road pixels. The 500 m distance was chosen to be neither too permissive nor restrictive when comparing the location of existing and simulated roads. We carried out this analysis for roads within and outside forestry operation areas, as road segments outside management areas had fewer constraints on their position and therefore represented a greater challenge for the extension to replicate.

1.3 Results

Measures of road density and fragmentation indices are presented for each study region as the percentage error between the variables measured on both simulated and existing road networks (Figure 1-6). Our results show that the FRS extension was able to reproduce the equivalent road density of the existing road network (9.59% in Mauricie, 11.11% in Côte-Nord) with small differences in both regions (error less than to 1%) (Figure 1-6). Moreover, most fragmentation patterns were reproduced with relatively small differences in the Mauricie region (error less than 10% for all indices; Figure 1-6) and in the Côte-Nord region (error less than 10% for TCA and PAFRAC), despite *Clumpy* presenting an error of around 13% (Figure 1-6). *Clumpy* and TCA were both under-estimated by the FRS extension in the two regions, indicating that simulated roads tended to fragment forest patches more than existing roads do (Figure 1-6). On the other hand, PAFRAC was higher in both regions under simulated roads than in the AQReseau+ database, indicating that the shape of habitat patches was more complex and less smooth when roads were generated by the FRS extension (Figure 1-6).

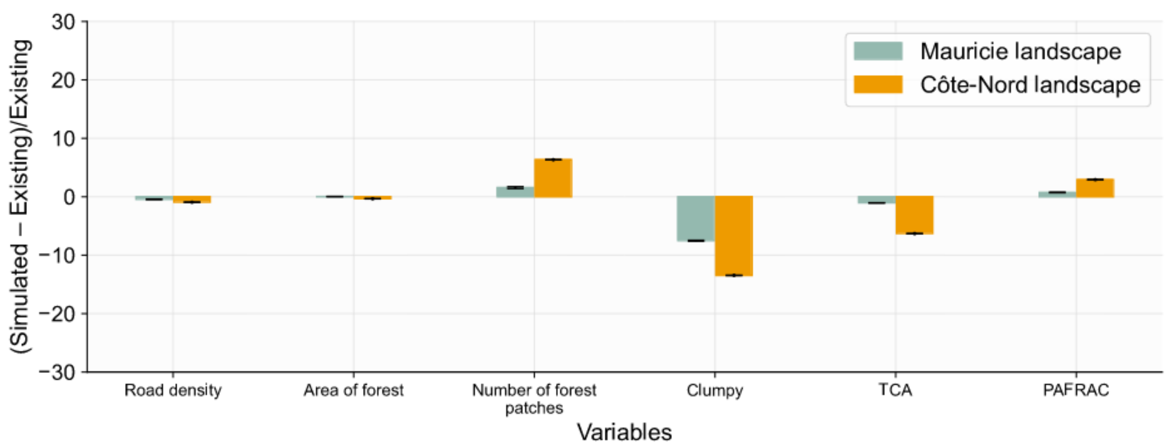


Figure 1-6 : Percentage error (%) between the ten simulated road networks and their existing counterparts, for the road density, the total area of forest, the number of forest patches and the three fragmentation indices for the Mauricie region (green) and the Côte-Nord region (yellow). Black bars indicate the standard error of the mean.

Table 1-2 shows the local correspondence between the location of simulated and existing road pixels. In both regions, the percentage of simulated road pixels located at a distance smaller than 500m from existing roads pixel was high when all road pixels were considered (approximately 90%). However, this percentage decreased by about 40% when only the roads located outside forestry operation areas were considered.

Table 1-2 : Percentages of simulated road pixel that are 500 m or less from existing road pixels for the two landscapes. Standard error is indicated for the ten simulations.

	For all roads	For roads outside of managed forest areas
Mauricie landscape	97.1%±0.0	60.5%±0.2
Côte-Nord landscape	89.3%±0.1	44.7%±0.1

1.4 Discussion

Overall, the FRS extension was able to reproduce the characteristics of two AQReseau+ database forest road networks with small differences. In particular, the extension was able to reproduce the density of roads in the landscape, their fragmentation of forest habitat, and their position. The replication is remarkable, given the complexity of the interactions between different agents and factors at play in the construction of forest roads: methodology and decision of forest road engineers, existing legislation, forest planning, future forestry activities, presence of protected areas, and so on. Our extension considered some of these factors by implementing cost layers based on landscape features, such as soils, and topographic and hydrographic obstacles, along with road categories based on food fluxes, road aging, and the possibility of building loops. The implementation of these factors can explain the performance of the FRS extension in replicating existing road networks.

Our results show that our extension reproduced generally well the four measured aspects of habitat fragmentation: subdivision (Number of patches), aggregation (*Clumpy*), core area (TCA) and shape (PAFRAC) of forest patches. Some variations between the existing and simulated landscapes were observed, especially for *Clumpy*, even though the FRS extension successfully reproduced the density of road pixels and the total amount of forest pixels in the landscapes (Figure 1-6). Hence, the deviation in the fragmentation of forest patches is not caused by differences in the quantity of forest pixels, but by differences in their distribution across the existing and simulated landscapes. A qualitative comparison between the existing road network in a section of the Mauricie region (Figure 1-7a) and one simulated road network in that same section (Figure 1-7b) provides insight into the differences in the measured fragmentation properties. We observe that in the simulated network, roads maintain a regular spacing across the landscape, which is explained, in part, by the constant skidding distance employed in the FRS extension. On the other hand, in the existing network, the separation between roads is not uniform and even closer than in the simulated networks. This heterogeneity in the spatial distribution of roads influence the division, the aggregation, and the shape of forest patches.

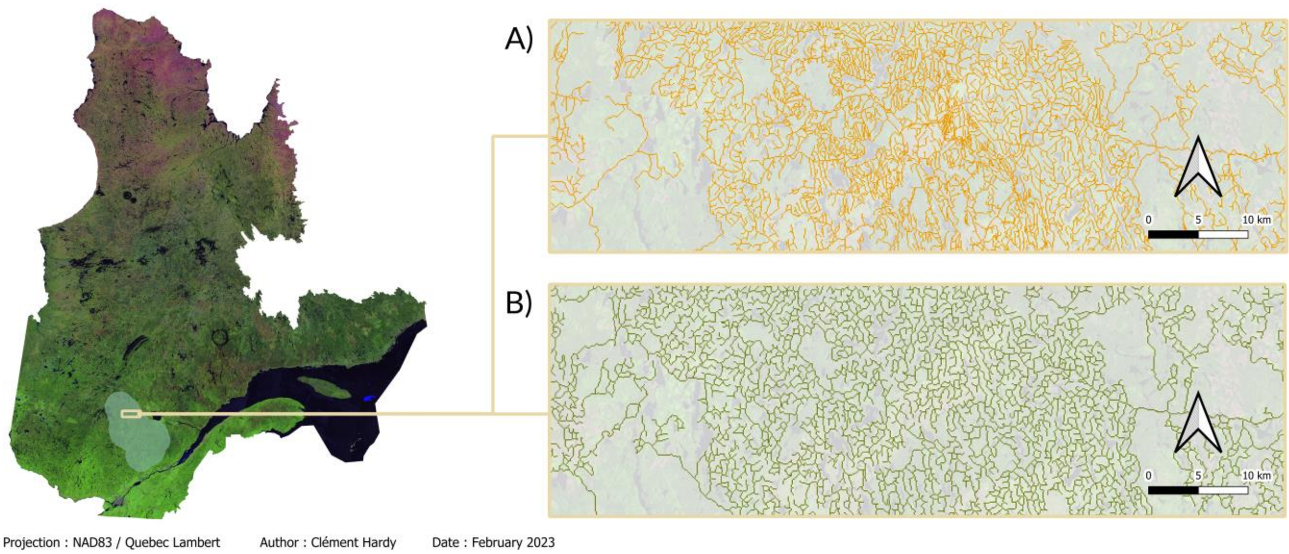


Figure 1-7 : Comparison of the road pixels corresponding to A) existing forest roads and B) forest roads simulated by the FRS extension, at a particular location in the Mauricie landscape used to test the extension. Satellite imagery is from the Landsat project, and accessed through the data portal of the Ministry of Forests, Fauna and Parcs of Quebec (https://geoegl.msp.gouv.qc.ca/ws/mffpecofor.fcgi?request=GetMetadata&layer=lsat_mos2020).

Together, the differences in the number of patches, *Clumpy* and TCA suggest that forest pixels were less aggregated in landscapes with simulated roads. This resulted in a higher number of patches with a generally smaller core area than in existing road landscapes. Moreover, the larger values of PAFRAC in the simulated landscapes indicates that shape of forest patches tended to depart from simple squares toward more convoluted shapes with a longer perimeter or outside edge. This, in turn, is also supported by the decrease in TCA which implies that forest patches have less core area and more edge.

Globally, our results showed notable local correspondence between the location of simulated and existing road pixels. The superposition of pixels within a 500 m radius was especially high for road pixels located inside forest operation areas, whereas this correspondence decreased by about 40% for road pixels located outside forest operation areas (Table 1-2) - representing between 10% and 30% of all road pixels in the landscapes. The poor overlap between existing and simulated road pixels outside forest operation areas is due to reduced constraints and hence greater freedom with which the FRS extension could position forest roads. A possible reason to explain this divergence is that FRS operates at the strategic scale, whereas existing roads are ultimately constructed at the operational scale. The strategic scale of forest management considers long-term and large-scale objectives such as the annual allowable cut, conservation objectives, or aggregation of forest harvesting. In contrast, the operational scale takes into account small scale and short-term factors such as the position of employee accommodations and timber landings, temporal constraints influencing the durability requirements of roads, the precise order in which forest operations will be executed, etc. Had the actual FRS extension considered these more operational factors, there might have been greater concordance between existing and simulated road pixels outside forest operation areas. While some factors could be integrated into future versions of the FRS extension (e.g., temporal constraints associated with each forestry activity), this increased precision would require more specific data that is rarely available. Moreover, some factors (e.g., timber landings) could never be considered in a model such as LANDIS-II because this would require fine-scale data related to the operational scale (e.g., fine topographical elements, technology available, etc.). Such fine-scale data could not be implemented in LANDIS-II without requiring many more parameters and input data, but also much longer simulations times due to the finer spatial and temporal resolution required. In addition, such efforts would always be limited by the discrete nature of space and time in the functioning of LANDIS-II. Hence, studies that focus on the precise location of forest roads outside forestry operation areas (e.g., roads that cross the habitat of an endangered species) should recognize that the FRS extension does not simulate forest roads at an operational level.

In the end, our results show that the FRS extension can be used in modelling studies that are interested in landscape-scale measures of road density and fragmentation, or that explore ecological or economic issues influenced by the spatial distribution of forest roads at a strategic level. However, these results must be interpreted in light of the transformations that we made on the existing road networks (rasterization, elimination of inadequate roads and forest operation areas; see section 1.2.2.3). Still, we believe that these transformations did not alter our results in an important way, as only a small portion of existing roads (around 10%) were deleted in the process. Moreover, the FRS extension employs a uniform skidding distance across the simulated landscape and should be used with caution in contexts where local features of the landscape (i.e., topography, soils, etc.) can influence the skidding distance. In these situations, the FRS extension might over- or under-estimate fragmentation or road density measures, depending on how the parametrization was made. In such cases, we recommend that users explore the sensitivity of their results to important parameters including the skidding distance or those used by the loop algorithm.

Additionally, in hilly terrain, the presence of steep slopes increases the complexity of road construction, which is represented in our model (Stückelberger *et al.*, 2007). Our two regions did not contain steep slopes (> 10%) in large quantity, preventing us from measuring the performance of the FRS extension in hilly terrain. However, as the model discourages road construction in cells with a steep slope or topographic obstacles (cliffs, etc.) by increasing their cost of construction, we expect that it will be able to provide adequate estimates of the position of roads in most hilly terrains. In a broader context, studies that focus on the precise location of forest roads could integrate appropriate cost layers at a smaller resolution than those used here (250 m, 100 m). Thanks to the open-source and collaborative nature of the FRS extension, we expect that future contributions will improve the precision of our model. Finally, the FRS extension also provides other measures and outputs that could not be tested here. In particular, the estimated construction cost of the road network can be used to compare different management scenarios that vary the economic incentives or obstacles to road construction for the forest industry.

1.5 Conclusion

We presented the FRS extension, a new extension for the LANDIS-II forest landscape model that simulates the construction of forest road networks. We showed that the FRS extension has few essential parameters and many options, allowing researchers to simulate the evolution of forest road networks with different levels of details. In its simplest form, the extension only requires a single raster map (elevation) and two cost parameters (basal distance cost and an additional cost due to the slope). More advanced options can integrate other spatial features of the landscape (e.g., streams or soils), roads aging, and timber flux through the road network. The execution time of the FRS extension is very fast and does not increase the simulation time of the LANDIS-II model by more than a few minutes at each iteration of the extension, for a landscape of 5 million active cells.

The FRS extension will enable researchers to include the construction of roads in forest management, furthering our understanding of its impacts on forest ecosystems. Consequently, it will offer the possibility to explore or re-explore research questions relative to forest roads including their impact on landscape fragmentation, animal movement, recreational access, fire management, social acceptability of timber harvesting, and cost at larger spatial and temporal scales. Simulating forest roads is crucial at a time when forests are perceived as providers of important ecosystem services, like carbon storage via afforestation and substitution, which can be affected by the construction and usage of forest roads.

1.6 Acknowledgments

We acknowledge the valuable contribution of Jérôme Garet from Quebec Ministry of Natural Resources and Forests in the parameterization of the module. We also thank members of the LANDIS-II Foundation and Yan Boulanger from the Canadian Forest Service for their help with LANDIS-II, as well as Guillemette Labadie from Université Laval for *sharing* her data of the Côte-Nord region. Finally, we thank Lana Ruddick for her language revision of the article.

1.7 Competing interests

The authors declare there are no competing interests.

1.8 Contributions

Clément Hardy: Conceptualization, Methodology, Software, Formal analysis, Investigation, Data Curation, Writing – Original Draft, Visualization Christian Messier: Conceptualization, Writing – Review & Editing, Supervision, Funding acquisition Osvaldo Valeria: Conceptualization, Software, Writing – Review & Editing Elise Filotas: Conceptualization, Methodology, Ressources, Investigation, Writing – Review & Editing, Supervision, Funding acquisition.

1.9 Funding

Funding was provided by the Natural Sciences and Engineering Research Council of Canada (NSERC) through a Collaborative Research and Development grant awarded to C. Messier (RDCPJ: 498998-16) with the joint support of Resolute Forest Products, and a Discovery grant awarded to E. Filotas (RGPIN: 2018-06156).

1.10 Data availability

The source code, binaries and user guide of the FRS extension are all available at https://github.com/Klemet/LANDIS-II-Forest-Roads-Simulation-extension.

CHAPITRE 2

Land *sparing* and *sharing* patterns in forestry: exploring even-aged and uneven-aged management at the landscape scale

Ce chapitre a été publié en anglais dans la revue scientifique Landscape Ecology en Septembre 2023. Il est reproduit ici avec la permission de Springer Nature. **Le texte ici présent contient cependant quelques modifications mineures recommandées par le jury**. Le text original a été rédigé par les personnes suivantes :

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Mots-clés : uneven-aged management, forest roads, LANDIS-II, forest fragmentation, forest landscape modelling, aggregated harvest

Citation complète : Hardy, C., Messier, C., Boulanger, Y., Cyr, D. et Filotas, É. (2023). Land *sparing* and *sharing* patterns in forestry: exploring even-aged and uneven-aged management at the landscape scale. Landsc Ecol. https://link.springer.com/article/10.1007/s10980-023-01742-7

Résumé : Dans ce chapitre, nous nous sommes intéréssés à identifier les impacts des stratégies d'aménagement forestières équienne et inéquienne sur le long terme et à l'échelle du paysage. Plus

particulièrement, nous nous sommes penché sur l'effet de ces stratégies sur la densité de chemins forestiers, la quantité de vieilles forêts et la fragmentation des vieilles forêts d'un paysage aménagé. Pour ce faire, nous avons simulé un paysage de 800 000 hectares dans la région de la Mauricie, au Québec, sur un horizon de plannification de 150 ans avec le modèle LANDIS-II. Pour simuler les chemins forestiers nécéssaires à l'aménagement forestier dans le paysage, nous avons utilisé l'extension FRS dévellopée dans le chapitre 1. Au total, nous avons comparé 30 scénarios d'aménagement variant la proportion d'aménagement équien ou inéquien utilisée dans le paysage, le niveau d'aggrégation des coupes, et la présence d'un réseau de chemins forestiers initial. Nos simulations montrent que l'aménagement inéquien tend à augmenter la densité de chemins forestiers, la quantité de vieilles forêts, mais aussi la fragmentation des vieilles forêts en comparaison à l'aménagement équien. Les feux de forêts plus présents dans le nord de notre zone simulé y ont réduit les différences entre scénarios. L'aggrégation des coupes et la présence d'un réseau de chemins forestiers initial n'ont pas eu d'effet sur le long terme pour toutes nos mesures. Nous concluons que le choix entre aménagement équien ou inéquien présente ainsi un compromis entre la quantité de chemins forestiers et la fragmentation qu'ils causent, et la quantité de forêts plus vieilles préservées dans le paysage. Nous concluons également que ce compromis est influencé par la présence de perturbations naturelles dans le paysage, mais pas par l'aggrégation des coupes.

2.1 Introduction

Forests provide essential ecosystem services for societies across the globe and harbour most of the Earth's terrestrial biodiversity (FAO et UNEP, 2020). Yet, the increasing demand for land and forest products as well as climate-induced changes to forest structure and functions are threatening forest health (Trumbore *et al.*, 2015). In fact, about 30% of the area covered by forest biomes in the world is managed, under varying intensities, to produce timber and non-timber forest products (FAO, 2020). In the temperate and boreal forests, forestry represents the most important form of anthropogenic disturbance on forests (Curtis *et al.*, 2018). Thus, designing sustainable forest management strategies that can balance the trade-offs between timber demand and conservation objectives is a pressing challenge (Brang *et al.*, 2014 ; D'Amato *et al.*, 2011 ; Lindenmayer *et al.*, 2012).

In North America, even-aged forest management is still widely used, especially in boreal and mixed forests. Indeed, clearcutting accounts for more than 80% of all forest harvesting in Canada, whereas it is around 40% in the USA (Oswalt et Smith, 2014 ; Statistics Canada, 2018). However, clearcutting has been increasingly perceived as having negative environmental impacts. Multiple scientific studies have shown that even-aged management can alter the structure of forest ecosystems and reduce biodiversity when applied across wide forest landscapes (Cyr *et al.*, 2009 ; Martin *et al.*, 2020) or in forests that are not adapted to stand-replacing disturbances (Burton et Canada, 2003 ; Nolet *et al.*, 2018 ; Park *et al.*, 2005). For example, Paillet *et al.* (2010) observed that clearcutting in European forests was associated with a decline in species richness of several taxa, and Cyr *et al.* (2009) showed that even-aged management shifted the age-class distribution of boreal forests towards younger stands, to the detriment of old-growth ones. Moreover, the unaesthetic scenery created by clearcutting has led to widespread public dislike of this method (Ribe, 2005).

Consequently, forest scientists and policy makers have recently shown renewed interest in uneven-aged forest management (Diaci *et al.*, 2011 ; O'Hara, 2002 ; Schütz *et al.*, 2012) where three or more age classes within harvested stands are preserved through selection cutting of individual trees or groups of trees (Hawley et Smith, 1962 ; Matthews, 1991 ; Puettmann *et al.*, 2009). Because uneven-aged management can maintain a constant forest cover and many important attributes associated with older stands, it is considered a relevant alternative to even-aged management whose widespread use tends to create open habitats and younger forest landscapes. This is particularly true in forest regions that are known to mostly experience rather fine scale, partial natural disturbances (e.g., windthrows, insect outbreaks) which are appropriately emulated by uneven-aged management (Kuuluvainen *et al.*, 2021).

Indeed, uneven-aged forest management has been associated with improved protection of biodiversity and sustainability of ecological services (Fedrowitz *et al.*, 2014 ; Pukkala, 2016). For example, some studies have shown that forests managed with uneven-aged compared to even-aged management captured more carbon (Strukelj *et al.*, 2015), had a greater abundance and diversity of birds (Tittler *et al.*, 2001), mammals (Ruel *et al.*, 2013), plants (Götmark *et al.*, 2005), bryophytes (Boudreault *et al.*, 2013 ; Stone *et al.*, 2008), and insects (Graham-Sauvé *et al.*, 2013 ; Joelsson *et al.*, 2017) several years following harvesting, and showed an increased resilience in landscapes associated with high fire frequency (Cyr *et al.*, 2022). Other studies have also shown that uneven-aged forest management conserved several important characteristics of old-growth forests, such as tree species diversity, species abundance, and a broader tree diameter distribution (Adamic *et al.*, 2017 ; Gronewold *et al.*, 2010).

However, evidence of the ecological advantages of uneven-aged management over even-aged management remains equivocal or disputed (Nolet *et al.*, 2018). In addition, most studies to date reporting on the benefits of uneven-aged management have been carried out at the stand scale and over a short period of time (e.g., 1-35 ha and 2-9 years in previously cited studies), while focusing on specific groups of species (1-3 taxa in previously cited studies). It is also currently unclear how disturbances created by uneven-aged management interact with the cross-scale spatiotemporal dynamics of natural disturbances, such as forest fire (Kalabokidis et Wakimoto, 1992 ; Kuuluvainen *et al.*, 2012 ; Nolet *et al.*, 2018). Hence, a comprehensive understanding of the impact of uneven-aged management on forest structure and function at spatial and temporal scales at which forest management is performed is currently lacking (Nolet *et al.*, 2018).

Uneven-aged management may have several shortcomings when it comes to conserving forest structure and diversity, especially at the landscape scale. During a planning horizon, uneven-aged management requires that silvicultural operations extend over larger forest areas than even-aged management to harvest the same volume of wood (Betts *et al.*, 2021a). Consequently, uneven-aged management may require the development of a larger and more permanent forest road network to reach distant stands over shorter rotation periods especially in countries or regions with vast expanses of remote forest like Canada (Alexander et Edminster, 1977 ; Nolet *et al.*, 2018 ; United States Department of Agriculture, 1997). As such, uneven-aged management likely increases landscape fragmentation, with potential negative effects on habitat connectivity for several species and ultimately on biodiversity (Haddad *et al.*, 2015). Forest roads have also been linked to other negative effects such as contributing to the decline in the macroinvertebrate soil fauna (Haskell, 2000), beetles (Koivula, 2005) and salamanders (Marsh et Beckman, 2004); facilitating the spread of invasive plants (Mortensen *et al.*, 2009); reducing habitat quality for some species of birds (Ortega et Capen, 1999); changing movement patterns of large mammals such as the elk (Witmer et deCalesta, 1985); increasing wolf predation on the caribou (James et Stuart-Smith, 2000 ; Vanlandeghem *et al.*, 2021 ; Whittington *et al.*, 2011); and influencing the spatial boundaries of forest fires (Narayanaraj et Wimberly, 2011). In addition, the shorter rotation periods of uneven-aged management would mean more frequent disturbances within stands, possibly reducing the potential benefits of uneven-aged management described above in the long-term.

From these considerations, even-aged management can be seen as a *land-sparing* approach where intensive harvesting is carried out on a smaller portion of the landscape while uneven-aged management can be seen as a *land-sharing* approach (Lindenmayer *et al.*, 2012). As such, even-aged management would "spare" the remaining forests of the landscape by reaching a harvest target over a reduced area; whereas uneven-aged management would "share" the entire forest landscape for different purposes through less damaging harvesting. This debate between the efficacy of *land-sparing* and *land-sharing* strategies for conservation has been long ongoing in conservation sciences, especially in the context of agriculture (Balmford, 2021). Consequently, this raises questions regarding the potential trade-offs between even-aged and uneven-aged approaches: How do these trade-offs evolve over time? How many more forest roads does uneven-aged management require? How do the effects of roads on forest fragmentation and composition differ between even- and uneven-aged management? Can some of these effects be altered by the aggregation of harvested zones in the landscape, as suggested by previous studies (Carlson et Kurz, 2007)? In the present study, we compare the effects of even- and uneven-aged forest management on the amount and fragmentation of old forests over a 150-year planning horizon and across an extensive management unit located in the northern temperate and boreal forests in Quebec, Canada. This management unit is almost entirely composed of forests and lakes with very few rural communities (covering only 0.1% of the unit area). As such, it is an interesting area to study the land *sparing* – land *sharing* question in forestry. Indeed, the large surface of forests in the area allows the implementation of both strategies on a large scale. In addition, its fragmentation will be sensitive to the presence of forests roads, which are the majority of roads found in the area (more than 90%).

We focus on old forests since they present characteristics that are essential for multiple species (MacKinnon, 1998 ; Terry *et al.*, 2000 ; Thompson, 1994) and provide key ecosystem services (Clark, 2011 ; Frey *et al.*, 2016 ; Luyssaert *et al.*, 2008), but have severely declined in managed landscapes (Cyr *et al.*, 2009 ; Martin *et al.*, 2020 ; Shorohova *et al.*, 2011). Therefore, conservation of old forests has become an important management objective in both governmental regulations and international certification programs (Knorn *et al.*, 2013 ; Merschel *et al.*, 2019 ; Strittholt *et al.*, 2006). Here, we used the term "old forests" to describe forests with old trees, to distinguish it from the term "old-growth forests" which can refer to a set of structural characteristics, processes, functions or legacies (Wirth *et al.*, 2009).

We use LANDIS-II, a forest landscape model (Scheller *et al.*, 2007), in combination with the Forest Roads Simulation module (FRS), a new extension that simulates the construction of forest roads needed to reach harvesting sites (Hardy *et al.*, 2023b). We simulated 30 different management scenarios where three factors modulating the impact of uneven-aged management on the fragmentation of old forests were varied: (i) the proportion of aboveground tree biomass harvested with uneven- vs. even-aged silviculture, (ii) the presence/absence of an initial forest road network, and (iii) the level of spatial aggregation of logging sites. Crucially, every management scenario had to harvest the same biomass target at every timestep so as to compare their effects properly. Our study area was composed of mixed and boreal forest, with frequent forest fires in its northern half, but not in its southern part. We compared these scenarios based on the temporal dynamics of the following response variables: the density of forest roads, their cost of construction and repair, and the amount and fragmentation of old forests. Our goal was not to attempt to find an optimal location for even- or uneven-aged cuts (i.e. Tittler *et al.* 2015), but to explore two broad management strategies and their interactions with forest roads and natural disturbances. Furthermore, we did not include climate change as a varying factor, in order to better isolate and interpret differences resulting from the different management strategies and reduce the computational load of the simulations.

We hypothesized that a *land-sharing* strategy, represented by a higher proportion of the landscape harvested under uneven-aged management, protects a larger amount of old forests compared to a *land-sparing* strategy characterized by more even-aged management. On the other hand, we also hypothesized that uneven-aged management increases road density and usage of roads, along with the fragmentation of the landscape. As such, simulations with a higher level of uneven-aged management would lead to higher road density, greater costs in road construction and maintenance, and higher fragmentation of old forests. Additionally, we hypothesized that aggregating harvesting areas requires less forest roads and thereby reduces their negative impacts on cost and fragmentation. Finally, we hypothesized that the presence of natural disturbances interacts with harvest prescriptions by lightening or exaggerating their effects on the composition of the landscape.

2.2 Methodology

2.2.1 Simulated area and study area

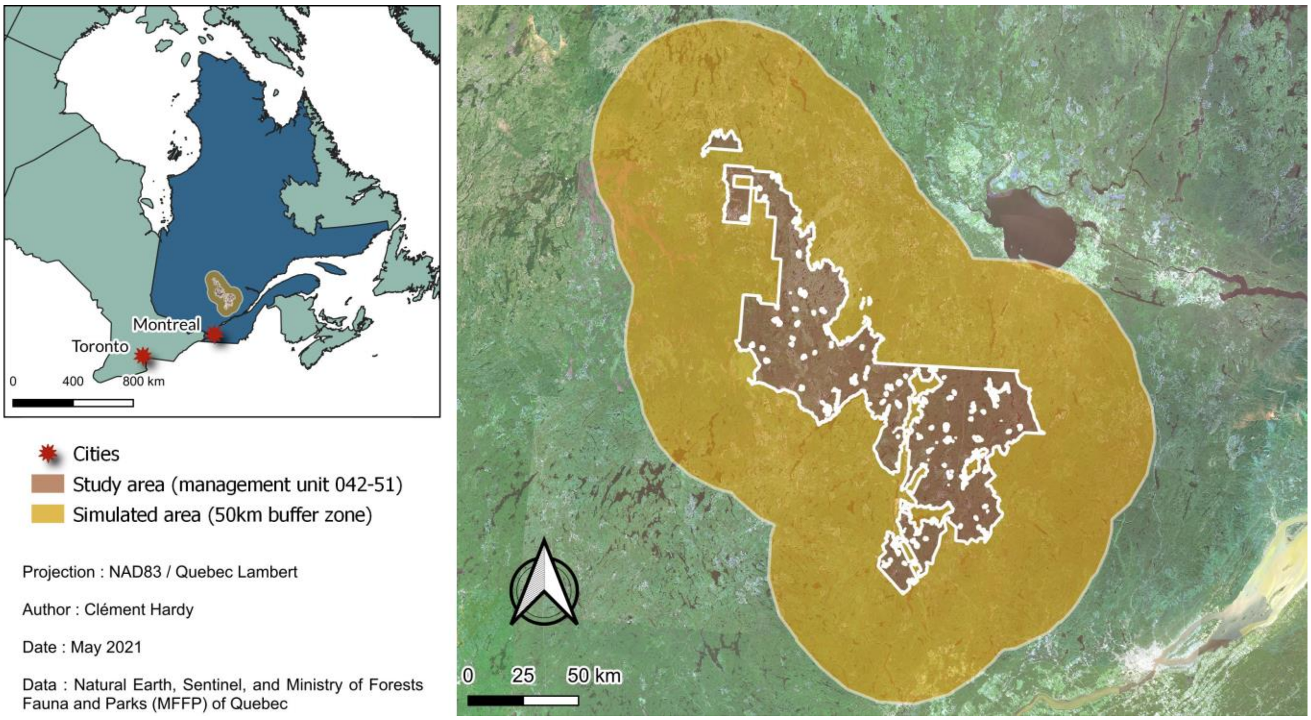

Figure 2-1 : Map of study area (brick colour in the centre) and simulated area (gold) in Mauricie (Quebec, Canada). The northern part is located in the boreal coniferous forest whereas the southern part is located in the temperate mixed wood forest.

Our study area corresponds to a forest management unit in Mauricie, Quebec (Figure 2-1) where both even-aged and uneven-aged management are currently being used (Messier *et al.*, 2009). The total area covers around 800 000 hectares and is typical of the temperate mixed wood and southern boreal coniferous forest. It extends mostly from the balsam fir and yellow birch region in the south to the balsam fir and white birch region in the north (Robitaille et Saucier, 1998). The dominant species in the southern part of the landscape are balsam fir (*Abies balsamea*), yellow birch (*Betula alleghaniensis*) and trembling aspen (*Populus tremuloides*). Further north, the main species are balsam fir, white birch (*Betula papyrifera*), trembling aspen, black spruce (*Picea mariana*) and jack pine (*Pinus banksiana*). While fire is an important natural disturbance in the region with a return interval of around 300 years in the north and 5300 years in the south (Boulanger *et al.*, 2014 ; Couillard *et al.*, 2022), recurrent spruce budworm outbreaks are also significant in balsam fir stands particularly in the south (Bergeron et Fenton, 2012). Because of the prevalence of severe forest fires, even-aged forests occur naturally in this landscape (MFFP, 2018b). The study area is characterized by few rural communities (covering only 0.1% of the unit area) connected by a sparse network of paved roads. Large-scale logging began in the south of the area during the late 1920s, progressively spreading to the north during the following decades (Messier *et al.*, 2009). Forest management until today consisted largely of clearcutting followed by planting or natural regeneration, mixed with different forms of partial cutting. Being almost entirely composed of public forests, forest management in the area is currently regulated by the Ministère des Ressources naturelles et des Forêts du Québec (MRNF, formerly MFFP). As of this 2018, most of the forest surface in the area was classified as being less than 70 years old, with only 11% being 90 years old or more; whereas 12% of the surface is considered as having an uneven-aged structure (3 age classes or more) (MFFP, 2018b).

2.2.2 The LANDIS-II forest landscape model

LANDIS-II is a spatially-explicit forest landscape model (FLM) that simulates forest dynamics through both stand-level processes (e.g., succession through intra- and interspecific interactions) and landscape-scale processes, such as seed dispersal (Scheller *et al.*, 2007) as well as natural (e.g., fire, wind, and defoliating insects) and human disturbances (e.g., harvesting and land-use change) based on the life-history traits of tree species. Trees are modelled as cohorts, i.e., as a group of trees of a given age class and species.

In LANDIS-II, the simulated landscape is composed of square cells that are either forested or non-forested (e.g., water, urban area, etc.). Each forested cell is categorized into an ecoregion, representing the effect of climate and/or soil on tree growth, and is assigned to a management area for harvesting purposes. The model is composed of multiple extensions, each simulating one of the main processes driving forest dynamics. During one iteration, each extension operates sequentially. Parameters for the extensions used in the present study are summarized in Appendix B.1.

Our simulations were executed with LANDIS-II v. 7.0 on a landscape of around 4 million forested cells encompassing our study area surrounded by a 50 km buffer zone to avoid border effects due to roads (Figure 2-1). However, all of our results only concern the study area in the center. We chose a cell size of 1 hectare, which is typical of LANDIS-II studies as it represents a good compromise between computational load and simulation of fine-scale processes (Shinneman *et al.*, 2010 ; Sturtevant *et al.*, 2012).

2.2.3 Parametrization of LANDIS-II

2.2.3.1 General parametrization

The parameter values for our study were derived from the protocol described in Boulanger *et al.* (2017) and subsequently used in several studies using LANDIS-II to simulate forest landscapes in Quebec and elsewhere in Canada (Boulanger *et al.*, 2017, 2019 ; Tremblay *et al.*, 2018). Simulation duration was set to 150 years (from the year 2000 to 2150) to reflect the strategic planning horizon of forest management in Quebec (Bureau du forestier en chef du Quebec 2013). All modules operated on a time step of 10 years, a compromise between simulating processes occurring over short and long terms (Boulanger *et al.*, 2017 ; Shinneman *et al.*, 2010 ; Sturtevant *et al.*, 2012).

The initial composition and age structure of the forest community within each cell were determined using data from the Canadian National Forest Inventory (Beaudoin *et al.*, 2014) and Quebec's permanent and temporary sample plots (Alain *et al.*, 2016b, 2016a). Each cell was assigned a sample plot using a nearest neighbour analysis based on age and species-specific biomass (Boulanger *et al.*, 2017 ; Tremblay *et al.*, 2018). In total, 17 different tree species were modelled (see Appendix B.2). In addition, the ecoregions of our landscape were defined by assigning cells to homogeneous units of soil (Mansuy *et al.*, 2014) and climate (Boulanger *et al.*, 2017). Each of the nine management areas in our landscape corresponded to those currently used by the MRNF. Finally, the forest stands (which are groups of cells within management areas) were defined spatially using the stands identified during Quebec's provincial inventory (MFFP, 2018a).

2.2.3.2 Biomass Succession

Succession was modelled using the "Biomass succession" extension (v. 5.2) of LANDIS-II, which computes the biomass and biomass increment of tree cohorts. This module relies on a collection of parameters that determine the competitive ability of each species and its establishment and growth potential under specific environmental conditions. The latter parameters were derived using the model PICUS (Lexer et Hönninger, 2001), an individual-based spatially-explicit model of stand dynamics, as in Boulanger *et al.* (2017) and Tremblay *et al.* (2018) (see Appendix B.3 for further information).

Life-history traits and shade tolerance parameters for the 17 tree species modelled were estimated from the literature (Appendix B.1). The values of the other parameters (e.g., growth curves and shade impact on productivity) were determined from calibration runs with the goal of minimizing differences with initial estimates of biomass made by LANDIS-II and biomass estimates from the National Forest Inventory for the simulated area (Appendix B.3).

2.2.3.3 Natural disturbances (Fire)

Two main natural disturbances are present in our landscape: forest fire, and epidemics of spruce budworm (*Choristoneura fumiferana*). We simulated forest fire, which consists of the dominant natural disturbance in our study area (see section 2.2.1) using the Base Fire extension of LANDIS-II, but for simplicity we decided not to simulate spruce budworm outbreaks. This choice facilitates the interpretation of the results since the simulated forest dynamics emerges from fewer interactions between forest management and natural disturbances. The Base Fire (v. 4.0) extension of LANDIS-II follows a simple dynamic (He et Mladenoff, 1999) where a fire is initiated within a cell according to a probability that increases with the time since the last fire that happened in the cell. The fire then spreads successively to neighbouring cells based on probabilities modulated by a randomly determined wind direction. Finally, local fire damage depends on fire severity and species composition. The minimal, average, and maximal sizes of fire, and its return interval can be specific to “fire regions” determined by the user.

Following the method employed in Boulanger *et al.* (2017), two fire regions were defined in our landscape from the delimitation of the homogeneous fire zones of Boulanger *et al.* (2014). The parameters of LANDIS-II for each fire region were estimated in order to reproduce the historical maximal and minimal fire sizes in each region as well as the percentage of area burned each year on the simulated landscape during the initial 30 years of simulation (see Appendix B.4).

2.2.3.4 Harvesting

Timber harvesting in LANDIS-II is modelled as a set of prescriptions (e.g., clearcutting, selective cutting, etc.) that are applied within user-defined management areas. Each management area can thus have a different set of prescriptions that are applied on a certain percentage of its surface. For each management area, prescriptions are realized one by one in a random order and are applied to selected stands based on a set of conditions and/or ranking algorithm (random, economic value, age structure, etc.). Cohort harvesting using a given prescription stops once the surface to be harvested in the management area has been reached. No particular zoning (i.e. TRIAD zoning; see Messier, 2022) was applied, meaning that every prescription could be used inside each management areas, but that no harvesting took place outside of these (i.e. in protected areas).

The current LANDIS-II harvest module can only define prescriptions based on a targeted percentage of the total surface of the management area to be harvested. However, our research question required that we harvest the same amount of biomass, not surface, with different prescriptions in order to properly compare the impacts of even-aged and uneven-aged management on the same landscape. Hence, we used the Magic Harvest extension for LANDIS-II to better control the biomass that was harvested by the Biomass Harvest extension via a custom Python script (Hardy, 2022) (see Appendix B.5).

The biomass target in each of the nine management areas was based on the annual allowable cuts (AAC) determined by Quebec Chief Forester's Office and the MRNF for the 2018-2023 period (MFFP, 2018b). The AACs are expressed in volume, to which we translated into biomass using wood density measures from (Gonzalez, 1990). The targets were also adjusted to the temporal resolution employed in our simulations (10 years instead of 5 years for the AAC; see Appendix B.5). The computation of the AAC in Quebec is a multi-constraint optimization exercise realized for each management unit. This optimization includes a constraint on the amount of older forests that must be preserved in the landscape, which was integrated in the harvest targets that we employed. Harvesting prescriptions and the criterion used to prioritize the stands to be harvested were defined to simulate typical harvesting operations conducted in Quebec today, which may differ from prescriptions in other countries.

The main prescriptions used in our study area include clearcutting, shelterwood cutting, and continuous-cover forestry (sometimes referred to as irregular shelterwood) (Table 2-1). The first two prescriptions correspond to even-aged management, while the third corresponds to uneven-aged management. All prescriptions were applied only to stands 30 years or older, meaning stands that had at least one age cohort of more than 30 years. This age constraint implied rotations of at least 30 years. However, these stands could contain age cohorts younger than 30 years that could be harvested by the clearcutting and shelterwood prescriptions. Clearcutting harvested all age cohorts older than 10 years in a cell (to mimic clearcutting with protection of regeneration as is currently done in Quebec). Shelterwood cutting harvested 90% of the biomass of all age cohorts older than 10 years in the cell in a first cut, and then harvested every remaining cohort older than 20 years in a second cut 20 years later. Finally, the irregular shelterwood was initially modelled to harvest 30% of the biomass of all age cohorts 30 years and older in the cell every 30 years. We fixed the proportions of clearcutting and shelterwood cutting (our two even-aged management prescriptions) to the proportions currently used in our study area. Hence, shelterwood cutting harvested 10% of the biomass target calculated for even-aged management, while clearcutting removed the remaining 90% (Table 2-1). Finally, we allowed the 30-year return cycle of the irregular shelterwood prescription to be overridden in the case where the biomass target couldn't be reached by respecting this delay. This could happen in management areas with a large biomass target, in scenarios where a high proportion of uneven-aged management was used.

Table 2-1 : Summary of the harvest prescriptions used in the simulated landscape.

Prescription	**% biomass harvested**	**First harvest**	**Second harvest**
Even-aged			
Clearcutting	90%	All age cohorts older than 10 years	None
Shelterwood	10%	90% of the biomass of all age cohorts older than 10 years	20 years later: all age cohorts older than 10 years
Uneven-aged			
Irregular Shelterwood	100%	30% of the biomass of all age cohorts 30 years and older	Same as the first, and repeated every 30 years

We varied the size of harvested patches among the different aggregation scenarios that we tested (see section 2.2.5 below). The maximal area that could be harvested in a single cut was defined according to historical records of cut size in our study area (MFFP, 2018b). Harvesting was simulated by “propagating” the harvested area from forest stand to forest stand until the maximum cut size was reached or until no adjacent stands satisfying the age constraint (30 years or older) were available.

2.2.4 The Forest Roads Simulation Module

To model the dynamics of forest roads, we used the LANDIS-II “Forest Roads Simulation” (FRS) extension (Hardy *et al.*, 2023b). The FRS extension generates or updates the forest roads needed to carry the harvested wood in the landscape in a realistic yet computationally efficient way. To do this, the FRS extension generates a path of road segments that connects cells from a harvested area to an exit point (either the location of a sawmill or a point on the existing main road network) with minimum construction costs based on landscape topography and existing roads.

The running of the FRS extension is described in Figure 2-2. The extension uses a raster layer, the "road landscape", to track the location of forest roads. Roads belong to a size category (tertiary road, secondary road, primary road, etc.) that accommodates an increasing flux of trucks needed to carry the harvested timber (Karlsson *et al.*, 2006). The FRS extension is activated during the initiation phase of a LANDIS-II simulation and at each iteration following the harvesting procedure.

Figure 2-2 : Diagrams showing the evolution of the forest road network simulated by the FRS module. Road pixels are coloured according to their size category. (1) Input data is read by the module; (2) the roads that are not connected to an exit point (brown pixels; e.g., paved roads, sawmills, etc.) are detected and linked to the rest of the network in the initialization phase; (3 and 4) at each time step, the FRS module generates forest roads connecting the harvested areas to the rest of the network using the Dijkstra algorithm. Roads can form loops in the network, as shown in (3), where two roads connect the harvested area on the right. Additionally, the size category of a road may change with an increase in wood flux where two tertiary roads merge into a secondary road (4). Road aging is also simulated where two segments of secondary roads disappear from the network (4).

During the initiation phase, the FRS extension computes the cost of constructing a potential forest road within each cell. This cost map considers the cost associated with several landscape features, including the "coarse" elevation (mean slope of the cell), the "fine" elevation (presence of breaks or cliffs requiring a detour), water bodies (such as lakes and rivers that would require a bridge), streams (requiring a culvert), and soils (Figure 2-3A, B).

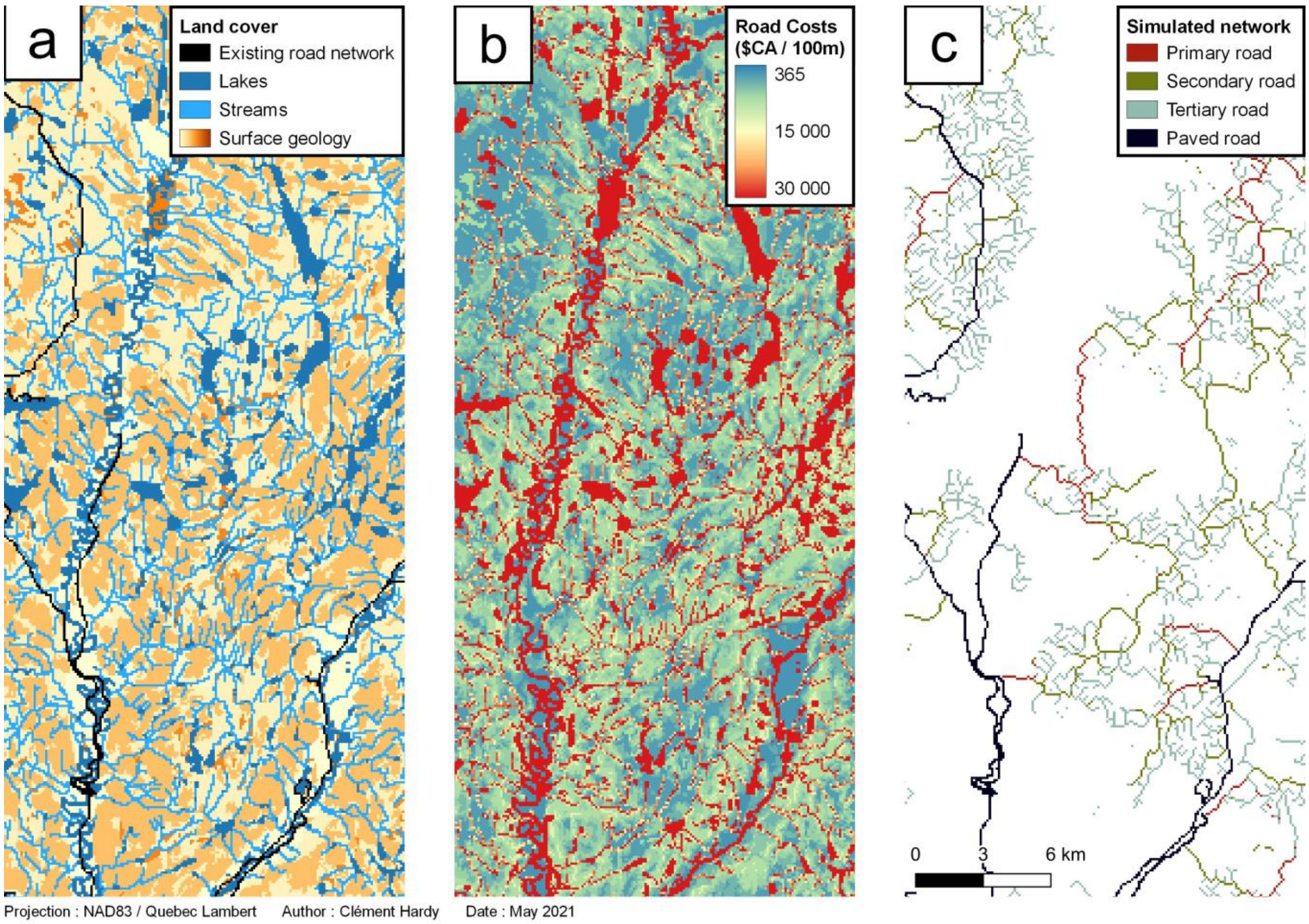

Figure 2-3 : Three maps showing the same section of simulated landscape in the centre of our study area: a) Landscape features; b) Construction costs – computed by the FRS module based on topography, soils, water bodies, streams and existing roads; and c) Simulated network of roads after 150 years of simulations. In c), isolated road segments correspond to remains from old deteriorating roads.

Following the initialization phase, and at each time step where the FRS extension is activated, an attempt is made to create a road toward every cell harvested since the last module activation (Figure 2-2). There are no interactions between the harvest and the FRS extensions. The harvest extension determines the position of cells to harvest, whereas the FRS extension simply creates roads towards these cells. Timber transportation through the network is then simulated, and the size category of roads is updated to accommodate an increasing traffic of logging trucks. Road aging was also considered by removing forest roads that reached their persistence limit (a road type parameter), if they were not repaired (see Appendix B.6). Finally, when a cell is harvested under uneven-aged management, the algorithm considers the necessity of future returns to the cell by selecting the least expensive option between creating a low-cost, nondurable road that may need to be reconstructed or repaired when returning to the cell and creating a higher grade, more durable road that will persist for future access.

In the present study, the FRS module was parameterized using several MRNF datasets reporting on the construction costs of forest roads (e.g., mean costs of construction of road segments by slope, soil type and road category), along with key characteristics of forest roads (e.g., real mean skidding distance, persistence by road type) (see Appendix B.6). Our exit points for the wood were simply the cells containing the main paved roads of the landscape. The transport of wood was oriented toward these cells, and no new main paved roads were created during the simulation. Calibration of the FRS module over our study area was performed in Hardy *et al.* (2023).

2.2.5 Scenarios

To compare the long-term and landscape-scale impacts of even- and uneven-aged management strategies, we defined harvesting scenarios varying three different factors (

Table 2-2). First, we considered the presence or absence of an initial road network in the simulated area. When present, the initial road network corresponded to the existing roads reported in the governmental database “AQReseau+” containing all terrestrial transport routes in Quebec, including forest roads (Gouvernement du Québec, 2015). When absent, the initial network consisted of the main existing paved roads only. To improve the realism of the simulated roads, we calibrated the harvest module such that in the absence of an initial road network, harvested areas and new forest roads would progressively spread in the landscape during the first five iterations (50 years) of the simulation (see Appendix B.5).

Table 2-2 : Summary of the variations of the three factors used to create scenarios. Each of the 30 scenarios corresponded to a unique combination of the three factors.

Factor	**% of biomass target harvested with uneven-aged management**	**Aggregation of harvest cuts**	**Presence of an initial road network**
Levels of factor	0%	Low Maximum size of cuts is half of the historical maximum for each cut type	Yes All roads from the AQReseau + database are initially present
	25%		
	50%	Historical maximum	
	75%	High Maximum size of the cuts is twice the historical maximum for each cut type	No Only the main, paved roads are initially present

Second, we compared the use of uneven-aged (i.e., irregular shelterwood) to even-aged management (clearcutting and shelterwood) by progressively varying the fraction of the total biomass target to be harvested under uneven-aged management (0%, 25%, 50%, 75% and 100%). The remaining fraction of the biomass target was harvested using even-aged management.

Finally, we considered the level of aggregation of harvest cuts given its potential influence on the extent of the resulting road network and on landscape fragmentation (Tittler *et al.*, 2015). The aggregation levels were defined in relation to the maximum size for all the areas harvested in our study area since the year 2000 (209 ha, 359 ha, 117 ha for clearcuts, shelterwood cuts, and uneven-aged cuts, respectively; MFFP, 2018b). As such, we set the maximum size of harvest events in LANDIS-II to the historical average for the intermediate aggregation level. Alternatively, we set the maximum size of harvest events to half and twice that of the intermediate aggregation level for the low and high levels of aggregation, respectively.

We developed one management scenario for each combination of levels for the three factors investigated, resulting in a total of 30 distinct scenarios. We simulated five replicates for each scenario to account for the stochasticity of LANDIS-II using the infrastructure of Compute Canada.

2.2.6 Data analysis

Three classes of metrics were calculated at each time step during our simulations: succession and disturbance dynamics, forest road network and landscape fragmentation. We compared the management scenarios through their effect on the size of each metric during the simulation period (0 – 150 years) (White *et al.*, 2014). Replicates for each scenario captured within-scenario variability due to stochastic processes such as forest fire and succession.

2.2.6.1 Succession and disturbance dynamics

As a validation exercise, we evaluated the impact of the different management scenarios on forest succession and disturbance dynamics. Succession was measured by the aboveground biomass annual net primary productivity (*P*; measured in $\mathrm{g} \cdot \mathrm{m}^{-2-\mathrm{year}}$), corresponding to the total change in aboveground biomass averaged across all forested cells, and by the average age of all age cohorts in the landscape (*A*, in *yr*). In addition, we assessed the landscape-scale impact of the simulated disturbances by measuring the total surface harvested (*Sh*, in hectares), the total biomass harvested (*Bh*, in Mg) and the total surface burned (*Sb*, in hectares) in the study area. These measurements are presented in the appendices of the article (Appendix B.8), as they are not of direct relevance to our research question.

2.2.6.2 Forest road network

We determined the size and monetary cost of the forest road network in our simulated landscape. Network size was measured by road density (*Dr*), which is the proportion of cells that contained a road. The cost (*Cr*), in 2020 Canadian dollars (CAD), for road construction and repair (i.e., size category upgrade and restoration of old roads) was estimated based on the cost parameters cited in section 2.2.4 (Appendix B.7).

2.2.6.3 Landscape fragmentation

Finally, we estimated the landscape-scale impact of forest management on the amount and fragmentation of old forests. We defined old forests as forest stands containing at least one age cohort older than 91 years based on the definition employed by the MRNF (MFFP, 2016). The amount of old forest in the landscape was measured as a percentage of the total forested area (*So*). Fragmentation was measured using the clumpiness index (*Clumpy*) which estimates the aggregation of pixels of a given type (e.g., pixels containing old forests) (equation in Appendix B.12; see McGarigal *et al.* 2012). More precisely, the index measures the deviation in the proportion of adjacencies between pixels of this type in the simulated landscape from the proportion that is expected under a random distribution of the pixels of that type. It varies from -1 (minimum aggregation possible) to 1 (maximum aggregation possible). *Clumpy* also presents a low correlation with habitat amount, making it ideal for measuring fragmentation *per se* (Wang *et al.*, 2014). We used the R package landscapemetric (Hesselbarth *et al.*, 2019) to compute the index.

2.3 Results

We present the temporal dynamics of each metric investigated over the planning horizon (150 years) in our study area. Results are presented for simulations that varied one factor (initial road network, ratio of use of uneven-aged management, or aggregation of cuts) while keeping the other factors at an intermediate value (i.e., no initial road network, 50% of the biomass harvested using uneven-aged management, and intermediate aggregation level). Additionally, results concerning the fragmentation of the landscape are illustrated for the northern and southern regions of the landscape separately (results for the entire landscape are available in Appendix B.11). In so doing, as fire regimes differ in both regions, we are better able to distinguish the potential contribution of fire to landscape fragmentation.

2.3.1 Road density and cost

We observe a much higher road density (Figure 2-4a) and cost of construction and repairs (Figure 2-4b) throughout the simulations with an increased use of uneven-aged management across the landscape. Indeed, the scenarios where more than 50% of the biomass target was harvested with uneven-aged management (green to yellow curves) presented more than twice the density of forest roads in the landscape and up to three times the cost of road construction and repairs than scenarios where only even-aged management was used (purple curve). As the proportion of uneven-aged management increases above 50%, differences between management scenarios become marginal for both the road density and road costs (Figure 2-4a, b). This suggests a saturating effect in the increased use of uneven-aged management for these two variables. The level of aggregation of the cuts, on the other hand, only had a minor effect on the forest road network where higher aggregation slightly reduced road density (Figure 2-4c) and had almost no effect on road construction and repair costs (Figure 2-4d). However, the effect of aggregation on road density was more visible for scenarios with only even-aged management (not shown in Figure 2-4; see Appendix B.10). Lastly, the presence or absence of an initial road network only had noticeable effects on the road density and the cost of roads in the first 20 years of simulations, after which no significant differences were observed (Figure 2-4g, h).

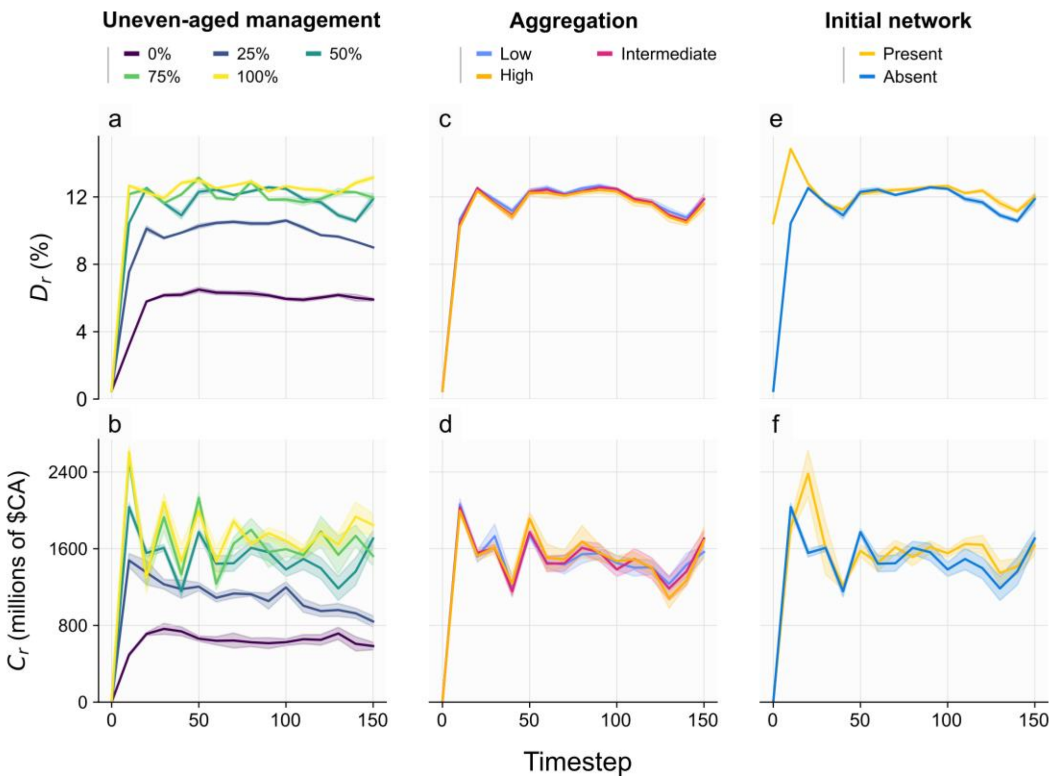

Figure 2-4 : Time dynamics of the road density (Dr; top panel) and the costs of road construction and repair (Cr; bottom panel) for each factor: percentage of the biomass target harvested using uneven-aged management (a, b), aggregation level of harvested areas (c, d), and presence of an initial forest road network (e, f). Each factor (management, aggregation, initial network) is varied while keeping the other factors at their intermediate value. Curves correspond to the average of 5 simulation runs, and shaded areas correspond to the standard deviation across runs.

2.3.2 Amount of old forests and fragmentation level

Following the first 70 years of the simulations, the quantity of old forest increased with an increase in uneven-aged management (Figure 2-5a, c). In contrast, *Clumpy* was lower in scenarios with more uneven-aged management, indicating that the fragmentation *per se* of the landscape increased (Figure 2-5b, d). We observed no strong effect of the aggregation level of cuts on the amount of old forests (Figure 2-5e, g). Moreover, an increase in aggregation level was associated with slightly higher values of *Clumpy*, indicating lower fragmentation in the southern region (Figure 2-5g), but the effect of increased aggregation was almost non-existent in the northern region (Figure 2-5f). Note that the northern and southern regions of our study area were both subjected to a similar intensity of management, as the areas harvested in each of these forests in our different scenarios were almost identical (Appendix B.8).

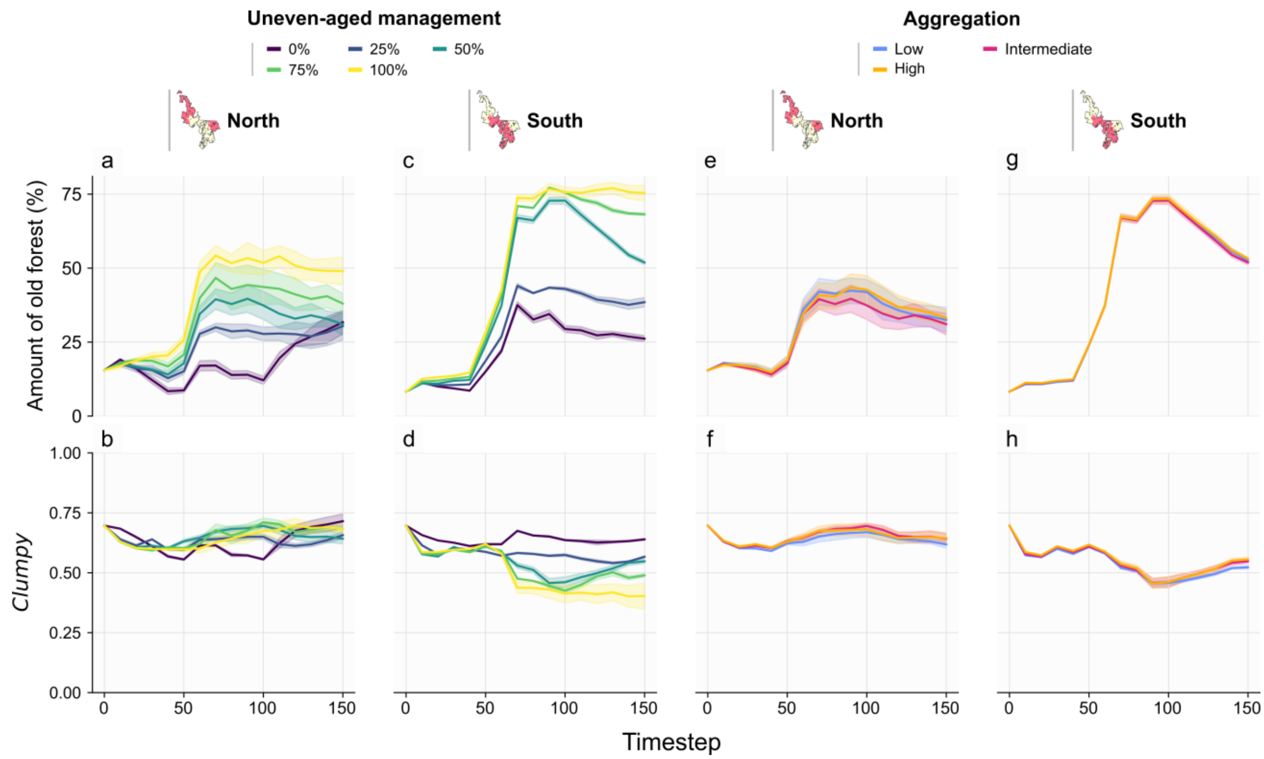

Figure 2-5 : Effects of forest management and fire on the fragmentation of old forests measured by the amount of old forests (top panel, a, c, e, g) and the clumpiness index, *Clumpy* (bottom panel, b, d, f, h). The two columns on the left vary the proportion of uneven-aged management used for the northern region (1st column) and the southern region (2nd column) of our study area. The two columns on the right vary the aggregation level of cuts for the northern (3rd column) and southern (4th column) regions. Each factor (management, aggregation) is varied while keeping the other factors at their intermediate value. Curves correspond to the average of 5 simulation runs, and shaded areas correspond to the standard deviation across runs.

Irrespective of the proportion of uneven-aged management and aggregation level, the amount of old forest increased rapidly after a 50-year time step period of stability. Then, it slightly decreases with time during the second half of the simulations, except for the 0% uneven-aged level in the north which showed a marked increase in the quantity of old forests, and the 50% uneven-aged level in the south which showed a steep decline (Figure 2-5a, c, e, g). *Clumpy*, on the other hand, showed a weaker contrast between scenarios. In the north, *Clumpy* decreased at the beginning of the simulation but increased afterwards with small differences between management scenarios (Figure 2-5b, f). In the south, from 50 years of simulation and onwards, *Clumpy* exhibits a marked decrease for scenarios with more uneven-aged management, indicating an increase in fragmentation. However, no such difference in the value of *Clumpy* is seen for aggregation scenarios (Figure 2-5f).

2.4 Discussion

The goal of this study was to compare the long-term and large-scale effects of even- and uneven-aged forest management on the amount and fragmentation of old forests. We wanted to determine whether forest roads, necessary to access harvested areas, played an important role in differentiating the effects of these management approaches on forest landscapes. Our results provide important insights into the impact of different types of forest management, and their interaction with existing natural disturbances.

2.4.1 The forest road network

Results from our simulations clearly indicate that uneven-aged management as defined in our study (irregular shelterwood with a rotation of 30 years) increases the density of forest roads in the landscape (Figure 2-4a). This outcome was expected since uneven-aged management scenarios extend over a larger surface to harvest the same amount of wood biomass as even-aged management scenarios (Nolet *et al.*, 2018) (see Appendix B.8). Consequently, more roads are needed to access vaster harvested areas. This tendency would have been even stronger had the FRS module been required to build forest roads lasting up to the next cycle of uneven-aged management cuts.

In addition, smaller differences in road density and costs between scenarios with increasing uneven-aged management suggest a saturation effect where a larger ratio of uneven-aged management does not increase the density of roads anymore. We believe that this saturation effect occurs when the additional areas to be harvested under uneven-aged management become accessible through existing roads. In our study, this saturation threshold corresponds to a proportion of about 50% in the use of uneven-aged forest management. This threshold value probably varies according to the biomass target for the landscape, and the amount of available harvestable wood (not simulated).

The increased costs of road construction and repair present a possible limitation for the use of uneven-aged management. In Quebec, where our study area is located, forest roads currently represent 10 to 18% of the operational costs for the forest industries and the government (Groupe DDM et MFFP du Québec, 2020). If these costs were to increase (i.e., by promoting the use of uneven-aged management), it is difficult to determine the economic consequences for the forest industry. The denser forest road network that results from using uneven-aged management increases the access to remote forest areas, which could be desirable for some stakeholders (e.g., hunters, trappers, residents, tourists, etc.) but not necessarily so for others (First Nations, conservation organizations, etc.) (Adam *et al.*, 2012). The increased human presence might lead to negative impacts on the ecosystems surrounding the roads (e.g., increased fire risk, spread of invasive plants, disturbance of wildlife, etc.) (Adam *et al.*, 2012 ; Hunt *et al.*, 2009 ; Kneeshaw et Gauthier, 2006). These potential landscape impacts of an increased road density reveal an important aspect of the *land-sharing* approach in forestry – represented by a prevalent use of uneven-aged methods – and should be taken into account when comparing the *sparing* and *sharing* approaches. However, increasing road density in the context of Reduced Impact Logging (RIL) in the Amazon was shown to increase species richness (Carvalho Jr *et al.*, 2021). Hence, social and ecological context is essential to understanding the impact of forest roads.

The aggregation of harvested areas decreased the density of roads in the landscape far less than what was expected and did not reduce the cost of road construction and repair (Figure 2-4c, d). In fact, the effect was undetectable at 50% uneven-aged management and remained minor even when harvesting was entirely done through even-aged management (see Appendix B.10). The limited effect of aggregation, especially under higher levels of uneven-aged management, can be explained by constraints on the available surface to be harvested. Indeed, when the surface to harvest is large compared to stand availability, cut zones become concentrated in the landscape regardless of the imposed aggregation level. As such, the density of roads in the landscape remains similar even under increased aggregation. Furthermore, aggregating the harvested areas concentrates the quantity of transported timber onto a smaller number of roads, which in turn requires that these roads be updated to a higher size category. The costs to update a few larger roads can be equivalent to those required to construct a higher number of smaller and less expensive roads, which could explain why aggregation does not reduce road costs. Our results might have been different if we had considered other factors such as the more frequent road repairs required with increased traffic of logging trucks, for example.

These results suggest that it would be difficult to compensate for the increase in road density associated with uneven-aged management by simply increasing the aggregation of cuts. In addition, aggregation may not always be operationally possible. Indeed, extending the area of harvest over surrounding stands may be constrained by the lack of trees of the appropriate age or composition or the absence of forest. Local legislation may also limit the size of cuts, which is the case in Quebec since 2018 (Gouvernement du Québec, 2018). As such, it seems unlikely that a high enough level of aggregation could be reached to offset the effects of a switch to more uneven-aged management. Still, aggregation might compensate for a smaller increase in uneven-aged management use if coupled with a strategic spatial distribution of cuts (Tittler *et al.*, 2015).

Our results also indicate that the presence of an initial forest road network did not have a long-term influence on road density and cost (Figure 2-4e, f). Differences in these measures were eliminated over the first 50 years of simulations (a period which also corresponds to the life expectancy of the largest roads in our study). Without the process of road aging and deterioration, we expect that the initial road network would have had a more persistent impact on the landscape. Nevertheless, including forest road aging in our FRS module is a more realistic scenario reproducing the inevitable decay of roads due to erosion, forest regrowth or wear, after which roads are ultimately either repaired or abandoned (Gucinski, 2001).

Additionally, the timing at which the scenarios with and without an initial road network converge (around 30 years) is determined by our calibration of the harvest module: new forest roads progressively spread in the landscape during the first five iterations of the simulation. Had we chosen a shorter or longer period for the expansion of new roads, the convergence of both networks would have occurred earlier or later during the planning horizon.

2.4.2 Amount of old forests

Our study shows that uneven-aged management, in the form of irregular shelterwood, can increase the quantity of older tree cohorts across the landscape. This can be an advantage over even-aged management in terms of conservation of old-growth attributes harbouring rare species (MacKinnon, 1998 ; Mosseler *et al.*, 2003), even though a much greater area needs to be harvested to achieve the same harvesting biomass. Indeed, the area harvested at each time step was about five times superior in scenarios using only uneven-aged management when compared with scenarios using only even-aged management (see Appendix B.8). Hence, certain uneven-aged management practices could be seen as promoting forest stands with more desirable attributes from both an ecological and aesthetic perspective, even though the younger and more early-successional forests generated with even-aged management can also present important attributes for both biodiversity and humans (Swanson *et al.*, 2011). However, these potential benefits of uneven-aged management may be limited by two factors.

First, forests with old-growth attributes are reduced when a stand-replacing disturbance such as fire occurs in the landscape, as evidenced by the stark difference in the amounts of old forests in the north and the south regions due to the different fire regimes between these two ecoregions. In the northern region, the total area of burned forest at the end of a simulation was about ten times larger than in the southern region (Appendix B.8), reducing the accumulation of old forests over time in that region. This suggests that stand-replacing disturbances constrain the capacity of uneven-aged management to preserve more mature forests. Critically, in future climates where natural disturbances such as forest fires are expected to be more frequent and severe, the ability of uneven-aged management to maintain old forests may be weakened (Dale *et al.*, 2001 ; Régnière *et al.*, 2012 ; Seidl *et al.*, 2017).

Second, while uneven-aged management can be perceived as less impactful than even-aged management at the local scale, it remains a disturbance to forests. Indeed, through the periodic returns to harvested stands, impacts on soil, understory, and local fauna may accumulate over time (Nolet *et al.*, 2018). Even if those impacts are reduced locally when compared to even-aged management (such as clearcutting), they extend across a larger surface in the landscape. This echoes some elements of the debate regarding *land-sparing* and *land-sharing* strategies (Edwards *et al.*, 2014). Hence, our study reveals some of the limits of using such an extensive *land-sharing* approach in a landscape to protect important ecosystem services. Yet, other arguments are present in the existing literature, advocating for the utility of both approaches (Edwards *et al.*, 2014 ; Mori et Kitagawa, 2014). Our results further reveal some interactions between even- and uneven-aged management. For instance, we observed a steep decline in the amount of old forest in the south of our study area, at the end of the simulation when the proportion of uneven-aged management reached 50% (Figure 2-5c). This decline results from the even-aged harvesting of old forests that had been preserved by prior uneven-aged cuts. In scenarios with less uneven-aged management (i.e., 0% and 25%), this decline is compensated by the higher quantity of younger forests that are in the process of transitioning to old forests. These trends suggest that the effects of both even-aged and uneven-aged management on forest age could become more similar over time, had we simulated these interactions beyond 150 years.

Our results also present artifacts resulting from both our definition of old forests and from our modelling choices. The first is represented by the sudden increase in the amount of old forests between 40 and 70 years of simulation especially in the south of our study area (Figure 2-5a, c) but also at 100 years of simulation in the north in scenarios with no uneven-aged management (Figure 2-5a). These increases are the result of many age cohorts of the landscape reaching the critical age of 91 years old from their initiation at the beginning of the simulation, or from earlier even-aged cuts and burned forests. Had we chosen a different definition of old forest and modelled all the disturbances that historically affected the landscape (see section "Limitations" below), this increase might have happened at a different time, or not at all. Still, we do not expect that this departure from the initial conditions of the landscape had any effect on our results or our conclusions since our analysis compared relative differences between scenarios.

2.4.3 Fragmentation of old forests

Our results point to a complex trade-off between the amount and fragmentation of old forests in uneven-aged management. Indeed, in the south of our study area, the consistent decrease of *Clumpy* with time in scenarios using uneven-aged management seems to indicate that while the number of old pixels increases, their contiguity is generally not preserved because of the higher road density required in these scenarios (Figure 2-5b, d, f, h). However, it should be noted that even-aged management did fragment old forests across the landscape through roads, and also by creating patches of younger forest. Intermediate scenarios also reveal changing trends in the values of *Clumpy* in the south of the area until the end of the simulation. This suggests that interactions between even-aged and uneven-aged cuts might have changed the results if the simulations had gone on for several decades. The potential effects of this trade-off are difficult to evaluate from an ecological perspective and will be highly dependent on how roads and younger forests actually impact species on the landscape. However, the presence of a stand-replacing disturbance seems to heavily influence this trade-off as this pattern almost disappears in the north of our study area characterized by much more frequent forest fires compared to the south. Therefore, the occurrence of stand-replacing disturbances should also be considered when comparing the effect of even- and uneven-aged management on old forests (see section 2.4.4).

Furthermore, the effect of fragmentation *per se* (i.e., differentiated from habitat loss) on biodiversity is currently a highly debated subject in the scientific literature, allowing for different interpretations of our results. Indeed, some authors argue that fragmentation has a generally neutral or even positive effect on biodiversity in the landscape (Fahrig, 2017) while others defend the opposing view (Fletcher *et al.*, 2018). Therefore, questioning the long-term, large-scale effects of uneven-aged management on biodiversity through its impact on habitat connectivity might provide different conclusions. For example, species such as the Pacific marten (*Marten caurina*) tend to select forest stands with a higher structural complexity to ease their movement in the landscape, avoiding "simpler" stands with characteristics resembling even-aged stands (Moriarty *et al.*, 2015). Although uneven-aged management creates more numerous and dispersed smaller roads, even-aged management generates fewer but bigger roads, and may have less impact on the terrestrial fauna that tends to avoid these larger pathways (Tittler *et al.*, 2012). As our study focused on structural connectivity at the landscape scale, we expect that pertinent information will come from future research exploring different management scenarios similar to this study but taking into account the effect on the functional connectivity of different species. Thus, differentiating the effect of even and uneven-aged management on biodiversity will require comprehensive functional connectivity analyses.

Surprisingly, changing the aggregation levels of the cuts had little to no effect on the fragmentation of old forests as measured by *Clumpy* (Figure 2-5f, h). This result could be explained by the fact that aggregation only slightly reduces the density of forest roads in the landscape (Figure 2-4c), which is an important element of fragmentation captured by *Clumpy*. Moreover, *Clumpy* is sensitive to the presence of patches of young forests that are generated by even-aged management and the occurrence of fire, but these patches are absent under uneven-aged management. Therefore, *Clumpy* was relatively unaffected when areas harvested with uneven-aged management were aggregated, and when the aggregation of young forest patches created by even-aged management was weakened by forest fires, which vary in their location and extent. The effect of forest fires is discussed in more detail in the following section.

2.4.4 Interactions with stand-replacing disturbances

Two types of fragmentation of old forests were operating in our simulated landscape: fragmentation due to forest roads and due to patches of younger forests resulting from stand replacing disturbances. In our simulations, the former is increased using uneven-aged management which increases road density (Figure 2-6a) while the latter is increased using even-aged management which creates patches of regenerating forests after clearcutting (Figure 2-6b). Indeed, the fragmentation of old forests was stronger than anticipated under even-aged management because it compounded the two types of fragmentation: one as a result of harvesting and the other due to construction of forest roads. However, the fact that uneven-aged management was associated with higher levels of fragmentation in the south of our study area indicates that roads had a bigger effect on *Clumpy* than the patches of young forest in this landscape.

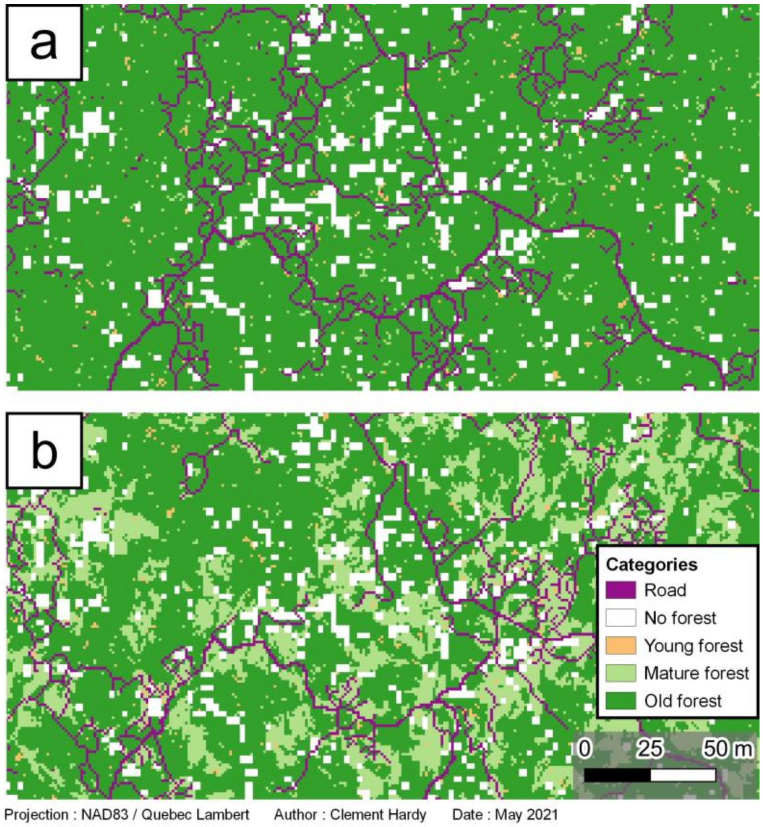

Figure 2-6 : A section of our simulated landscape at t = 100 years during a simulation where a) only uneven-aged management is used, and b) only even-aged management is used. Young forests are defined as pixels with trees no older than 15 years; mature forests are pixels whose oldest trees range between 15 and 91 years of age; old forests are pixels comprising trees older than 91 years. It clearly appears that the fragmentation of old forests is associated with the extent of the road network under uneven-aged management (a) while it is also associated with the distribution of patches of younger forests under even-aged management (b).

With these two fragmentation types present throughout the study area, the factor responsible for the difference observed in the values of *Clumpy* between the north and the south of our study area becomes clearly apparent (Figure 2-5b, d), namely forest fires. In our simulations, the harvested surface was similar in both regions of our study area, but the total burned surface was almost 10 times superior in the north than in the south, increasing the area of young forests (see Appendix B.8). Consequently, our results suggest that fragmentation in the northern region is mainly driven by forest fires, as varying the proportions of uneven-aged management only slightly affected *Clumpy* (Figure 2-5b). On the other hand, the southern region, which is almost unaffected by forest fires, displays clear differences among uneven-aged management scenarios (Figure 2-5d).

Therefore, our simulation results suggest that stand-replacing disturbances, such as forest fires, can reduce the differences observed between even- and uneven-aged management in terms of fragmentation and amount of old forests. This could be crucial in areas such as the boreal forest, where frequent forest fires would reduce the capacity of uneven-aged management to preserve more mature forests (see section 2.4.2). Nonetheless, this lower quantity of old forests in the north is expected no matter the type of harvesting used as the fire regime naturally leads to a lower proportion of old forests. Indeed, using the equations from Van Wagner (1978) and the fire zone characteristics of Boulanger *et al.* (2014), it is expected that the northern region of the area would contain 20% fewer forests older than 90 years than the southern region at a theoretical equilibrium induced by the fire regime of both regions. This 20% difference is approximately what we observed in our results, when comparing the amount of old forests in the north and the south of the study area after about 60 years of simulation, for similar management scenarios (Figure 2-5a, c). Consequently, the differences in the effect of *land-sharing* and *land-sparing* management approaches on the amount and fragmentation of old forests will heavily depend on the occurrence, extent and severity of natural disturbances.

Our results also suggest that uneven-aged management in the boreal forest increases the fragmentation by forest roads in a landscape already fragmented by patches of younger forest (due to forest fires). In contrast, even-aged management could be seen as emulating a type of fragmentation already present in boreal forests (patches of younger forests), while reducing a second type of fragmentation not naturally present (forest roads). Nevertheless, forest roads created for uneven-aged management could facilitate access to recently burned forests for salvage logging and reforestation purposes (Cyr *et al.*, 2022). In addition, even-aged management could bring an additional fragmentation to that already caused by forest roads in regions where stand-replacing disturbances are less present (Guldin, 1996). This distinction of where best to use even-aged or uneven-aged management touches on one of the key concepts of ecosystem-based management: sustainability of forests can be achieved under management methods able to reproduce their natural disturbance regime (Bergeron *et al.*, 2002 ; Gauthier et Vaillancourt, 2008 ; Harvey *et al.*, 2002). Furthermore, the large surfaces of forest and the dense road network needed for uneven-aged management could also present an advantage in the context of global changes. Indeed, the uneven-aged harvested forests could be managed in a way that helps them transition more rapidly to a different state of structure and composition that would make forests more resistant and resilient to the new conditions created by global changes (Messier *et al.*, 2019).

While both the fragmentation due to roads and to patches of younger forest are operating simultaneously, their effects are not equivalent. Indeed, forest roads are associated with diverse management and ecological issues, such as the spread of invasive species (Meunier et Lavoie, 2012 ; Mortensen *et al.*, 2009), collisions with fauna (Lugo et Gucinski, 2000), increased presence of humans (Gucinski, 2001), and corridors or barriers to animal movement that can disturb their population dynamics (Marsh *et al.*, 2005 ; Whittington *et al.*, 2011). It has also been suggested that roads create up to twice the amount of habitat edges than clearcutting does (Reed *et al.*, 1996). Moreover, forest road usage is increased under uneven-aged management since it requires periodic re-entry into stands and a larger surface to harvest (Nolet *et al.*, 2018), causing an increase in the transit of forestry vehicles which in turn intensifies disturbances on fauna and flora, and emits additional greenhouse gases. Forest roads also represent a constant loss of productive forest surface over time, contrary to young forests, as shown by Figure 2-4a where up to 12% of the cells of our landscape are continuously occupied by forest roads after the first 50 years of simulation. Although individual forest roads deteriorate over time, if forest management is maintained, other roads will be reconstructed elsewhere. For their part, young forests generated by even-aged management will eventually grow into mature forests. However, if stand-replacing harvesting is maintained across the landscape, old forests will be reduced. Moreover, forest regeneration must be ensured following even-aged cuts by the proximity of mature trees and seed banks, the planting of saplings, or the protection of the youngest cohorts during harvesting. If these three elements are missing, which might be the case in some contexts, then forest regeneration will be compromised, leading to long-lasting habitat loss (Timoney et Peterson, 1996).

2.4.5 Limitations

Our modelling approach presents some limitations worth mentioning. First, our results are highly dependent on the way we define old forests. We followed the definition of old forest used by the MRNF: a forest containing at least one age cohort older than 91 years (MFFP, 2016). Yet this definition does not fully capture the concept of old-growth forest often used when discussing the conservation value of older forests, which is both complex and context-dependent (Hilbert et Wiensczyk, 2007 ; Wirth *et al.*, 2009). An "old-growth forest" (McMullin et Wiersma, 2019) can refer to a certain forest structure (e.g., old and large trees, logs and snags or a wide distribution of tree size), specific successional processes (e.g., climax forest or steady-state condition), or even certain biogeochemical processes (e.g., decay of snags and logs or nutrient retention) (Wirth *et al.*, 2009). Moreover, an old-growth forest in a boreal biome can be much younger than what would be considered an old forest in other biomes, e.g., a tropical biome. In addition, the definition of old forest that we used does not distinguish between an “untouched” old forest that has not been harvested or disturbed for a long time (i.e., an old-growth forest) and a forest harvested by uneven-aged management, but that still presents some relatively old cohorts (i.e., a forest with old trees). Therefore, we expect that uneven-aged management would have fared worse regarding the fragmentation of “untouched” old forests. This could be important for species that are sensitive to the quality rather than the quantity of available old forests (e.g., Regolin *et al.* 2021). Lastly, while we did not measure the diversity in forest age classes across the landscape, it can be argued that the quantity of old forests can capture this structural heterogeneity. Indeed, old forests tend to contain more age classes even if originally disturbed by a stand replacing disturbance such as a severe forest fire or even-aged cut (Bergeron, 2004 ; Martin *et al.*, 2022).

A second limitation comes from the geographical context of our study area. The Mauricie region is characterized by vast expanses of mixed and boreal forests located far away from the main road network or any community. In more densely populated regions where the permanent road network is closer to harvested areas, we expect that increasing the use of uneven-aged management would not necessarily lead to an increase in forest road density. In addition, had we simulated epidemics of spruce budworm, we would likely have observed a less important difference in the proportion of old forests between the southern and northern regions. This significant agent of disturbance affects mature fir-spruce stands in the south of our study region but does so less frequently in the north where the growing season is too short for completing their life cycle (Régnière *et al.*, 2012). Tree mortality due to repeated years of spruce-budworm defoliation contributes to the rejuvenation of old stands.

A third limitation is that our results may be sensitive to the particularities of the forestry methods that we simulated, which are based on the common harvesting prescriptions in Quebec. As such, our results may not be generalized to different combinations of harvesting methods used in other countries. In addition, the harvesting stand prescriptions that we simulated with LANDIS-II are quite simple and broad in their application. This is because LANDIS-II currently does not allow prescriptions based on a complex assignment of stands including, for example, species, age structure, or soil conditions. As such, further studies might be needed to explore the nuances that might result from more complex prescriptions assignments.

Fourthly, we did not include the effect of a changing climate in our simulations, even if their effects on forest regeneration, precipitation, and fire regimes are expected to take place on a 150 year horizon. Indeed, in Quebec, climate change is predicted to increase the frequency of forest fires and affect the growth of many tree species (Boulanger *et al.*, 2023). To better distinguish and interpret differences resulting from different management strategies, and to reduce the computational load of the simulations, we opted not to consider the impact of climate change. But as highlighted previously, more frequent fires might reduce the differences in fragmentation and quantity of old forest observed in our study between even-aged and uneven-aged management scenarios. Additionally, changes in species growth might require different harvesting prescriptions to facilitate the regeneration of species more impacted by climate change in the future.

Fifthly, we did not simulate the different processes in which forest roads, forest management and forest fires might interact in reality. Roads can serve as fire-breaks, as do clear-cuts generated by even-aged management (Narayanaraj et Wimberly, 2011), and can additionaly help firefighters move more easily in the landscape. But they can also increase the amount of ignitions by increasing human presence in remote areas, or by creating dryer conditions in the surrounding forests (Narayanaraj et Wimberly, 2012). As such, it is difficult to predict how different our results would have been if such effects had been taken into account, but it could have resulted in an increase or decrease in the amount of area burned. In turn, this could have affected our measures realted to old forests in the north of the area.

Finally, it is important to recall that the density of forest roads (*Dr*) reported here corresponds to the proportion of cells in the landscape that contain a road. Because the actual surface occupied by a road is smaller than the spatial resolution of the model (1 ha), the road density variable certainly overestimates the density of forest area transformed into roads. Thus, while forest roads were present in up to 12% of the cells of our simulated landscape (Figure 2-4a), they did not occupy 12% of the surface of the landscape.

2.5 Conclusion

Our simulation study demonstrates that a *land-sharing* strategy in forestry, characterized by a prevalence of uneven-aged management methods,can increase the density of roads, the level of fragmentation, the cost of road construction and repair, and the amount of forests with old-growth attributes in the landscape. It also suggests that fragmentation could be slightly lowered by aggregating harvested areas. The presence of an initial road network did not alter fragmentation in the long term. Thus, our study shows that despite the potential benefits of uneven-aged management for conservation at the stand level, it can contribute to additional landscape fragmentation when compared to even-aged methods. This finding, along with the additional costs associated with the construction of a more extensive road network, needs to be considered when balancing objectives through strategic forest management planning.

Our results also imply that choosing between even- and uneven-aged management involves a trade-off between the proportion of old forests in the landscape and their level of fragmentation. However, the consequences of this trade-off are closely linked to the specific even- and uneven-aged methods employed, the definitions used for fragmentation and habitat, the patterns of fragmentation resulting from natural disturbances, and the perceived effects of fragmentation on the landscape. Overall, our study indicates that uneven-aged management at the landscape scale, which can be considered as a *land-sparing* strategy, is not necessarily better than even-aged management, a *land-sparing* strategy, for conserving forest ecosystems. Hence, our study emphasizes the notion that over large spatiotemporal scales, no single forest management strategy is both economically and ecologically "better" than another (Nolet *et al.*, 2018 ; Puettmann *et al.*, 2009).

While neither even-aged nor uneven-aged management is without flaws, we anticipate that a mix of these two methods, implemented in the right place, could present the best compromise. Indeed, management scenarios that combined both methods presented intermediate values in all the measured variables, suggesting that their use could be fine-tuned to obtain the desired compromise between the conservation of certain forest habitats, and the construction of forest roads to maintain timber production. To further optimize such fine-tuning, management plans that strategically position areas harvested with uneven-aged management could reduce landscape fragmentation. For example, clustering areas managed near existing roads or organizing the landscape into zones of different harvesting intensities (as in the TRIAD approach) could reduce the need for additional roads when harvesting larger surfaces, thereby facilitating connectivity between sensitive or important habitats (Messier *et al.*, 2009 ; Tittler *et al.*, 2012).

2.6 Acknowledgments

We would like to thank Jérôme Garet of the MRNF for helping us gather valuable data to parameterize the FRS module.

2.7 Funding

Funding was provided by the Natural Sciences and Engineering Research Council of Canada (NSERC) through a Collaborative Research and Development grant awarded to C. Messier (RDCPJ: 498998-16) with the joint support of Resolute Forest Products, and a Discovery grant awarded to E.Filotas (RGPIN: 2018-06156).

2.8 Competing interests

The authors have no competing interests to declare that are relevant to the content of this article.

2.9 Author contributions

Clément Hardy: Conceptualization, Methodology, Software, Validation, Formal Analysis, Investigation, Data Curation, Writing – Original Draft, Visualization; **Christian Messier**: Conceptualization, Methodology, Resources, Writing – Review & Editing, Supervision, Funding Acquisition; **Yan Boulanger**: Software, Data Curation, Writing – Review & Editing; **Dominic Cyr**: Software, Data Curation, Writing – Review & Editing; **Elise Filotas**: Conceptualization, Methodology, Investigation, Resources, Writing – Review & Editing, Supervision, Project Administration, Funding Acquisition.

2.10 Data availability

The data concerning the preliminary simulations to calibrate certain parameters of LANDIS-II, all of the parameter files for all of the simulations, the scripts used to launch the simulations on Compute Canada's clusters, the raw results and the scripts used to analyze the results and produce the figures are all available on the following figshare private repository: https://figshare.com/s/1e84862cf4114b336a7f . The data of the repository will become public and identified with a DOI once the manuscript will be ready to be published.

CHAPITRE 3

Climate is stronger than you think: Exploring functional planting and TRIAD zoning for increased forest resilience to extreme disturbances

Ce chapitre est, à la date de la soumission de cette thèse, prêt pour publication dans une revue scientifique dont le choix est à venir. Il a été rédigé par les personnes suivantes :

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Résumé : Dans ce chapitre, nous présentons le concept d'une nouvelle stratégie d'aménagement forestier qui combine l'aménagement en TRIAD à un régime de plantations dites « fonctionnelles » dont le but est d'augmenter la diversité fonctionnelle des peuplements traités. Le but de ces plantations fonctionnelles est alors d'augmenter la résilience des forêts traitées face à des perturbations futures incertaines, en augmentant leur diversité de traits de réponse. Ce nouvel aménagement – nommé TRIAD+ – devrait ainsi présenter un bon compromis entre les objectifs de conservation et de production (par le biais du zonage en TRIAD), mais aussi d'adaptation (par le biais des plantations fonctionnelles). Nous avons testé la capacité de la TRIAD+ à augmenter la résilience des forêts d'un paysage en le comparant à 3 autres scénarios d'aménagement : un aménagement en TRIAD « normal » sans plantations fonctionnelles, et deux scénarios *Business as Usual* (BAU), un avec et un autre sans plantations fonctionnelles. Nous avons simulé l'application de ces 4 stratégies d'aménagement dans le même paysage simulé dans le chapitre 2

situé dans la région de la Mauricie, au Québec, durant 200 ans de simulation. Nous avons testé la capacité de chaque scénario d'aménagement à augmenter la résilience des forêts du paysage face à une de trois perturbations extrèmes survenant en milieu de simulation (t = 100) : un feu touchant tout le paysage, une sêcheresse sévère et une épidémie de dendrochtone du pin. De plus, nous avons fait varier le climat au sein des simulations selon trois scénarios climatiques différents (référence, RCP 4.5 et RCP 8.5) pour affecter la croissance des espèces et le régime de feu dans la région. Nous avons au final simulé chaque combinaison unique d'aménagement, de perturbation extrème et de climat, et avons mesuré trois variables durant les simulations : la biomasse des arbres matures du paysage, la biomasse mature des différents groupes fonctionels d'arbres dans le paysage, et la diversité fonctionnelle des peuplements du paysage. Nous avons également calculé trois mesures de résilience à l'échelle du paysage suite à l'activation des perturbations extrèmes : la résitance, le changement net et le taux de récupération de la biomasse mature du paysage. La TRIAD+ a résulté en un bon compromis en augmentant la diversité fonctionnelle des peuplements et la résilience de leur biomasse mature en comparaison à la TRIAD normale et au BAU sans plantations fonctionnelles. De plus, la TRIAD+ a permis de récolter la même quantité de bois tout en réservant une plus grande surface de forêt à des fins de conservation que les scénarios BAU. Cependant, l'augmentation de la diversité fonctionnelle des peuplements et de la résilience des forêts du paysage ont été limitées, et le climat était le principal facteur affectant nos différents variable ainsi que nos mesures de résilience. Notre étude révèle ainsi des informations cruciales pour le dévellopement de nouvelles stratégies d'aménagement, mais aussi sur la practicalité des plantations fonctionnelles pour augmenter la résilience des forêts. Elle souligne également le besoin urgent de mieux simuler la dynamique des peuplements forestiers dans les modèles à l'échelle du paysage.

3.1 Introduction

Forests occupy a special place among the many components of the earth's biosphere. Holding much of the terrestrial biodiversity, they are crucial to the processes of life on Earth (FAO et UNEP, 2020). Forests also regulate the Earth's climate through their influence on the biogeochemical cycles of water, carbon and nitrogen, thereby playing an important role in climate regulation (Ellison *et al.*, 2017 ; Fowler *et al.*, 2013 ; Mitchard, 2018). But for humans, they also represent places of wonder; of spirituality; homes, and vital sources of different resources, ranging from construction material to food or medicine (Clark, 2011 ; FAO et UNEP, 2020). Yet these numerous roles played by forests are now under threat from many different environmental pressures (Burrows *et al.*, 2011 ; Millar et Stephenson, 2015 ; Referowska-Chodak, 2019). Chief among them is global change – the combination of direct anthropic pressures, human-induced climate change, and other processes interacting with humans such as the invasion of exotic species and pathogens (Seidl *et al.*, 2020 ; Steffen *et al.*, 2006 ; Vitousek, 1994).

Forestry is one of the pressures that global change applies on forests. While humanity has been harvesting wood from time immemorial, the quantity of wood harvested throughout the world has grown enormously since the Industrial Revolution. In the recent years, this quantity surged from 2.5 billion m^3 of roundwood per year produced worldwide in 1960 to almost 4 billion m^3 in 2020 (FAOSTAT, 2019a, 2019b) through increasing human demographics and the emergence of new harvesting technologies that made harvesting cheaper and faster (FAO et OAA, 2012). Wood and timber production worldwide has shown no sign of slowing and is instead increasing yearly. Forestry is thus a recurrent and intense disturbance affecting forests around the world, leading to their gradual transformation. This transformation occurs through practices such as the selection of species of commercial interest (Chaudhary *et al.*, 2016), or through the large-scale application of methods such as clear-cutting that reduces the quantity of older forests (Cyr *et al.*, 2009). Forestry has thus changed the structural diversity (Dieler *et al.*, 2017 ; Martin *et al.*, 2020), species composition (Torras et Saura, 2008), and connectivity (Haddad *et al.*, 2015) of forests at large temporal and spatial scales. Consequently, forestry also altered the habitat that forests provide to many other species and tempered their ability to sustain important ecological functions (Betts *et al.*, 2021b).

In contrast to these trends, several authors have recently proposed a more optimistic view of forestry. In this view, the inherent capacity of forestry to influence the structure and composition of forests could be used as an advantage, rather than being an inconvenience. Specifically, forestry could be used as an opportunity to diversify forests across different spatial scales to improve their resilience to the uncertain future perturbations caused – or influenced – by global changes (Messier *et al.*, 2013, 2015 ; Sasaki *et al.*, 2015). In that way, forestry would join the global effort to increase the diversity of responses that both human and natural systems exhibit in the face of disturbances (Loreau *et al.*, 2021 ; Walker *et al.*, 2023). A potential way to bring ecosystems to a state of higher resilience is by using the concepts of functional response traits and response diversity. Functional response traits aim to capture the biological, structural or behavioral characteristics of an individual associated with its response to environmental changes (e.g., root length, bark thickness, etc.; (Díaz *et al.*, 2013)) and its effects on the environment. As such, a community presents a range of responses to the environment among its organisms (e.g. individuals, species) depending on the variability between their functional response traits (Ross *et al.*, 2023). Therefore, it is suggested that a community with a high response diversity will be more resilient to disturbances, as it increases the chance that some individuals have the favorable traits to resist or recover (de Bello *et al.*, 2021 ; Messier *et al.*, 2019 ; Mori *et al.*, 2013 ; Oliver *et al.*, 2015). Forests could thus be diversified or altered by forest management practices at both stand and landscape levels to present a higher functional response diversity to future disturbances, and, as a result, a greater resilience (Aquilué *et al.*, 2020 ; Mina *et al.*, 2022 ; Seidl *et al.*, 2016).

While this new vision of forestry based on preparing forests for future conditions is promising, it differs from current forest management strategies in several ways. The main difference relates to the prevailing management goal of conserving or retrieving a "reference state" of forests. For example, strategies adopted in many temperate and boreal forests promote an ecosystem-based forest management (Gauthier et Vaillancourt, 2008) or close-to-nature forestry (Brang *et al.*, 2014 ; Franklin *et al.*, 2018), which focus on emulating the "historical range of variation" of natural disturbances through forest management. According to this approach, forest cuts should reproduce the size, intensity and impacts of natural disturbances with which forests have evolved and developed regeneration mechanisms. Yet, applying this approach is complex due to the difficulty of obtaining the reliable historical characteristics of natural disturbances. Moreover, recent studies have criticized implementations of ecosystem-based management that focus exclusively on reproducing past disturbances patterns as current and future forests are likely to be confronted to novel environmental conditions and disturbance regimes (Keane *et al.*, 2009 ; Messier *et al.*, 2019 ; Puettmann *et al.*, 2015). Hence, adding forest resilience to global change as a new management goal may be at odds with current management strategies that are largely focused on wood production and conservation.

A potential approach to address this challenge is to amend current management strategies by integrating the notion of resilience, adaptation and guided change of forest structure and composition. In this article, we propose a resilience-based modification of the TRIAD zoning approach, originally developed by (Seymour et Hunter, 1992). The conventional TRIAD approach consists in dividing a managed forest landscape into different specialized areas dedicated to different goals: intensively managed areas to maximize production; extensively managed areas to accommodate a broader range of ecosystem services and conservation objectives, and reserves for conservation purposes only (Messier *et al.*, 2020). These zones are positioned in the landscape in order to minimize trade-offs between conservation, production, and social acceptability (e.g., setting intensive areas far from conservation areas) while supplying the same amount of harvested wood as a landscape without zoning (Côté *et al.*, 2010). Intensive zones aimed at maximizing productivity such that harvest targets are reached more easily on these smaller areas, thus allowing to increase the size of conservation areas (Tittler *et al.*, 2016). Because of these potential benefits, the TRIAD zoning is currently being tested in several areas of the world (Blattert *et al.*, 2023 ; Tittler *et al.*, 2016).

The TRIAD approach offers the opportunity to accommodate the new resilience objective by   increasing the functional response diversity of forest stands in extensively managed zones. This can be done through enrichment planting, which consists of planting trees into an already existing forest overstory that has been thinned (Paquette *et al.*, 2006). It is further called functional enrichment or functional planting when used to diversify the functional response traits present in the forest rather than simply improving species richness (Aquilué *et al.*, 2020). Therefore, functional enrichment in forests could improve their ability to respond to future disturbances as they would contain a more diversified portfolio of response traits (Aubin *et al.*, 2016 ; Mori *et al.*, 2013). Through long-term planning, the establishment of functionally enriched plantations distributed across the landscape could allow resilience to scale up from the plantation to the landscape scale through seed dispersal (Craven *et al.*, 2016 ; Messier *et al.*, 2019). We call TRIAD+ this new version of TRIAD that includes functional enrichment via plantations.

While this avenue is appealing, several factors must be investigated to assess the efficacy of functional enrichment plantations – and of the TRIAD+ as a whole – to improve forest resilience. As an example, future climatic conditions could make it difficult for the species selected for functional diversification to coexist at the stand scale if their growth is impeded by changes in temperature or water availability. Also, the accumulation of more frequent, severe and varied disturbances might increase tree mortality or hamper their growth even in functionally rich plantations (e.g., through repeated fires, droughts, windthrow, etc.; (Stevens-Rumann *et al.*, 2018)). Functional enrichment may also require productive sites to be successful, thereby making these sites unavailable for intensive management areas. Finally, implementing functional enrichment plantations will often require preliminary forest cuts that, if too numerous in the landscape, might negatively impact forests in other ways, canceling the positive effects of such enrichment. For example, such cuts might change the age distribution towards younger forests in the landscape (Cyr *et al.*, 2009), necessitate more forest roads (Hardy *et al.*, 2023a), and degrade the habitat quality for certain wildlife specialist species (Baccini *et al.*, 2017 ; Betts *et al.*, 2021b).

In this study, we set out to explore these uncertainties by measuring the potential of functional enrichment planting to improve the resilience of a vast forest landscape to different disturbances. Specifically, we assessed the capacity of a TRIAD+ zoning approach at improving the resilience of the mature forest biomass to climate change and three future potential extreme disturbance events: a large fire, a severe drought, and an insect outbreak. To these ends, we used the forest landscape simulation model LANDIS-II (Scheller *et al.*, 2007) on a management unit in Mauricie (Quebec, Canada). We simulated and compared the TRIAD+ scenario with a classic TRIAD zoning scenario, and two business-as-usual harvesting scenarios with and without functional enrichment planting. Furthermore, we simulated three different climate change scenarios (Baseline, RCP 4.5 and RCP 8.5) in which these management and the extreme disturbance scenarios took place. We then measured the resilience of the mature forest biomass, at the landscape scale, following one of the three extreme disturbance events (fire, drought or insect outbreak). Finally, we assessed the benefits of each management scenario by considering the trade-offs between the objectives of production, conservation and adaptation.

3.2 Methods

3.2.1 Simulated area

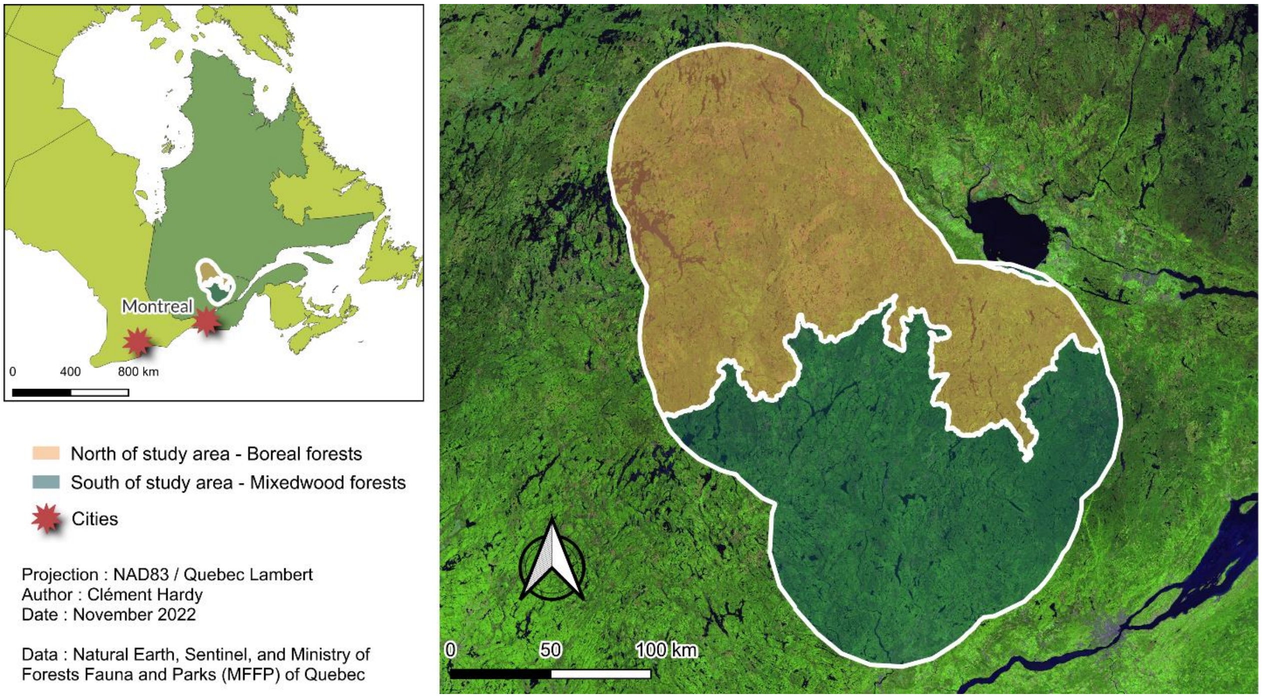

Figure 3-1 : Map showing the location and extent of our study area located in the Mauricie region of Quebec, Canada.

Our simulated area is a forest landscape extending over more than 4 million hectares in the Mauricie region (Quebec, Canada; Figure 3-1). It consists of a forest management unit surrounded by a 50 km buffer zone, which has been simulated with LANDIS-II in a previous study (Hardy *et al.*, 2023a). As of 2020, the landscape comprised 339 117ha of protected forests or around 9% of the total forest surface of the simulated area. These protected forests were divided approximately among 400 areas, five of them being relatively large (> 20 000 ha) and most of them relatively small (< 300 ha). The south of the area is mainly composed of mixedwood forests dominated by balsam fir (Abies balsamea), yellow birch (Betula alleghaniensis) and trembling aspen (Populus tremuloides). In contrast, the north of the area contains boreal coniferous forests with a dominance of balsam fir, white birch (Betula papyrifera), trembling aspen, black spruce (Picea mariana) and jack pine (Pinus banksiana). Forest fires are an important disturbance in the north, while spruce budworm outbreaks are more present in the south (Bergeron et Fenton, 2012 ; Boulanger *et al.*, 2012). Being located at the transition from the temperate to the boreal forest, this study area thus presents a clear dichotomy of forest composition and natural disturbances, making it a good choice to explore the effects of functional enrichment and zoning in two contrasting ecological contexts.

3.2.2 Experimental design

To explore the effects of functional enrichment and zoning on forest resilience, we simulated forest dynamics over 200 years using different scenarios that varied three distinct factors: the forest management strategy used, the intensity of climate change, and the occurrence of a catastrophic disturbance event imposed at year 100of the simulation (Figure 3-2). Here, we outline the factors and their levels. Their precise implementation within LANDIS-II is detailed in the following sections.

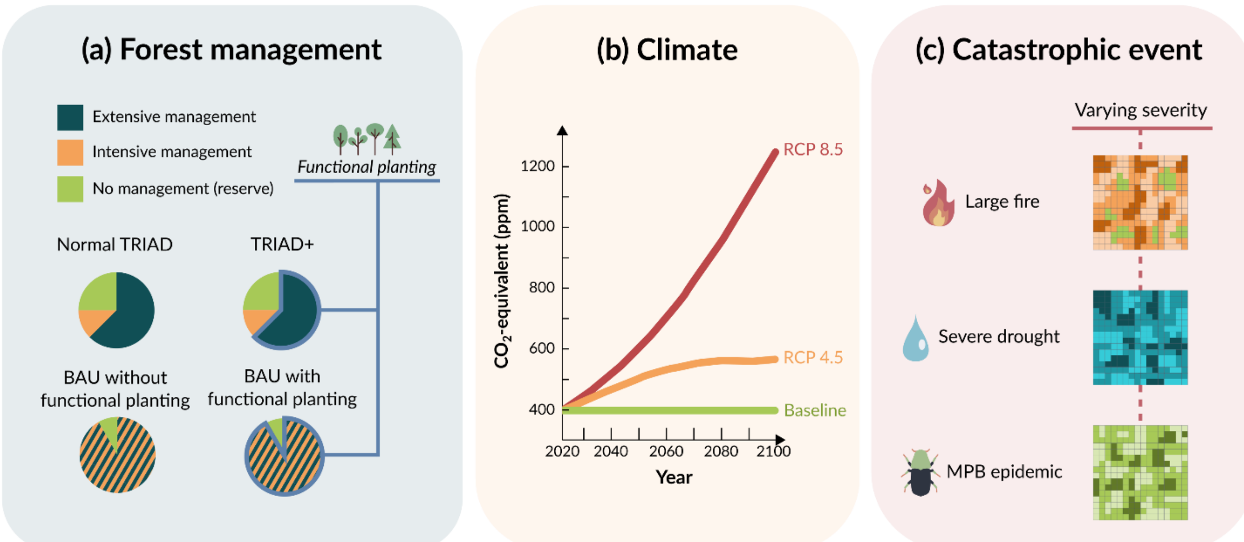

Figure 3-2 : Visual representation of the three categories of scenario simulated with LANDIS-II: forest management, climate, and catastrophic event.

We designed four forest management strategies each defined by the presence or absence of functional enrichment through planting (hereinafter referred to as "functional planting") and of TRIAD zoning (Figure 3-2a). The first two forms were a "Business as Usual" (BAU) scenario with (BAU-PlantFunct) and without (BAU-NoPlant) functional planting. In these scenarios, no TRIAD zoning took place: forest harvesting of different intensities could be carried anywhere in the landscape except in the current protected areas. All simulated prescriptions are detailed in section 3.2.3.5. In essence, the BAU-NoPlant scenario acted as a control scenario for our study area. The other two forms were TRIAD+ and the "normal" TRIAD without functional planting. In these scenarios, the landscape was divided into intensive, extensive and protected areas, with functional planting taking place in the extensive areas of TRIAD+, and intensive areas located on the most productive forests of the landscape (see section 3.2.3.5). The existing protected areas were expanded using a 5000-m buffer compared to the BAU scenarios, increasing their total percentage from around 9% to 24% of the forested area. This allotment was devised to match the philosophy of TRIAD zoning, where intensive areas are made as productive as possible to spare more forest from exploitation (Messier *et al.*, 2020 ; Tittler *et al.*, 2016). Importantly, all four management scenarios had to harvest the same amount of biomass at every 10-year time steps. This biomass target was based on the current harvest levels in the simulated area during the period 2018-2023, as indicated in documents produced by the Ministère des Forêts, de la Faune et des Parcs du Québec (Hardy *et al.*, 2023a ; MFFP, 2018b).

The climate varied according to three different Representative Concentration Pathway (RCP) scenarios (Moss *et al.*, 2008): Baseline (no change in climate as compared to 2020) which served as a control, RCP 4.5 and RCP 8.5 (Figure 3-2b). Three catastrophic disturbances were chosen to represent potential – but quite unpredictable – future disturbance events that could take place in our study area, triggered by the effects of global change. Our disturbance scenarios consisted of a large forest fire covering 70% of our simulated area; an intense drought; and an outbreak from an insect not yet present in the area (Figure 3-2c). We also simulated scenarios without the occurrence of these catastrophic disturbances to serve as a control. Fire and droughts were chosen as they are both expected to increase in frequency and severity with climate change, making the occurrence of extreme cases more probable (Seidl *et al.*, 2017, 2020). In Quebec, a single fire of more than 1.2 million hectares was indeed observed in 2023 (NASA, 2023) during a fire season that resulted in more than 5 million hectares of forest burned at the provincial scale (Paddison, 2023). This fire season was caused in parts by droughts (Jain *et al.*, 2024), which as predicted by Global Climate Models will become more frequent and intense in the future (Zhao *et al.*, 2020). Such large fires could become even larger in the future, according to observed trends in Canada (Hanes *et al.*, 2019). For the insect pest, we selected the Mountain Pine Beetle (Dendroctonus ponderosae; MPB), a wood-boring species of bark beetles whose ongoing outbreak in western Canada is causing excessive damages by attacking a wide range of pine species. Previous studies have shown expansion of the MPB into the forest of eastern Canada as a possibility in the future, helped by a changing climate and by the presence of host species (Cooke et Carroll, 2017 ; Nealis et Peter, 2008 ; Safranyik *et al.*, 2010). Therefore, we defined a scenario where a large outbreak of MPB would impact the pine species of our landscape, to which forest managers would be initially unprepared. Simulating this disturbance allowed us to study the potential perverse effect of functional plantations which by increasing pine trees in the landscape would increase its sensitivity to MPB outbreaks.

We simulated one scenario for each unique combination of these three factors (management, climate, and catastrophic disturbances), resulting in 48 distinct scenarios. We accounted for stochasticity associated with wildfires, seed dispersal and regeneration by running five replicates for each scenario, resulting in an ensemble of 240 individual simulations. In each simulation, we measured two variables of interest at the landscape scale (total mature biomass and mean functional response diversity) and we assessed the variations between these measures according to the three different factors (management, climate, catastrophic disturbances). We also measured three indicators of resilience to the catastrophic disturbances (see section 3.2.4) to evaluate whether the different forest management strategies were associated with increased or decreased forest resilience values. Moreover, we assessed whether an increased resilience implied a trade-off with another variable of interest (e.g., mature biomass).

3.2.3 LANDIS-II model

LANDIS-II is a spatially explicit Forest Landscape Model (FLM) that simulates forest dynamics via two main processes: forest succession (growth, mortality, recruitment, etc.), and natural or human-induced forest disturbances (harvesting, forest fire, insect outbreaks, etc.) (Scheller *et al.*, 2007). The processes are individually simulated by extensions that are activated sequentially during each time step. These extensions are chosen by the user and can simulate the dynamics of different ecological variables (e.g., biomass, carbon stocks, , etc.). In LANDIS-II the simulated landscape is composed of square cells representing a forested or non-forested area (e.g., water, urban area, etc.). All forested cells are assigned to different ecoregions to integrate the effect of climate and soil in the simulated processes.

3.2.3.1 Core parameters

The main parameter values used in our study were derived from the protocol of several recent studies that used LANDIS-II to simulate forest landscapes in Quebec (Boulanger et Pascual Puigdevall, 2021 ; Tremblay *et al.*, 2018). We simulated 17 different tree species that were among the most abundant in our study area, with their life-history traits being derived from several sources (books and previous studies; see appendix C.1 and (Boulanger *et al.*, 2017)). We used a grid size of 100 m (1 ha) and a time step of 10 years as a compromise between computation time and level of detail, as is often done in LANDIS-II studies (Boulanger *et al.*, 2017 ; Shinneman *et al.*, 2010 ; Sturtevant *et al.*, 2012). Finally, we set the total simulation length to 200 years in order for the simulated forest management strategies to take effect in the landscape during the first 100 years, and then measured the response of stands impacted by the catastrophic disturbance events happening at t = 100 during the remaining 100 years.

The initial composition and structure of the forest in each grid cell were established using ecoforestry maps from the province and data from cohort studies conducted in the province's permanent and temporary forest inventory plots (Alain *et al.*, 2016b, 2016a ; MFFP, 2018a). The composition and age structure of the inventory plots were translated into LANDIS-II age-cohorts. These age cohorts were then assigned to the forest stands identified in the ecoforestry maps through a k-NN assignation method (Boulanger *et al.*, 2017). Furthermore, we defined forest stands as groups of forested cells having identical composition, age structure and abiotic conditions according to the 5th provincial forest inventory of Quebec (MFFP, 2018a). The position of forest stands remained constant through time.

These maps were converted into a raster format with a resolution of 250 meters (equivalent to 6.25 hectares). Subsequently, each cell was allocated to a uniform spatial unit, known as a "landtype," characterized by consistent soil and climatic conditions (Boulanger et Pascual Puigdevall, 2021). Cells where over half of the area was occupied by non-forest cover types were categorized as non-active.

3.2.3.2 Biomass Succession

We used the Biomass Succession extension (v5.2) of LANDIS-II to simulate the succession dynamic and the living aboveground tree biomass of age cohorts within forest cells. This extension uses three important parameters that vary for each species, ecoregion, and time step: the probability of establishment, the maximum growth rate, and the maximum biomass that an age cohort can reach. These parameters were obtained using PICUS (Lexer et Hönninger, 2001), an individual-based model that simulates tree growth at the stand scale for specific soil and climatic conditions. Following the methodology (Boulanger et Pascual Puigdevall, 2021), we used projections of future climate data from the Canadian Earth System Model version 2 (CanESM2) and soil data from (Sylvain *et al.*, 2022) in PICUS simulations. From these simulations, we derived species-specific parameters for each climate scenarios (baseline for 2020, RCP 4.5 and RCP 8.5) and for all ecoregions of the landscape. The available climate projections only reach year 2100. Since our investigated scenarios were simulated until 2220, we assumed that climate conditions beyond 2100 remained constant.

The other parameters required by the Biomass Succession extension – such as growth curves or the impact of shade on productivity – were derived from calibration runs of LANDIS-II. The goal of these calibrations was to reduce the difference between the initial estimates of biomass by the extension, and the biomass estimates from remote sensing estimates (Beaudoin *et al.*, 2014). These biomass estimates computed by

Biomass Succession are based on the initial community structure for each cell (see section 3.2.3.1) following the methodology described in (Scheller et Mladenoff, 2004).

3.2.3.3 Base Fire

Throughout all simulations, we simulated the natural disturbance regime specific to this landscape: forest fires and spruce budworm outbreaks. The catastrophic pulse disturbances (see section 3.2.3.6) therefore occur in addition to these natural recurrent disturbances. Forest fires were simulated with the Base Fire extension (v4.0) (He et Mladenoff, 1999). This extension simulates fire ignition and propagation in the landscape based on three characteristics that together determine a fire regime: fire size, number of fires and fire severity. Different regions with specific fire regimes can be determined by the user. We defined two homogeneous fire regions in our landscape using the methodology of (Boulanger *et al.*, 2017) and the data from (Boulanger *et al.*, 2014). For each of these two fire regions, we used calibration runs in LANDIS-II to find parameters for the Base Fire extension that would replicate the correct minimum and maximum size of fires and the percentage of annual area burned. As the fire regime in each region changes with time due to the shifting climate, we obtained a set of parameters for each simulated climate scenario, and for three time periods (2020-2040, 2041-2070, 2070-beyond) according to the predictions of (Boulanger *et al.*, 2014). The fire regions in the simulated landscape presented a high variability in fire size each year. To account for this stochasticity, we simulated 30 replicates of each calibration run to obtain an average annual area burned corresponding to the existing projection for every climate scenario (Boulanger *et al.*, 2014). We only simulated forest fires with the highest severity, corresponding to crown fires, as those are the most frequent and important fires in our study area (Jayen *et al.*, 2006).

3.2.3.4 Spruce Budworm

We simulated SBW outbreaks via the Biological Disturbance Agent (BDA) extension (v4.0.1). The extension simulates new epicenters as probabilistic events in landscape cells, from which outbreaks propagate to surrounding cells. Cells are disturbed with different degrees of severity depending on their host proportion which in turn influences the probability of mortality of the different age cohorts. Host tree species for the SBW were, from most to least vulnerable, balsam fir (Abies balsamea), white spruce (Picea glauca), red spruce (Picea rubens) and black spruce (Picea mariana). While climate is predicted to alter outbreak dynamics by changing the tree species composition across the landscape (Régnière *et al.*, 2012), for simplicity we omitted any direct effects of climate change on SBW outbreaks. We parameterized the BDA extension using parameters from Boulanger *et al.*, (2018). This study derived the parameters through a calibration and validation exercise using a forest ecosystem landscape similar to the one present in our study area. This parametrization led to the simulation of SBW outbreaks with a periodicity of 40 years and a duration of 10 years (Boulanger *et al.*, 2012).

3.2.3.5 Harvesting

We developed a new harvesting extension for LANDIS-II named "Magic Harvest" that works in tandem with the existing harvest extensions of LANDIS-II (Hardy, 2022). This extension allowed implementing more complex harvest prescriptions at the stand level. Using this new extension, we implemented six different types of stand-level harvest prescriptions (Table 3-1). CC-PlantIntens consisted of a complete clearcut followed by the establishment of "intensive" plantations. These plantations combined a fast-growing hybrid tree species with a marketable tree species (e.g., black spruce) to maximize wood production. The model parameters used for characterizing the growth of these hybrid species were defined to emulate the rapid growth of hybrid poplar and hybrid larch recently developed in North America. As such, the main physiological parameters of these hybrid species (e.g., shade and fire tolerance) were similar to their non-hybrid alternatives (the trembling aspen and the tamarack; see Larocque *et al.*, 2013 ; Perron, 2011), but their maximum ANPP was doubled and their maximum biomass was increased by 15% in every ecoregion. Moreover, the longevity of these hybrid species was divided by two, leading to the faster mortality. We based these parameter changes on expert opinion and on data from the Quebec Sylvicultural Guide (Government of Quebec). CC-PlantFunct, CC-NormalPlant and CC-NoPlant, all consisted of a clearcut with protection of advanced regeneration, but in CC-PlantFunct this clearcut was followed by the establishment of functionally enriched plantations, whereas in CC-NormalPlant it was followed by tree planting of the dominant species prior to harvesting, and in CC-NoPlant it was not followed by any kind of plantations. SC consisted of a selection cutting harvesting 1/3 of the stand biomass of all age cohorts older than 30 years, every 30 years during a 90-year period. Finally, CT consisted of a commercial thinning removing 60 to 80% of the biomass of younger tree cohorts but only 5 to 40% of the biomass of older cohorts, and was repeated twice in a 50-year interval in TRIAD intensive areas. Stands restricted for harvesting with SC and CT became available to any other prescription after the 90- (SC) or the 50year (CT in intensive TRIAD areas) period. The details of each prescription are given in appendix B. All prescriptions targeted the stands with highest biomass available for harvest in the landscape or in their area of application (see below), except for CC-PlantFunct which targeted the stands with the lowest functional response diversity (see section 3.2.4). Prescriptions were applied in an arbitrary order, harvesting their biomass target one after the other until all prescriptions for the given scenario were considered.

Table 3-1 : Description of the harvest prescriptions simulated

	Cut		Planting	Repetition
	Percent biomass removed	Age of cohorts	Species planted	Timing of prescription
CC-PlantIntens	100%		Hybrid poplar or hybrid larch depending on the latitude, combined with black spruce	None
CC-PlantFunct	90%	> 10	Species from functional groups absent or rare in the stand	None
CC-NormalPlant	90%	> 10	Dominant species in the stand	None
CC-NoPlant	90%	> 10	None	None
SC	30%	>= 30	None	Every 30 years for 90 years
CT	80% 66% 60% 40% 5%	<= 30 31-50 51-90 91-100 > 120	None	• At year 20 and year 50 in TRIAD intensive zones; • No repetition in BAU.

Furthermore, in the enriched plantations of CC-PlantFunct, species were selected based on the functional groups present in the targeted stand. The functional groups in this study were identified through a clustering analysis of the 17 simulated species using nine different functional response traits related to our three catastrophic disturbances (see section 3.2.3.6): maximum height, seed dry mass, wood density, leaf nitrogen content per leaf dry mass, specific leaf area (SLA), bark thickness, fire tolerance, drought tolerance and shade tolerance (appendix D). The clustering method resulted in five functional groups: three gymnosperm groups with different tolerances to shade, drought, and fire; and two angiosperm groups with a clear distinction between pioneer and mid to late-succession species. When species from one or several functional groups were not present in the targeted stand, we planted one new age cohort of one species for each missing functional group. The selected species for each missing group was chosen randomly among those with the highest probability of establishment, which varied by ecoregion and could thus differ with time due to climate change. If all functional groups were already present in the stand, we selected a species from the group with the smallest abundance (as estimated by their biomass). This methodology ensured a local increase in functional response diversity.

The four forest management strategies harvested the same biomass target in different ways (Table 3-2). In the two types of BAU scenarios, all unprotected forests were available for harvesting by SC, CT, and CC-NormalPlant. In addition, CC-PlantFunct was available in BAU-PlantFunct and CC-NoPlant in BAU-NoPlant. In contrast, the TRIAD+ and normal TRIAD scenarios restricted the use of SC, CC-PlantFunct (in TRIAD+) and CC-NoPlant (in Normal TRIAD) to their extensive zones. In the intensive zones, forests were instead harvested with CT and CC-PlantIntens to maximize wood production using hybrid species and commercial thinning. As such, thinning (CT) in the intensive areas of TRIAD scenarios was repeated twice during the 50 year-period following a first thinning to simulate an intensive commercial thinning (Table 3-1). The intensive zones were fixed and corresponded to 16% of the forest surface with the highest Annual Net Primary Productivity (ANPP) during a calibration run made with no natural disturbances. The gain in productivity in the intensive zones allowed increasing the size of the unharvested areas. By creating a 5000m buffer around the largest current protected areas (as of 2020), conservation reached 24% of the forested area. Only the size of the 10% largest protected areas was increased. Most existing small protected areas were created by the provincial government to act as biological refugia dispersed across the landscape (Poulain, 2014 ; Riva et Fahrig, 2022). In agreement with this original goal, the size of small protected areas was not increased. Thus the TRIAD zoning ratio consisted in 16% of the forested surface dedicated to intensive management, 60% to extensive management, and 24% to conservation. This ratio is similar to the one recommended by Blattert *et al.*, (2023) in their estimation of an optimal TRIAD zoning in Finnish landscapes.

Table 3-2 : Percentage of the biomass target harvested with the different harvest prescriptions in the four forest management scenarios

	CC-PlantIntens	CC-PlantFunct	CC-NormalPlant	CC-NoPlant	SC	CT
TRIAD+++	25% [I]	37.5% [E]			12.5% [E]	25% [I]
Normal TRIAD	25% [I]			37.5% [E]	12.5% [E]	25% [I]
BAU-PlantFunct		37.5% [A]	45% [A]		12.5% [A]	5% [A]
BAU-NoPlant			45% [A]	37.5% [A]	12.5% [A]	5% [A]

[I] *: In intensive areas only*

[E] *: In extensive areas only*

[A] *: in all the landscape except protected areas*

3.2.3.6 Catastrophic disturbance events

We simulated catastrophic disturbance events in LANDIS-II using the Biomass Harvest extension where mortality is represented by the loss of biomass in impacted forest stands . While originally designed to simulate harvesting, this extension can be employed to simulate any disturbance by removing tree biomass according to different severity and spatial distribution patterns. The severity of these events at the stand scale, i.e., the amount of removed biomass of a given species age cohort, was determined based on the functional response traits of that species as well as those of the other species in the community. The severity of disturbances also varies according to small-scale factors such as topography, soil, and microclimate. For simplicity, we did not include these factors and rather focused on the influence of species composition on disturbance severity.

3.2.3.6.1 Large fire

We defined the large fire as a disturbance extending across the entire landscape but creating numerous unburned forest patches, i.e., fire refugia (Figure 3-2c). The total area covered by the refugia was fixed to 30% of the landscape, a proportion intermediate to the minimum (20%) and maximum (57%) values determined by Walker *et al.*, (2019) when measuring fire refugia from satellite imagery in coniferous and mixed forests landscapes. The size of individual refugium was sampled from a power-law distribution varying between one and 100 ha to make large refugia uncommon (Eberhart et Woodard, 1987 ; Walker *et al.*, 2019) (appendix C.3). Each refugium was created by first randomly choosing a forest stand in the landscape to be at the center of the refugia, and then increasing its size from stand to stand until the sampled size was reached. For simplicity, the stand at the center of the refugia was selected randomly since modelling the influence of fine scale factors (e.g., slope, topographic wetness, etc.) on the creation of refugia was beyond the scope of our study (Krawchuk *et al.*, 2016). Refugia were added one by one in the landscape until 30% of the landscape's surface was reached.

Table 3-3 : Proportion of biomass loss for each catastrophic event at the species and stand level. The total proportion of biomass loss for a given species in a given stand is computed by multiplying both species age cohort and stand level effects.

Catastrophic event	Age-cohort level		Stand level	
	Factor influencing biomass loss	**Biomass loss (%)**	**Factor influencing the reduction of biomass loss**	**Reduction of biomass loss (%)**
Large fire	**Species fire tolerance**		**Stand CMW of fire tolerance**	
	0-1	100	0-1	0
	1-2	90	1-2	10
	2-3	80	2-3	20
	3-4	70	3-4	30
	4-5	60	4-5	40
Severe drought	**Species drought tolerance**		**Stand functional diversity**	
	0-1	70	0-1	0
	1-2	60	1-2	5
	2-3	50	2-3	10
	3-4	40	3-4	15
	4-5	30	4-5	20
MPB epidemic	**Host status**		**Stand host abundance (%)**	
			100-90	0
	Pinus strobus	80	90-80	8.6
			80-70	17.2
	Pinus resinosa	80	70-60	25.8
			60-50	34.4
	Pinus banksiana	80	50-40	43
			40-30	51.6
	Non-host species	0	30-20	60.2
			20-10	68.8
			10-0	77.4

Within burned stands, we assumed that tree biomass was consumed by fire according to a species-level process and a stand-level process. At the species level, we used the fire tolerance trait to determine the proportion of biomass loss of an age cohort. This proportion varied from 60% loss at high tolerance (i.e. trait value between 4 and 5) to 100% at low tolerance (trait value less than 1) (He et Mladenoff, 1999)

(Table 3-3). At the stand level, we computed the Community Weighted Mean (CWM) of the fire tolerance trait over all species present (Table 3-3 and Figure 3-2c). This stand-level tolerance measure served to determine a protection effect from the community which in turn modulated the loss of biomass of each age cohort within the stand (based on Moris *et al.*, 2022). The protection effect varied from 0 (no protection), when the stand CMW fire tolerance was less than 1, to 40% when it was highest (value between 4 and 5) (Table 3-3). The resulting biomass loss of an age cohort due to the large fire was calculated by multiplying the stand-level protection effect with the species-level proportion of biomass loss. These species and stand-level processes were parameterized based on our expert knowledge of forest fires and fire tolerance since the existing literature could not provide direct parameter values (Table 3-3).

Note that whereas biomass loss was a function of fire tolerance during this single catastrophic fire, periodic fires generated by the Base Fire extension consumed all biomass irrespective of species tolerance to fire. Indeed, the Base Fire extension was used to simulate smaller but more intense fires, whereas the Biomass Harvest extension was used to simulate an extremely large and long fire event varying in intensity according to forest composition. In addition, following the large fire, we did not simulate the regeneration of serotinous species, as their fire tolerance implied that their age cohorts would never entirely disappear from a burned cell.

3.2.3.6.2 Severe drought

In contrast to the refugia created during the large fire disturbance event, the severe drought affected the entire landscape. The loss of biomass resulting from drought mortality was also determined through a species-level and a stand-level process. For each species' age cohort, biomass loss depended on the species' drought tolerance trait (Paquette et Messier, 2011). At the stand level, higher values of functional response diversity (see section 3.2.4 for its measurement) increase the drought tolerance for all species present. This stand-level effect was based on studies suggesting that functional response diversity was more important than species diversity in improving drought tolerance through resource partitioning and facilitation (appendix C.3; Fichtner *et al.*, 2020 ; Grossiord, 2020). As for the large fire, we parameterized this process based on expert estimation of the biomass lost for species of different drought tolerance and according to the at the stand functional diversity.

3.2.3.6.3 Mountain Pine Beatle outbreak

The MPB outbreak affected all forest stands containing any of the potential host species: the white pine (Pinus strobus), the red pine (Pinus resinosa), and the jack pine (Pinus banksiana) species (Table 3-3). We assumed that age cohorts of these species lost 80% of their biomass. We based this proportion on the study of Long et Lawrence (2016) which reported a pine tree mortality exceeding 80% in MPB-infected landscapes of western Montana (USA). At the stand level, we hypothesized that a dilution effect from the presence of non-host species would reduce host mortality caused by MPB (Civitello *et al.*, 2015). Therefore, we assigned a protection effect that increase as a function of the abundance of non-host species present in the stand. We used the study of Jactel *et al.* (2021) measuring the effect of stand diversity on damages caused by borer insects to estimate this stand-level protection effect (see appendix C.3, Table 3-3). Again, the resulting loss in biomass for a given age cohort was calculated by multiplying the proportion of biomass loss at the species-level with the protection effect at the stand-level.

3.2.4 Data analysis

In each scenario, we measured two variables associated with forest ecosystem functions. Firstly, we measured the biomass of mature cohorts (defined as being 40 years or older) of all species present in each stand of the landscape. The mature biomass is an important proxy for several forest functions like carbon storage and seed production, as mature trees can reproduce and create seeds while storing more carbon than smaller trees (Birdsey *et al.*, 2023 ; Hoover et Smith, 2023). In addition, mature trees are importance to the forest industry given the higher market value of large diameter trees. The total mature biomass in the landscape (B_L) was calculated by summing the stand-scale mature biomass (B_S) across all stands. Secondly, we measured the functional response diversity in each stand (FD_S) using the exponent of the Shannon Diversity index, applied to the relative biomass abundance of each functional group in the stand:

$$\mathrm{FD}_s = exp\left(-\sum_{i=1}^{n} p_i \cdot log(p_i)\right) \qquad \text{Equation 3-1}$$

where p_i is the relative abundance of functional group *i* (expressed by the biomass of its age cohorts) from the *n* functional groups present in stand *S*. FD_S measures the effective number of functional groups in a stand, quantifying its functional response diversity in a simple yet meaningful way (Jost, 2006 ; Mina *et al.*, 2022). Furthermore, we computed the mean functional response diversity across stands of the landscape

($\overline{FD}_S$) as a stand-area weighted mean of FD_S. We used $\overline{FD}_S$ to observe if functional planting did effectively increase the functional response diversity of stands at the landscape scale, and if this increase could further be linked to changes in our resilience measures (see below). In addition, we computed the total biomass of all age cohorts in the landscape for each functional group (referred as B_{FG}). The three resulting measures (B_L, $\overline{FD}_S$ and B_{FG}) were subsequently averaged across the five simulations replicates for each scenario combination.

Moreover, we measured the resilience of the mature biomass at the landscape scale (B_L) following one of the catastrophic events at t = 100., Resilience is a notably complex concept to capture that has led to the development of varied measures (e.g., speed of recovery, critical slowing down, etc.) (Folke, 2016). In the context of this study, we define resilience as the ability of a system to maintain essential functions in the face of a disturbance (Seidl *et al.*, 2016). In particular, we followed the methodology of Cantarello *et al.*, (2017) and measured the resilience of B_L through three different metrics (Figure 3-3): resistance (R), net change (NC) and rate of recovery (RR). As noted above, the mature biomass acts here as a proxy for several important ecosystem functions such as seed production, carbon storage, and wood production.

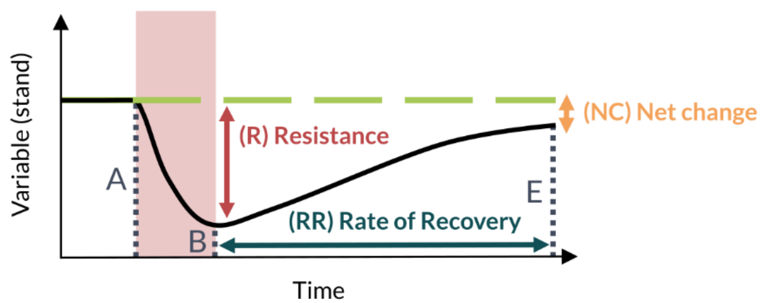

Figure 3-3 : Measures of resilience used in our study. A, B and E refers to the value of the variable of interest before the disturbance (A), right after it (B), and at the end of the simulation (E), and are used in equations 2 and 3.

R was defined as the variation between the value of the variable immediately before (B, at t = 90) and after (A, at t =100) the catastrophic using the following equation:

$$R = 1 - \frac{2B}{A + B} \qquad \text{Equation 3-2}$$

Hence, R varied between 1 (no change in the value of Bs) and 0 (maximum change) (Figure 3-3a). In contrast, NC corresponded to the percentage difference between the value of the variable at the end of the simulation (E, at t = 200) and its value before the catastrophic event (B, at t = 90), relative to its value before the event (Figure 3-3b, Equation 3-2).

$$NC = \frac{E - B}{B}$$ Équation 3-3

NC was therefore negative if E at t = 200 was lower than its pre-disturbance value A and positive if it exceeded A. Lastly, RR corresponded to the inverse of the recovery time (in years) that BS took to reach its pre-catastrophic event value, which can also be interpreted as the percentage of mature biomass recovered every year (Figure 3-3c). If the variable did not retrieve its pre-disturbance value before the end of the simulation, RR was set at 0.01, corresponding to the inverse of the maximum recovery time possible in our simulations (100 years). As such, an increase in each of these three measures represented an increase in the resilience of the mature biomass in the landscape.

We expected R, NC and RR to complement each other since they measure distinct aspects of resilience. Indeed, R measures the magnitude of the initial impact of the catastrophic event, but does not consider the temporal dynamic of BS following the event. In contrast, NC and RR are both influenced by the legs of the disturbance event and other sources of mortality (e.g. regular fires) that may occur during the 100-year period of recovery. But while NC revealed how well a stand had recovered during that period, RR showed how fast this recovery was.

3.3 Results

We present the temporal dynamic of the total mature biomass (B_L) and the mean functional response diversity of the forest stands ($\overline{FD}_S$) for each combination of climate scenario, catastrophic disturbance event, and forest management strategy. We also show the total biomass of each functional group (B_{FG}) for scenarios without a catastrophic disturbance event. Additionally, we use bar plots to display how each resilience measure for the mature biomass in the landscape (B_L) (resistance R, net change NC and rate of recovery RR) varies across scenarios and replicates following a catastrophe.

3.3.1 Temporal dynamics of mature biomass and mean functional response diversity

Our results show that the temporal dynamic of B_L differed little between the four forest management strategies implemented (Figure 3-4). In most catastrophe and climate scenarios, the BAU-PlantFunct management strategy tended to increase B_L slightly (by up to 8%) compared the other strategies (Figure

3-4, yellow curves). In contrast, the normal TRIAD strategy tended to produce smaller B_L (Figure 3-4, red curves). Furthermore, the variation in B_L between replicates was negligible for each scenario, indicating that the stochasticity of the forest dynamics had little effect when considering the mature biomass at the landscape scale.

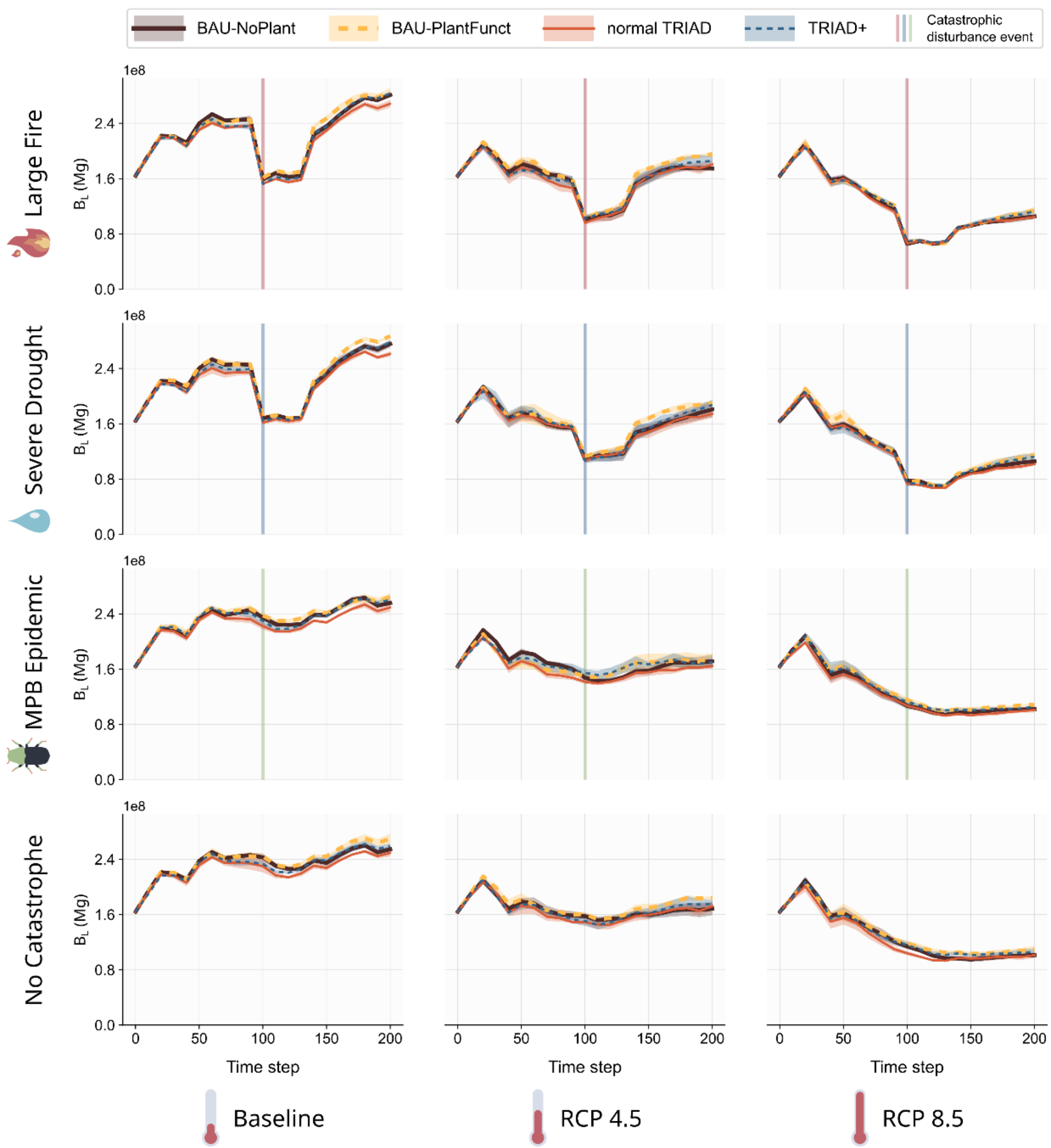

Figure 3-4 : Temporal variation of the Total Mature Biomass in the landscape B_L for each combination of management, climate and catastrophe scenario. Solid lines are mean values and envelops are standard deviation across 5 simulation replicates

The main differences in B_L were observed between the catastrophic disturbance scenarios and between the climate scenarios (Figure 3-4). Indeed, the large fire and severe drought catastrophes resulted in important reductions in B_L at t = 100 (around 30%), impacting its dynamic for the following decades (Figure 3-4, first and second row). However, B_L ultimately recovered in both cases after 50 years or so and reached values similar to the "no catastrophe" scenarios during the last 50 years of the simulations. In contrast, the MPB outbreak had almost no impact on B_L when compared to the "no catastrophe" scenario (Figure 3-4, third row). The most important factor impacting the dynamic of B_L was climate. Indeed, B_L either increased in the baseline climate scenario (Figure 3-4, left column), remained relatively stable in the RCP 4.5 scenario (Figure 3-4, middle column) or decreased in the RCP 8.5 scenario (Figure 3-4, right column) throughout the 200 years of the simulations. These trends remained regardless of the forest management strategy involved or nature of the simulated catastrophe.

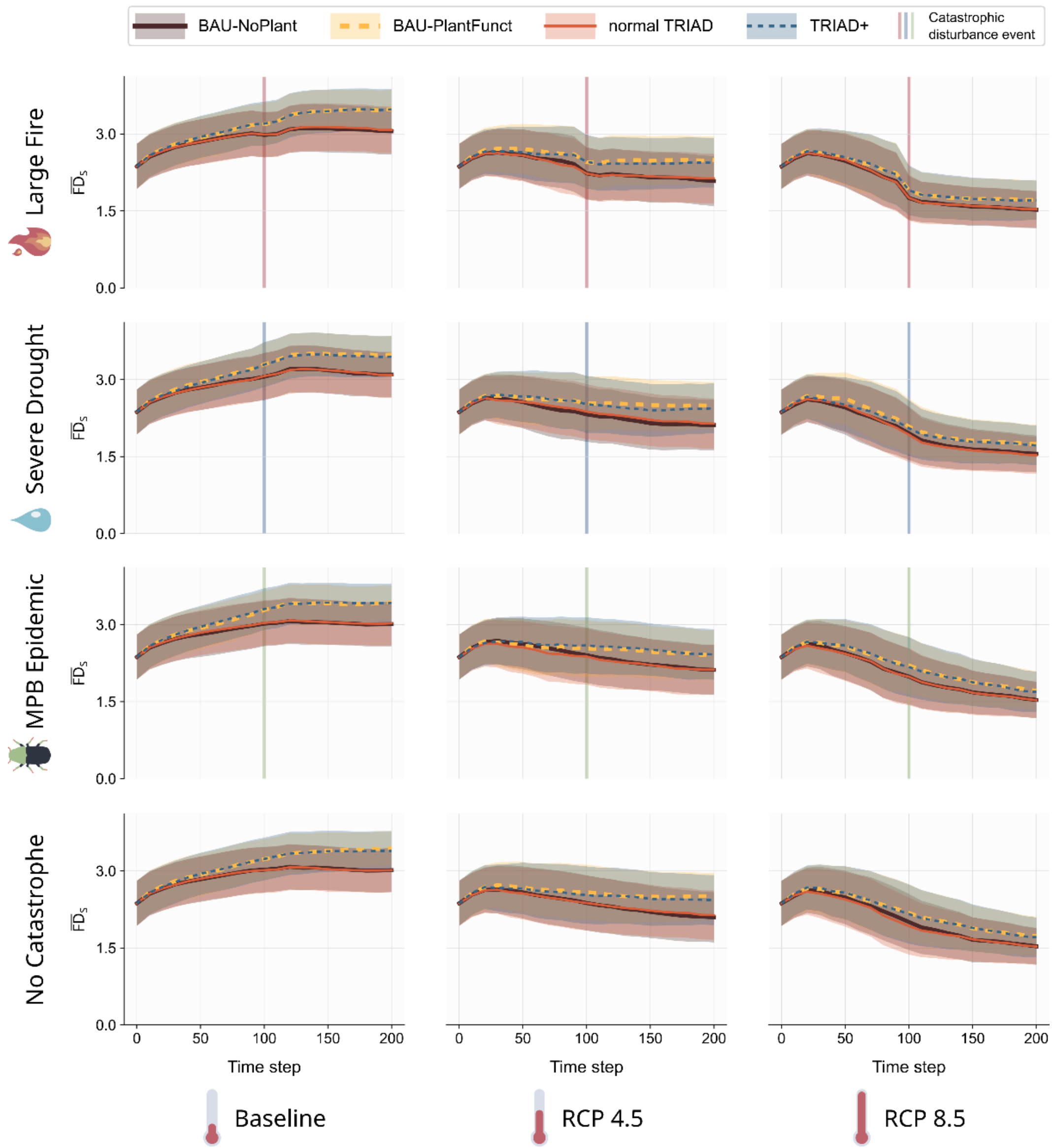

Figure 3-5 : Temporal variation of the mean Functional Diversity across all stands of the landscape ($\overline{FD}_S$) for each combination of management, climate and catastrophe scenario. Solid lines are mean values and envelops are standard deviation across stands and 5 simulation replicates.

In contrast, the temporal dynamics of the mean functional diversity, $\overline{FD}_S$, varied according to the different forest management strategies (Figure 3-5). Indeed, regardless of the climate and catastrophe scenario, $\overline{FD}_S$ increased by up to 15% in management scenarios with functional planting (TRIAD+ and BAU-PlantFunct, Figure 3-5, blue and yellow curves) compared to scenarios without functional planting (normal TRIAD and

BAU-NoPlant, Figure 3-5, red and brown curves). This boost in $\overline{FD}_S$ increased with time but seemed to plateau towards the end of the simulations. However, the wide variability envelopes indicate important variations in functional response diversity between stands and between replicates (Figure 3-5).

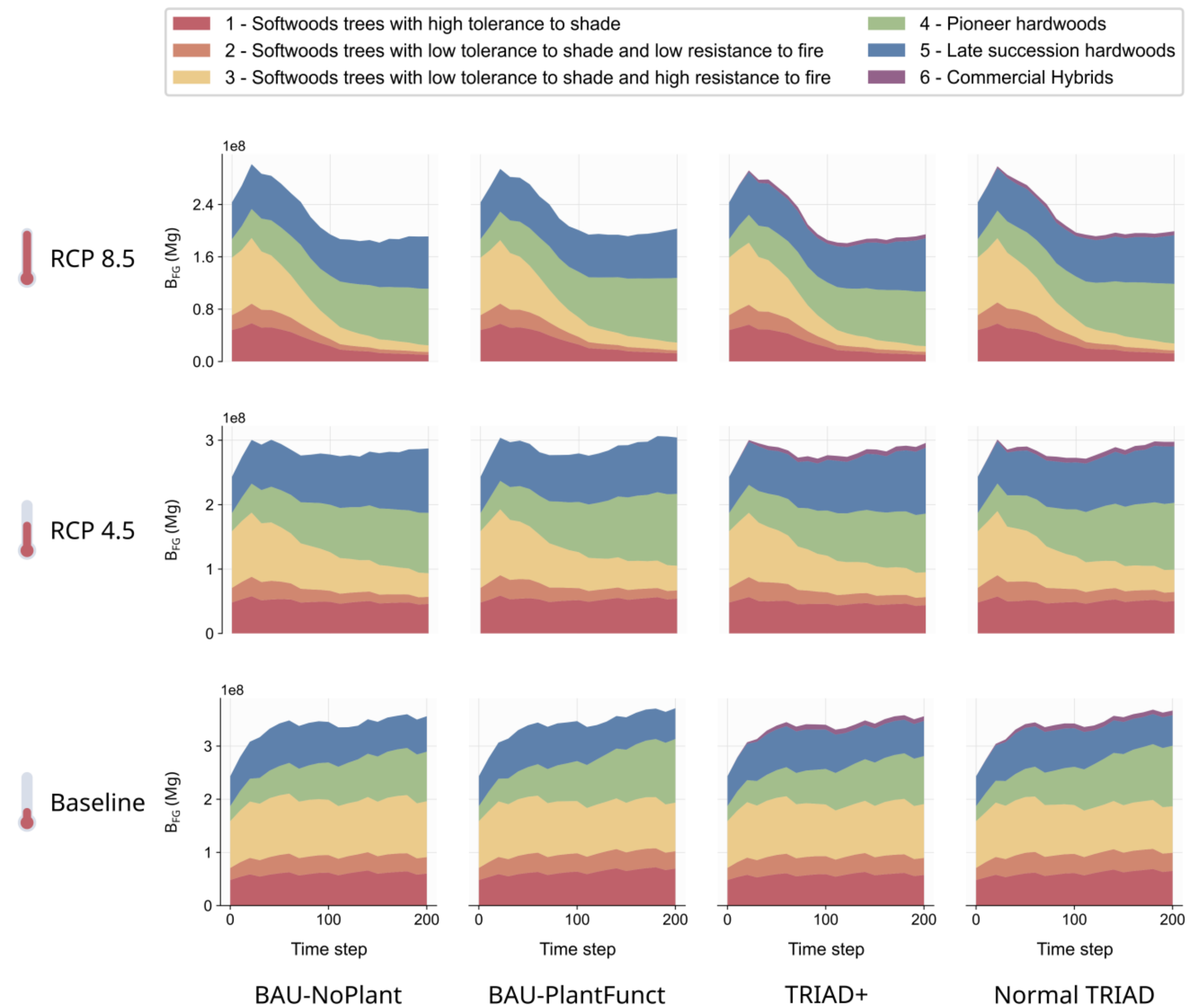

Figure 3-6 : Evolution of the total biomass for the six functional groups of trees defined in our study for each combination of climate and management scenario, but without a catastrophic disturbance event at t = 100. Values at each time step are mean values across 5 simulation replicates.

The presence of catastrophes had a small effect on the temporal dynamics of $\overline{FD}_S$, contrasting with the results for the total mature biomass (Figure 3-5, first to last row). However, climate once again played an important role in shaping the long-term trend in $\overline{FD}_S$. Indeed, similarly to B_L, $\overline{FD}_S$ increased in the baseline scenario (Figure 5, left column), decreased slightly in the RCP 4.5 scenario (Figure 3-5, middle column), and decreased more steeply in the RCP 8.5 scenario (Figure 3-5, right column) throughout the simulations.

Finally, the dynamic of total biomass of each functional group (B_{FG}) showed very little variation between management scenarios (Figure 3-6). TRIAD+ and BAU-PlantFunct scenarios show a very slight increase in the biomass of the rarest functional group in the landscape (group 2, softwoods with low tolerance to shade and fire) compared to the normal TRIAD and BAU-NoPlant scenarios, with a baseline and RCP 4.5 climate (Figure 3-6, orange areas, first and second row). Overall, the evolution of the biomass of each functional group thus changed according to the climate, but not the management strategy used.

3.3.2 Resilience of mature biomass

Our results show that all resilience measures for B_L were again more sensitive to the simulated climate and catastrophic disturbance event than to the forest management strategy (Figure 3-7). Nonetheless, in most cases, the mean value of R, NC and RR were higher for the BAU-PlantFunct and TRIAD+ scenarios, pointing to a slightly higher resilience when functional planting was used (Figure 3-7, yellow and blue bars).

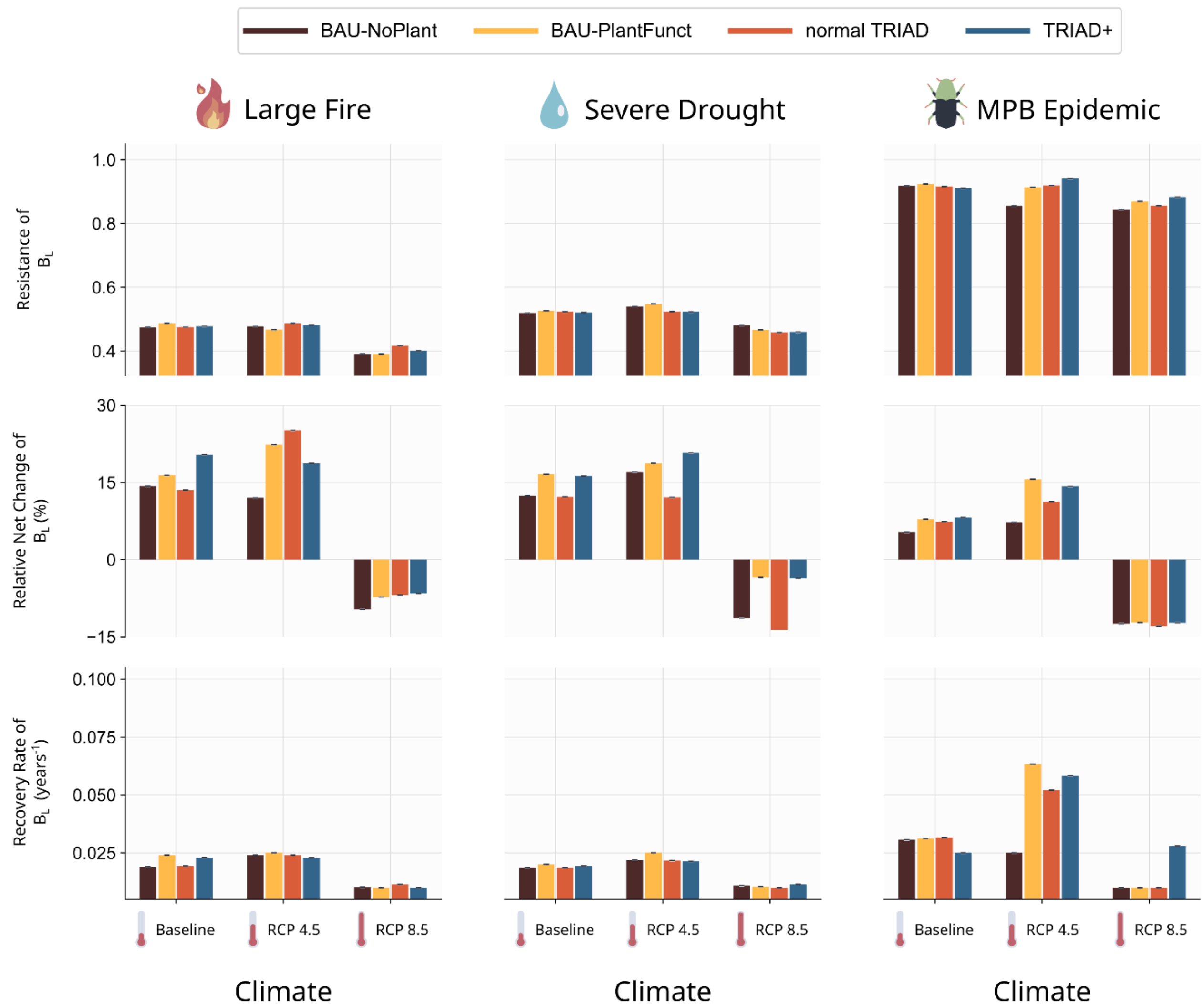

Figure 3-7 Bart plots showing the resilience values of the total biomass of the landscape (BL) for each resilience measure (row) and each catastrophic event (column). The height of the bar are mean values and error lines are standard deviation across 5 simulation replicates.

The resilience measures for B_L varied according to the type of catastrophic disturbance event simulated. The large fire resulted in the smallest values of resistance, R, followed by the severe drought and the MPB outbreak (Figure 3-7, first row). But while the large fire and severe drought resulted in lower values of R than the MPB outbreak (Figure 3-7, first row), the latter was associated with lower values of NC (Figure 3-7, middle row). This means that whereas B_L was less impacted by the MPB outbreak (leading to higher R values), the landscape did not recover its pre-disturbance B_L as well as with the other catastrophes. On the other hand, the rate of recovery, RR, was slightly higher for the MPB outbreak scenario than for the other two catastrophes, implying a faster recovery of B_L following this outbreak (Figure 3-7, last row).

Overall, the most coherent signal throughout all resilience measures came from the climate scenarios. Scenarios with the most intense level of climate change (RCP 8.5) was systematically associated with low R, NC and RR values, indicating an overall loss of mature biomass resilience at the landscape scale (Figure 3-7). However, the RCP 4.5 scenario led to resilience values similar or higher than for the Baseline scenario for all three measures and all three catastrophes.

3.4 Discussion

Our study aimed at investigating how new forest management strategies could help prepare forests to an uncertain future by increasing their resilience to different extreme disturbance events. In this perspective, we designed a new type of TRIAD zoning, "TRIAD+", that included functional enriched plantations, with the goal of achieving a good compromise between conservation, production and adaptation in a forest landscape. Our results indicate that the TRIAD+ strategy improved the functional response diversity of forests compared to other management strategies, while increasing the size of protected areas and harvesting the same amount of wood. Our results also show that TRIAD+ increased the resilience of mature biomass in the landscape following different severe disturbances. This could in turn help sustain several important forest ecosystem functions (e.g. seed production, carbon storage, etc.) under future unpredictable extreme events. However, our results also highlight important issues in the practical aspects of functional enrichment, the difficulty of trying to prepare forests to multiple possible extreme events as well as climate changes, and the limitations of landscape-scale models like LANDIS-II to model the effects of functional response diversity.

3.4.1 Climate as the main driver of the dynamic and resilience of the landscape

Among the three factors that we varied in our scenarios, climate stood out as having the largest effect throughout our simulations (see section 3.3). This is particularly visible when comparing the temporal dynamics of the total mature biomass in the landscape (B_L) in scenarios with or without a catastrophe (Figure 3-4 and Figure 3-6). In the scenarios with the large fire and the severe drought, B_L dropped drastically at t = 100 but rapidly (within 50 years) recovered to similar values found in the corresponding scenarios without catastrophes. In addition, the scenarios without any catastrophe clearly illustrate that B_L decreased with increasing intensity in climate change (Figure 3-4, bottom row). Taken together, these observations suggest that climate conditions were the strongest drivers determining the dynamics of B_L, and not the simulated management strategies nor the punctual disturbances.

Our resilience measures also captured the impact of climate on species' growth. Indeed, scenarios with more intense climate change (i.e., RCP 8.5) presented lower resilience values compared to the baseline and RCP 4.5 scenario (Figure 3-7). In the RCP 4.5 scenarios, climate change increased the number of fires (see appendix C.5) and reduced the growth capabilities of certain species compared to the baseline; but it also improved the growth of other (thermophilic) species, reducing the differences with baseline scenarios. In contrast, in the RCP 8.5 scenarios, climate increased fire activity and was associated with reduced growth parameters. Our results thus reflect the reported uncertainty regarding the general effect of climate change on forest growth since it may be beneficial through certain processes (e.g., longer growing seasons, CO2 fertilization, etc.; not all considered in this study), but detrimental through others (e.g., more frequent natural disturbances, more hydric stress, etc.) (Kurz *et al.*, 2008 ; Reyer *et al.*, 2017).

Besides changes in B_L, climate change was also responsible for important shifts in B_{FG} (Figure 3-6). Specifically, the total biomass of functional groups of hardwood species like red oak, trembling aspen, or red and sugar maple strongly increased throughout all simulations, but even more under the RCP 4.5 and 8.5 climate scenarios (Figure 3-6, blue and green areas). On the other hand, the total biomass of functional groups of softwoods species like black spruce, white spruce or balsam fir decreased markedly with time, especially under the RCP 4.5 and 8.5 climate scenarios (Figure 3-6, yellow, orange and red areas). This decrease in softwood biomass resulted in the collapse of the current main economic tree species in Quebec, echoing previous concerns in the literature as to the sustainability of Canada's forestry sector under climate change (Brecka *et al.*, 2020).

3.4.2 Species growth limiting the effect of functional enrichment on functional response diversity

Functional planting, implemented in the TRIAD+ and BAU-PlantFunct scenarios, led to an increase in the mean functional diversity ($\overline{FD}_S$) over time compared to the BAU and TRIAD scenarios (Figure 3-5). However, this increase plateaued with time – a trend that can be observed for both management scenarios in the temporal evolution of $\overline{FD}_S$ (Figure 3-5) and of B_{FG}, the total biomass of functional groups (Figure 3-6). This saturation suggests that fine scale mechanisms, such as species growth and competition, may limit the ability of functional enrichment to diversify this landscape in the long term, especially in the context of climate change. In our model, we implemented functional enrichment by systematically planting a single age cohort of tree species from rare or missing functional groups at the stand scale. This planting followed a clear cut that kept 10% of the stand biomass, hence preserving species from the original functional groups. As such, once planted, rarer functional groups would often be unable to compete with the original species that were thriving under local climate conditions, even though the planted species were the most adapted to the given ecoregion. As such, these original species would rapidly outgrow the newly introduced ones. Given that we used a measure of functional diversity weighted by the abundance of functional groups, the increase in $\overline{FD}_S$ was thus saturated by the limited performance of species from additional functional groups.

3.4.3 Low effect of the functional response diversity on stand resilience

The relatively small effect of our management strategies on the resilience of the mature biomass in the landscape (B_L; Figure 3-7) can be interpreted in two ways. On the one hand, our implementation of functional planting might have not increased the functional response diversity of stands enough to influence their resilience sufficiently (see previous section). On the other hand, the effect of functional response diversity on stand resilience might be small to begin with. Indeed, several empirical studies have observed that the effect of functional response diversity on forest resilience was either absent, small, or highly contextual (Espelta *et al.*, 2020 ; Grossiord *et al.*, 2014 ; Ross et Sasaki, 2023 ; Spasojevic *et al.*, 2016). More generally, Yang *et al.*, (2018) argue that functional traits can be poor predictors of tree demographic when the focus is on species rather than individuals, when there is missing contextual information about the trait values (e.g., biogeographic, phenotypic), or when functions important to tree demographics are not being captured by the most measured traits.

Contrasting with these empirical studies, simulations of tropical forest dynamics using stand-scale individual-based models revealed a positive effect of functional response diversity on forest resilience (Sakschewski *et al.*, 2016 ; Schmitt *et al.*, 2020). However, these effects were conditional on how vegetation dynamics was represented in the models. In particular, Schmitt *et al.* (2020) indicated that functional response diversity improved stand resilience to disturbances through complementarity mechanisms between species with different traits (niche partitioning and facilitation). They also noted that the effect of functional response diversity was only temporary as complementarity effects quickly gave way to interspecific competition. Therefore, the effect of functional response diversity on forest resilience may depend on the studied disturbance, the local environmental conditions (slope, etc.) or the forest age. It is also possible that individual-based models are better suited to capture the effect of functional response diversity compared with models, such as LANDIS-II, that are based on age-cohorts dynamics (see section 3.4.5 on model limitations).

In addition, functional enrichment planting with the wide goal of increasing functional response diversity may sometimes be less effective than other forms of planting to increase forest resilience. For example, several studies have found that planting species adapted to specific disturbances or climate conditions could increase the resilience of forest landscapes following these disturbances (Buma et Wessman, 2013 ; Lucash *et al.*, 2017). We argue that this reflects a trade-off between protecting against a broad range of uncertain future conditions and protecting against specific disturbances or aspects of climate change. While the broader strategy might prove less effective when the threats are well known and defined, it might be more effective when facing the unexpected threats that await forests.

3.4.4 Is TRIAD+ a good compromise?

An important goal of our study was to assess whether TRIAD+ could produce a good trade-off between conservation, production and adaptation in the landscape. At first glance, our results might indicate that TRIAD+ was slightly less effective than the BAU-PlantFunct strategy, as the latter resulted in slightly higher values of B_L, $\overline{FD}_S$, R, NC and RR in several cases. Specifically, when compared with values for the TRIAD+, the BAU-PlantFunct scenario led to slightly higher values of B_L (Figure 3-4), similar values of $\overline{FD}_S$ (Figure 3-5), and an equal or slightly improved resilience for the mature biomass in the landscape (Figure 3-7). However, the performance of the BAU-PlantFunct scenario must be contrasted with the fact that TRIADs scenarios contained more than twice the quantity of protected areas (reserves) than the BAU scenarios. Indeed, the TRIAD+ scenario had 24% of its forested surface defined as protected areas compared to 9% for the BAU-PlantFunct scenario, and still resulted in similar values of B_L, $\overline{FD}_S$ and resilience. Although the tested management scenarios mostly produced small differences in mature biomass, functional diversity, and resilience (appendix C.6), TRIAD+ can be considered as an appealing avenue to satisfy multiple management goals including resilience to future changes.

3.4.5 Limitations

Our results also suggest some limitations in our modelling methodology. Firstly, the strong effect of climate on the dynamics of B_L, $\overline{FD}_S$ and resilience could be explained by how succession dynamics and biomass accumulation are represented in the LANDIS-II Biomass Succession extension we employed. In this extension, each tree species is associated with a maximum annual net primary productivity (ANPP) and a maximum biomass per age cohort, that depend on ecoregion and climate (Scheller et Miranda, 2021). Thus, climate acts on the mature biomass of age cohorts both as a "hard ceiling" via the maximum biomass parameter, and as an "escalator" via the maximum ANPP. Therefore, following any disturbance events, the mature biomass of remaining age cohorts could quickly recovered. As such, the strong effect of climate reduced the potential variations in mature biomass caused by other factors, such as forest management and catastrophic events.

Secondly, the Biomass Succession extension may also be responsible for the relatively poor growth of species from rare functional groups. Indeed, the main drivers of forest dynamics in this extension are relatively simple spatially implicit competition for shade and growth, with growth being mostly influenced by climatic conditions in our study area. More complex succession extensions of LANDIS-II, such as the PnET succession extension (Gustafson *et al.*, 2023), might therefore have yielded different results by dynamically simulating other mechanisms that influence species' performance at the local scale such as water availability, spatially explicit self-thinning, nutrient competition, complementarity effects, etc.

3.4.6 Guidelines and considerations for future studies

Our study highlights important considerations on the potential effects of forestry and climate change on future forest ecosystems which can serve as guiding principles in investigating novel resilience-based management strategies. Firstly, the strong effect of climate on the dynamics of mature biomass and functional diversity, compared to the four tested management strategies, suggests that human efforts to shape future forests composition and resilience under climate change will need to be considerable. It will require frequent interventions carried across large scales to successfully guide and maintain forest ecosystems into resilient states.

Secondly, our results emphasize the challenges surrounding the practice of functional planting. In particular, species selected for functional plantations may not always be able to thrive in the long-term. In our case, our functional planting strategy was to select species from rare functional groups. However, even if these new species have functional traits well adapted to future conditions they may be poor competitor in these novel communities. In such situations, frequent interventions in stands (e.g., via commercial thinning) across a large spatial scale might be required to balance the relative abundance of different functional groups through time. This implies that sustaining the effects of functional enrichment may be a complex and costly operation in certain forests. Other planting strategies focused on specific disturbances or on species expected to “win” under climate change could be considered, although the uncertainty of future forest disturbances may limit the success of such strategies.

Thirdly, the contrast between our findings and those of other studies suggest that the simulation of fine-scale processes of forest dynamics should be improved in forest landscape models such as LANDIS-II. Indeed, the intra-cell dynamics in LANDIS-II remains spatially-implicit and does not simulate some small-scale processes such as nutrient and water competition or complementarity effects. As such, in its current form, the ability of LANDIS-II to capture the regeneration dynamics of forest stands following silvicultural intervention is limited.

Fourthly, the collapse in the abundance of several commercial tree species suggests that the wood-building industry in tandem with the forest industry, will have no choice but to adapt their practices to stay viable in the future. Both industries will have to harvest and use trees of a more diverse set of species than what is currently the norm (Osborne *et al.*, 2023).

Fifthly, we assumed that catastrophic disturbances affected all age cohorts equally. However, natural disturbances are known to affect trees of different ages in different ways. For example, recent studies suggest that older and younger trees can differ in their resistance and resilience to drought, due in part to their root system (Au *et al.*, 2022 ; Bennett *et al.*, 2015). Fire resistance is also expected to increase with age due to bark thickness (Moris *et al.*, 2022). In addition, tree age can be an important factor in predicting tree mortality from some insect outbreaks, such as the spruce budworm which causes increased mortality to mature trees (Houndode *et al.*, 2021). Should we have taken these nuances into account, it is possible that our catastrophic events might have influenced forest resilience differently, by favoring either old or younger tree survival depending on the event.

Finally, our study echoes concern that developing multifunctional forest management strategies is no longer a sufficient goal, as shown by the very similar performance of the classic TRIAD and BAU scenarios regarding forest resilience. But management decisions now can shape tomorrow’s forests, and therefore must include objectives of adaptation. As the climate warms and humanity’s population and resources consumption keep increasing, the future of forests worldwide becomes more and more uncertain. It is therefore our responsibility to find ways to help forests adapt to a world where everyone – from humans to trees – is now facing the unknown of the Anthropocene.

3.5 Acknowledgements

We thank Hervé Jactel for his help with the effect size of stand mixture on insect damage.

3.6 Competing interests

The authors have no competing interests to declare that are relevant to the content of this article.

3.7 Author contributions

Clément Hardy: Conceptualization, Methodology, Software, Validation, Formal Analysis, Investigation, Data Curation, Writing – Original Draft, Visualization; Christian Messier: Conceptualization, Methodology, Resources, Writing – Review & Editing, Supervision, Funding Acquisition; Yan Boulanger: Software, Data Curation, Writing – Review & Editing; Dominic Cyr: Software, Data Curation, Writing – Review & Editing; Elise Filotas: Conceptualization, Methodology, Investigation, Resources, Writing – Review & Editing, Supervision, Project Administration, Funding Acquisition.

CONCLUSION

4.1 Principales contributions de la thèse

Cette thèse avait pour objectif d'explorer comment l'aménagement forestier pourrait, dans le futur, satisfaire certaines des nombreuses missions qui lui sont maintenant associées. En particulier, je me suis intéressé à l'impact des stratégies de *land-sparing* et *land-sharing* tout en prenant en compte la construction de chemins forestiers nécessaires à l'aménagement. Je me suis également penché sur le développement d'une nouvelle stratégie d'aménagement à l'échelle du paysage, la TRIAD+, ayant pour but de satisfaire à la fois des objectifs de conservation, de production et d'adaptation en lien avec les forêts. Mes travaux ont ainsi permis d'apporter différentes réponses aux questions surlignées durant la succès d'introduction.

En premier lieu, les travaux de mon premier chapitre montrent le développement d'un algorithme permettant de simuler la construction des chemins forestiers à de larges échelles spatiales et temporelles. Cet algorithme est maintenant implémenté au sein du modèle LANDIS-II comme un module gratuit et dont le code est ouvert. Le test de cet algorithme a révélé qu'il était capable de répliquer différentes caractéristiques de deux réseaux routiers forestiers existant avec une bonne précision. La densité de routes fut ainsi répliquée avec un écart de moins de 2 % dans les deux paysages étudiés (Figure 1-6). De plus, les taux de superposition des chemins réels et simulés dans une distance tampon de 500 m y ont atteint 97 et 89 % respectivement (Table 1-2). Bien que le module présente certaines limitations (voir sections suivantes), il prouve néanmoins que la simulation de chemins forestiers à l'échelle du paysage est possible au sein de Forest Landscape Models (FLMs). L'algorithme utilisé montre également que la simulation de réseaux routiers forestiers ne demande pas forcément de simuler un réseau routier totalement optimisé en termes de coûts. Ce genre d'approche est utilisée par d'autres modèles de chemins forestiers fonctionnant à des échelles spatiales plus basses (p. ex. PLANEX; voir Epstein *et al.*, 2006). Bien que ces approches soient utiles pour des problèmes opérationnels d'aménagement forestier (p. ex., «comment construire des routes pour accéder à ces forêts ?»), il est discutable qu'elles soient pertinentes pour explorer les effets de l'aménagement forestier à l'échelle du paysage (p. ex., «comment les routes résultantes d'une stratégie d'aménagement donnée vont-elles fragmenter les forêts de la zone ?»). Les réseaux routiers réels sont en effet rarement le fruit d'une réflexion à long terme cherchant à créer un réseau totalement optimisé dans l'espace. Dans la réalité, ils sont ainsi plus souvent le résultat d'une construction séquentielle de segments ajoutés pour atteindre de nouveaux endroits (Strano *et al.*, 2012). En répliquant cette construction séquentielle de segments, l'algorithme utilisé a pu réduire les temps de calcul tout en permettant une simulation relativement réaliste à de grandes échelles spatiales et temporelles. Néanmoins, des résultats récents pourraient remettre en question l'idée que les réseaux de chemins forestiers ne sont pas totalement optimisés dans l'espace (voir section suivante).

En second lieu, mes travaux du second chapitre ont exploré la variation de la proportion de coupes équiennes ou inéquiennes à l'échelle du paysage, résultant globalement en des stratégies de *sparing* ou de *sharing*. Mes résultats ont démontré que ces variations ont génèrent des compromis importants au sein des paysages simulés entre la quantité de forêts plus âgées et la fragmentation de ces forêts par les chemins forestiers. La récolte de 50 % d'une même cible de volume de bois avec des méthodes inéquiennes a ainsi multiplié la densité de routes dans le paysage par un facteur de deux (Figure 2-4). En conséquence, cette augmentation de la densité de routes tendait à fragmenter les forêts plus âgées du paysage, bien que les méthodes inéquiennes tendaient aussi à augmenter leur quantité. À ma connaissance, ces travaux représentent l'une des premières investigations des chemins forestiers générés par des stratégies d'aménagement différentes à l'échelle du paysage. Elle valide ainsi les deux hypothèses proposées : le *land-sharing* en foresterie (incarné ici par l'aménagement inéquien) protège plus de forêts âgées ou structurellement complexes, mais nécessite aussi plus de chemins forestiers. Bien que ces contributions puissent paraitre triviales de premier abord – plus de surface récoltée demande plus de chemins forestiers –, ces travaux ont révélé des considérations importantes. La première provient de l'interaction avec les feux de forêt, qui ont énormément réduit les différences observées entre scénarios en altérant la quantité et la fragmentation des vieilles forêts du paysage en dépit de tout aménagement forestier. Cette découverte implique que les compromis observés entre le *land-sharing* et *land-sparing* varient certainement en fonction des perturbations affectant les forêts. La deuxième considération provient de l'effet de «seuil» observé dans mes résultats : à partir de 50 % de volume récolté avec des méthodes inéquiennes, la quantité de chemins forestiers augmentait beaucoup moins (Figure 2-4). Cet effet résulte de la saturation du paysage par les chemins, alors que toute nouvelle zone de coupe finit par toujours se trouver près d'un chemin existant. Cela suggère que le compromis observé serait moins présent dans des paysages contenant plus de routes goudronnées ou permanentes.

Enfin, mes travaux du troisième chapitre ont permis de tester l'utilisation d'une stratégie d'adaptation pour augmenter la résilience de certaines fonctions des forêts face aux changements climatiques. Cette stratégie – la TRIAD+ – avait pour but de réussir cette adaptation par le biais de plantations fonctionnelles, qui devaient augmenter la diversité fonctionnelle des peuplements du paysage. Elle utilisait également un mélange de *sharing* et de *sparing* qui devait présenter un bon compromis entre production de bois et conservation des forêts à l'échelle du paysage. Les simulations utilisant une stratégie TRIAD+ ont alors réussi à augmenter la surface d'aires protégées du paysage sans compromis notable sur le volume de bois récolté ou récoltable. Elles ont également légèrement augmenté la résilience de la biomasse mature des forêts face à trois événements perturbateurs extrêmes. Cette augmentation fut cependant suffisamment petite pour remettre en question sa pertinence pour des objectifs d'aménagement forestier (appendice C.6). Elle n'était également pas toujours présente selon les perturbations et les indices de résiliences considérées. Ainsi, mes travaux n'ont pas permis de valider l'hypothèse selon laquelle la formulation actuelle de la TRIAD+ est réellement capable d'augmenter la résilience des forêts sur le long terme. Ils montrent néanmoins qu'il est possible d'inclure une composante d'adaptation au sein d'une stratégie d'aménagement complexe comme la TRIAD. Ces résultats remettent également en question la pertinence des plantations fonctionnelles comme stratégie d'adaptation. Toutefois, plusieurs éléments suggèrent que les extensions de LANDIS-II que j'ai utilisé pourraient avoir surestimé l'effet du climat, et sous-estimé l'effet de ces plantations fonctionnelles (voir section 4.3). Ces dernières conclusions sont alors à prendre avec un certain recul.

4.2 Comparaison avec d'autres études

Mes résultats s'alignent globalement avec ceux des études existantes en écologie forestière, mais avec certaines nuances importantes. L'algorithme du modèle de simulation de chemins forestiers présenté dans le Chapitre 1 est ainsi différent de la plupart des modèles existants dans ce domaine. Comme je l'ai déjà mentionné, ces modèles souvent faits pour des ingénieurs tendent à créer un réseau totalement optimisé, qui correspond à un arbre couvrant de poids minimal (*Minimal Spanning Tree* en anglais, abrégé par la suite MST) en théorie des réseaux (Heinimann, 2017). En contraste, mon algorithme résout un problème d'accès à de multiples cibles (Multiple Targets Access Problem ou MTAP) par le biais d'une heuristique. Il génère alors un résultat non optimal, mais sûrement réaliste compte tenu de la construction des réseaux forestiers dans la réalité (voir section précédente). On pourrait alors penser qu'il existe une dichotomie claire entre des modèles de simulation comme le mien faits pour la recherche en écologie, et des modèles d'optimisation utilisés pour des opérations forestières. Mon algorithme se rapproche néanmoins du fonctionnement d'un outil récent proposé dans le logiciel Woodstock de Remsoft, qui est utilisé pour préparer des opérations forestières (Remsoft, 2019). En contraste, des chercheurs à Environnement et Changement Climatique Canada ont récemment développé une bibliothèque R de simulation de réseaux routiers à l’échelle du paysage (Endicott *et al.*, 2023). Celle-ci propose différentes options de simulation, dont une optimisation complète du réseau par génération d’un MST. Il existe ainsi un mélange de ces deux approches dans les domaines de la recherche et de l’ingénierie, sûrement adaptés à des échelles et à questions différentes.

L’algorithme développé dans le Chapitre 1 à une performance similaire ou même supérieure que d’autres modèles existants. Par exemple, l’algorithme pour optimisation de Chung *et al.* (2008) a résulté en un réseau routier avec 20 % de routes en moins que celui proposé par des ingénieurs forestiers sur un même territoire. En contraste, le modèle ROADPLAN de Newnham (1995) (qui fonctionne de manière plus similaire à mon algorithme) était capable de correctement répliquer des réseaux planifiés par des ingénieurs forestiers. La performance de mon algorithme est cependant contestée par des simulations de Lochhead et Muhly du Ministère des Forêts de la Colombie Britannique, dont la méthodologie fut publiée en détail sur GitHub (Lochhead et Muhly, 2018). Leurs travaux ont comparé la performance de trois algorithmes pour répliquer la position de pixels de routes réels : optimisation via MST; génération via séquence de chemins les moins coûteux (correspondant le plus à ma méthode); et génération via séquence de chemins «à vol d’oiseau» (ligne droite entre la zone de coupe et le chemin le plus proche). Leurs résultats montrent que l’optimisation via MST prédit le plus de pixels de chemins réels parmi les trois méthodes sur 24 paysages différents, et génère une quantité de chemins réels la plus similaire à la réalité. Les paysages étudiés étaient cependant plus petits que les zones utilisées pour le test du Chapitre 1 (inférieur à 2 millions d’hectares), et les écarts de prédictions étaient relativement bas. Toutefois, ces résultats suggèrent que la résolution du MTAP implémentée dans le Chapitre 1 pourra être améliorée dans le futur, et que les réseaux routiers réels sont peut-être plus optimisés que ne le suggère leur développement.

Les résultats de mon Chapitre 2 semblent également bien se comparer avec les études existantes, bien que les données chiffrées sont rares sur le sujet. Nolet *et al.* (2018) suggèrent ainsi que l’aménagement inéquien utiliserait trois à cinq fois plus de surface pour récolter la même quantité de bois que l’aménagement équien, en prenant en compte des productivités forestières similaires. C’est exactement ce que j'ai observé dans le Chapitre 2, alors que les scénarios n’utilisant que des méthodes inéquiennes récoltaient environs cinq fois plus de surface à chaque pas de temps que les scénarios n’utilisant que des méthodes équiennes (Figure B-11). Tahvonen (2009) et Sharma *et al.* (2016) ont tous deux observé que l’utilisation de l’aménagement inéquien ou équien à l’échelle du paysage ne maximisait pas toutes les mesures désirées, ce que j'ai également observé. D'autres études ont également suggéré le lien entre la fragmentation des forêts et les feux révélé dans mes simulations (Driscoll *et al.*, 2021). En contraste, l'étude de modélisation de Gustafson (2007) a observé que l'utilisation d'aménagement équien tendait à plus fragmenter les différents types de forêts (par classe d'âge) à l'échelle du paysage que l'aménagement inéquien. Bien que ce résultat soit l'inverse de ce que j'ai observé, il est à noter que cette étude n'a pas pris en compte la présence de chemins forestiers, qui pourrait alors être déterminante dans ce cas précis. Additionnellement, je n'ai pas réussi à reproduire la réduction de la fragmentation du paysage par l'agrégation des coupes telles qu'observées par Gustafson (1998) et Gustafson (2007). Cela est sûrement dû au fait que l'agrégation des coupes n'a que peu changé la quantité de routes dans mes simulations, et que ces dernières étaient le facteur de fragmentation le plus important dans notre paysage. De plus, je n'ai pas utilisé les mêmes mesures de fragmentation que ces deux études qui ne se sont pas uniquement concentrées sur la fragmentation des vieilles forêts comme je l'ai fait. Il est également possible que les niveaux d'agrégation que j'ai utilisés étaient trop bas, et qu'ils auraient dû être augmentés pour observer un effet sur la fragmentation du paysage.

Concernant mon Chapitre 3, mes résultats sont similaires à ceux d'autres études qui ont observé que la TRIAD pouvait réaliser un meilleur compromis entre conservation et protection que des scénarios sans zonage (Côté *et al.*, 2010 ; Tittler *et al.*, 2015). Ces études restent cependant des études de simulation, comme la nôtre. L'évidence empirique de l'efficacité de la TRIAD reste ainsi manquante, et en attente des résultats d'implémentation comme dans la région de la Mauricie au Québec dans les années 2000 (Himes *et al.*, 2022 ; Messier *et al.*, 2009). Bien que les concepts utilisés dans ce chapitre s'inspirent de celui des réseaux fonctionnels de Messier *et al.* (2019), il est difficile de comparer mes résultats à ceux des études existantes sur cette notion. Les études de Aquilué *et al.* (2020, 2021) et Mina *et al.* (2021, 2022) ont ainsi modélisé l'effet de plantations fonctionnelles sur la résilience de différents paysages. Néanmoins, ces quatre études utilisent principalement la diversité fonctionnelle, la connectivité fonctionnelle et la redondance fonctionnelle comme indicateurs de la résilience des forêts modélisées. La diversité fonctionnelle y étant ainsi une variable réponse, ces études ne permettent pas de tester si son augmentation affecterait réellement la résilience de différentes fonctions des forêts comme je l'ai fait.

Plus globalement, l'hypothèse du lien entre résilience et diversité fonctionnelle est difficile à explorer au sein de la littérature. Comme exemple, je citerais ici deux études dont les résultats contrastent avec la nôtre, mais dont la méthodologie révèle la difficulté de ces comparaisons. Lucash *et al.* (2017) ont ainsi utilisé des simulations de LANDIS-II pour tester l'efficacité de plantations d'espèces considérées comme adaptées aux conditions climatiques futures. Ces plantations ont alors augmenté la résilience des peuplements touchés par des tempêtes ou des chablis dans leurs simulations. Hof *et al.* (2017) ont également utilisé LANDIS-II pour tester l'efficacité de différentes stratégies de plantations suite à des coupes forestières. Leurs simulations révèlent que des plantations qui diversifiaient les forêts tendaient à augmenter leur résilience. Plusieurs raisons rendent alors la comparaison entre mes résultats et les leurs difficile. La première est que LANDIS-II n'est pas capable de fixer le volume de bois récolté par différentes stratégies d'aménagement; c'est la raison pour laquelle j'ai développé l'extension Magic Harvest durant cette thèse qui m'as permis de surpasser ce problème. Mais les études existantes n'ont ainsi pas pu fixer ce volume, ce qui rend la comparaison entre stratégies d'aménagement plus compliquée. Il en effet difficile de savoir si l'effet d'une stratégie ou d'un scénario d'aménagement est dû à son niveau de récolte ou à ses méthodes de récolte, si le niveau de récolte n'est pas fixé entre scénarios. La deuxième est que ces études utilisent des mesures de résilience très différentes. Lucash *et al.* (2017) utilisent un indice qui décrit le retour de la biomasse et de la composition en espèces (via un indice de dissimilarité) sur ces deux dimensions, après qu'un peuplement ait subi une perturbation. En contraste, Hof *et al.* (2017) utilisent directement différentes variables à l'échelle du paysage (p. ex. biomasse aérienne, productivité primaire nette, indice de diversité de Shannon, etc.) comme des indicateurs de résilience. Leur étude ne considère pas la récupération ou le changement de ces mesures suite à une perturbation, mais simplement leur variation au cours de la simulation. Il est ainsi impossible de comparer l'évolution de mes mesures avec ceux de ces deux études, la résilience des forêts y étant définie de manière radicalement différente. Mais au-delà des mesures employées, celles-ci considèrent des fonctions différentes des forêts. Ces obstacles rendent l'unification des résultats entre différentes études complexe. Il me semble alors important de mieux coordonner les objectifs de recherche à travers le domaine de l'écologie forestière, que ce sois en favorisant les contacts entre équipes et projet, ou bien par la promotion de certains standards quant à la mesure de la résilience des forêts.

4.3 Limites des méthodes employées

Les limites de cette thèse reposent principalement sur l'utilisation du modèle LANDIS-II, qui fut employé à travers mes trois chapitres. Dans le Chapitre 1, ces limites sont principalement liées à la résolution spatiale à laquelle LANDIS-II fonctionne, et à sa discrétisation de l'espace en cellules. Dans la réalité, les chemins forestiers sont des lignes vectorielles dont la forme ne peut pas entièrement être capturée par un espace discrétisé. Malgré mes efforts, cette limitation reste importante, et pourra sûrement influencer les résultats obtenus dans les paysages ou la topographie locale est complexe. L'extension FRS risque également d'être plus limitée dans des paysages plus denses en routes permanentes et en lieux résidentiels, où l'évolution des routes est déterminée par des facteurs sociaux et économiques complexes qui dépassent totalement le cadre d'une simulation de LANDIS-II. L'extension reste ainsi principalement adaptée à des paysages non urbains.

Les résultats des Chapitres 2 doivent quant à eux être vus dans le contexte des définitions que j'ai employées ainsi que des processus que j'ai inclus dans les simulations. Comme noté dans la discussion du Chapitre 2, la définition de vieille forêt que j'y ai utilisée influence grandement mes conclusions. En particulier, l'aménagement inéquien aurait été encore moins efficace dans mes simulations si j'avais choisi de parler de forêt «non perturbée» ou «non récemment perturbée». Mes conclusions quant aux avantages de l'aménagement inéquien concernent alors le type particulier de vieille forêt que j'ai utilisé, et ne peuvent être généralisées à tous les objectifs de conservation. S'ajoute à cela le fait que certains facteurs importants n'ont pas été pris en compte dans le Chapitre 2, par souci de ne pas trop complexifier les simulations et par contraintes de temps. Le premier de ces facteurs concerne les changements climatiques, dont l'inclusion aurait en particulier influencé les régimes de feux de notre paysage qui seraient devenus plus fréquents. Leur inclusion aurait alors pu réduire davantage les différences entre scénarios d'aménagements (en termes de quantité de vieille forêt et de fragmentation) comme l'ont fait les feux de forêt dans la partie nord de notre paysage simulé. Les effets des chemins forestiers pourraient également se combiner avec ceux des changements climatiques en augmentant les taux d'ignition de feux (Narayanaraj et Wimberly, 2012), mais aussi aider à stopper leur progression (Narayanaraj et Wimberly, 2011). Ils pourraient aussi aider à diffuser des espèces invasives dont la croissance serait facilitée par les changements climatiques (Mortensen *et al.*, 2009), et perturber davantage les interactions entre espèces comme dans le cas du Caribou et de ses prédateurs (Labadie *et al.*, 2023). Je n'ai également pas inclus de mesures d'acceptabilité sociale dans mes simulations, qui auraient pu désavantager les scénarios les plus extrêmes (100 % équien ou inéquien). Je n'ai aussi pas pu intégrer de variabilité dans la demande de bois, qui pourrait fortement influencer la dynamique des chemins forestiers avec des années ou les coupes seraient beaucoup plus ou moins nombreuses. L'introduction d'une variabilité annuelle liée à des facteurs sociaux complexes dans les surfaces récoltées de LANDIS-II serait maintenant possible grâce à l'extension Magic Harvest que j'ai développée. Elle demanderait toutefois une paramétrisation basée sur des analyses économiques et sociales, et aussi en lien avec la perception sociale des coupes et les impacts des changements climatiques. Enfin, je n'ai pas exploré la possibilité de décommissionner des chemins forestiers après leur construction. En dehors de certain chemins d'hiver, la décommission reste un processus très coûteux qui est, à ma connaissance, très peu utilisé au Québec où les chemins sont souvent utilisés par les communautés locales une fois construits. En contraste, on assiste plutôt au Québec à la dégradation des chemins forestiers laissés à eux même (Jutras, 2022). Il est difficile de dire si la décommission de chemins aurait pu réduire la quantité de chemins générée dans les scénarios utilisant des méthodes inéquiennes. Les coupes inéquiennes nécessitant des retours périodiques dans les peuplements, il parait en effet inconcevable que les chemins soient démantelés et reconstruits en permanence à chaque rotation.

Concernant les limites des résultats du Chapitre 3, j'amène ici un autre point important qui est venu à ma connaissance depuis sa rédaction, et qui expliquerait d'autant plus les forts effets des changements climatiques observés dans mes simulations. Brice (2023) a présenté une comparaison de la dynamique des espèces d'arbres au Québec sous l'influence des changements climatiques, telle que simulée par différents modèles. Parmi ces modèles, on trouve le modèle PICUS qui a été utilisé pour paramétrer le fonctionnement de l'extension Biomass Succession pour les différents chapitres de cette thèse. Cette paramétrisation était particulièrement cruciale pour le Chapitre 3, car elle devait refléter l'impact des changements climatiques sur la croissance des espèces simulées. Les résultats présentés par Brice ont révélé que PICUS avait tendance à avoir une vision beaucoup plus pessimiste que d'autres modèles basés sur des mesures empiriques (p. ex. ARTEMIS et NATURA du MRNF; voir Auger et Power 2019). Ce pessimisme de PICUS surestimait l'impact des changements climatiques sur la croissance de certaines espèces, ce qui pourrait être dû à une mauvaise simulation de leur optimum thermique (Brice, 2023). Ce problème lié à PICUS a pu ainsi s'ajouter aux manques actuels de LANDIS-II en ce qui concerne la représentation de processus à l'échelle du peuplement (p. ex. compétition et complémentarité) pour produire des résultats moins réalistes que ce que j'envisageais. Malgré cela, je pense que mes recherches ont donné une direction claire quant aux futures recherches qui devront être faites dans le domaine, surtout en qui concerne des stratégies d'aménagement futures et l'amélioration des modèles utilisés pour les explorer.

4.4 Implications et importance des contributions de la thèse

Les contributions de cette thèse au domaine de l'écologie forestière me paraissent essentielles dans le contexte actuel. La quantité de bois récoltée chaque année dans le monde ne fait qu'augmenter, et à cette récolte s'ajoute à la présence toujours plus lourde des changements globaux. Ces contributions ne viennent malheureusement pas avec des réponses simples, mais avec des réponses nuancées qui reflètent la complexité des écosystèmes forestiers. Pris dans leur ensemble, les travaux de ma thèse suggèrent ainsi qu'il est difficile de conjuguer les différentes missions actuelles de l'aménagement forestier. Ils suggèrent également que les changements globaux rendront l'union de ces missions d'autant plus difficiles. Les méthodes de *land-sparing* et *land-sharing* en foresterie semblent en effet impliquer des compromis complexes liés à la surface totale récoltée et à l'impact des coupes forestières. Comme je l'ai déjà mentionné, cette conclusion s'aligne avec celles d'autres études qui ont exploré les effets des stratégies de *sharing* et *sparing* (Betts *et al.*, 2021a ; Sharma *et al.*, 2016 ; Tahvonen, 2009). Mes travaux ont cependant surligné l'importance particulière des chemins forestiers dans ce compromis, les stratégies intensives et extensives ayant de forts impacts sur leur quantité. Ceci est particulièrement le cas dans des régions aux vastes étendues forestières comme le Québec, mais aussi dans des zones tropicales comme en Amazonie ou les chemins deviennent souvent des «frontières» où la déforestation progresse vers des forêts encore intouchées (Kaimowitz et Angelsen, 1998 ; Nepstad *et al.*, 2001 ; Uhl *et al.*, 1991). Les futures recherches devraient alors profiter des opportunités présentées par les outils libres et gratuits comme l'extension FRS pour incorporer la simulation des chemins forestiers. À ce sujet, l'extension FRS a depuis été utilisée dans l'étude de Labadie *et al.* (2023) pour simuler l'impact de l'aménagement forestier et des changements climatiques au sein de LANDIS-II sur trois espèces : le loup, le caribou et l'original. Le développement des chemins fut simulé par l'extension FRS, et les chemins créés furent utilisés pour alimenter un modèle multi-agent (*agent-based model*) simulant le déplacement d'individus des trois espèces. L'extension FRS est donc maintenant utilisée pour des questions plus spécifiques liées à la conservation de la biodiversité. Plus globalement, la création de cette extension semble cruciale à une époque où les chemins forestiers sont considérés avec beaucoup d'attention par le public, par la communauté scientifique, et par les instances gouvernementales pour leurs impacts sur les écosystèmes forestiers.

Mon deuxième chapitre présente quant à lui un message fort concernant l'adoption de l'aménagement inéquien ou plus extensif à de grandes échelles. Mes simulations suggèrent que ces solutions seront coûteuses, et pas forcément désirables pour protéger les écosystèmes forestiers et leurs fonctions. Elles n'invalident cependant pas leur utilisation, qui comporte des avantages clairs en ce qui concerne la conservation de forêts plus matures et complexes. En cela, il est possible de voir mes travaux comme un argument supplémentaire pour l'utilisation de stratégies similaires à la TRIAD, qui proposent un mélange stratégique d'intensif et d'extensif. Le contexte de notre zone d'étude se prête à l'implantation de telles stratégies d’aménagement audacieuses sur le long terme, car les forêts y sont principalement gérées par le gouvernement. Il sera cependant plus difficile d’implémenter des mélanges de coupes intensives et extensives plus stratégiques dans des territoires composés de nombreuses parcelles de forêt privée, car cela demandera une coordination entre un plus grand nombre d'acteurs (Tittler *et al.*, 2016). Ces contributions sont importantes à une époque où la foresterie intensive reste mal perçue par une partie du grand public (Bliss, 2000 ; Hemström *et al.*, 2014 ; Yelle, 2013), et où le débat entre équien et inéquien continue d’exister dans la littérature scientifique (Nolet *et al.*, 2018). Mes travaux impliquent alors qu’un effort soutenu de communication avec le grand public sera important pour expliquer la pertinence des stratégies intensives à l’échelle du paysage, et que les futures recherches devront considérer cette échelle plus souvent. Néanmoins, la communauté scientifique doit rester méfiante quant à la possibilité de l’instauration du paradoxe de Jevons dans le cas où des stratégies intensives seraient plus acceptées par le public. En effet, si de telles stratégies ne mènent pas forcément à la destruction des écosystèmes forestiers, elles peuvent facilement devenir un outil de cette destruction sous certaines motivations économiques. Notre monde regorge ainsi d’exemples où les nuances amenées par la science furent utilisées pour justifier des pratiques douteuses, ou bien pour entretenir un doute permettant de cacher les effets de ces pratiques (Oreskes et Conway, 2010). Des recherches supplémentaires resteront ainsi nécessaires pour mesurer l’évolution des compromis entre *sparing* et *sharing* par le biais d’observations, d’expérimentation ou de modélisation. Il sera important d’effectuer ces recherches dans différents contextes, comme des pays plus industrialisés ou plus pauvres où les niveaux de déforestation et d’urbanisation sont différents.

Pour finir, les travaux de mon Chapitre 3 ne sont pas rassurants en ce qui est des perspectives à long terme de l’aménagement forestier. Mes simulations ont montré un effet très fort du climat sur mes différentes mesures (p. ex. biomasse mature et diversité fonctionnelle), face auquel les stratégies d’aménagement que j'ai implémentées étaient peu effectives. En particulier, les plantations fonctionnelles que j'ai implémentées n’ont pas eu beaucoup d’effet sur mes mesures de résiliences. Ces résultats suggèrent des implications différentes selon leur interprétation. La première est que les stratégies d’adaptation basées sur l’augmentation de la diversité de réponse des forêts ne seront pas efficaces pour augmenter leur résilience de manière suffisante. La plupart des stratégies d’adaptation développées aujourd’hui sont toutefois basées sur l’augmentation de la complexité structurelle ou la diversité spécifique des forêts (Jandl *et al.*, 2019). Mes résultats suggèrent alors d’explorer d’autres options que ces stratégies. De manière plus pessimiste, mes résultats pourraient aussi être annonceurs d’une vision plus sombre du futur;

que quelles que soient nos actions, les forêts subiront de graves bouleversements qui ne pourront être évités. À la date de la rédaction de ces mots, l'agence de presse Reuters a récemment publié un reportage concernant les efforts de plantations d'espèces plus résistantes aux changements climatiques en Allemagne (Reuters, 2023). Les forestiers interviewés rapportent que la plupart des plants sont morts de sécheresse, et qu'il devient «de plus en plus difficile» de changer la composition des forêts pour les préparer aux conditions futures. Les pépinières forestières du Québec semblent également avoir de plus en plus de mal à fournir un nombre suffisant de plants (Carles, 2023). Ce problème semble causé par de forts taux de mortalité liés à des aléas climatiques de plus en plus fréquents et intenses. Dans un tel contexte, même si les plantations fonctionnelles s'étaient révélées efficaces, il est alors difficile de dire si elles pourraient alors être implémentées à grande échelle. Bien que des études empiriques sur le sujet seront absolument cruciales dans le futur, on peut se demander si elles pourront être faites à temps, alors que les changements climatiques continuent de bouleverser les écosystèmes du monde entier.

Il est cependant possible de voir mes résultats d'un autre œil. Comme noté dans la discussion du Chapitre 3 et dans la section 4.3, il est possible que notre approche de modélisation ait donné trop d'importance à l'effet des changements climatiques. Dans cette vision, mes résultats souligneraient alors la nécessité d'un effort urgent de développement et de raffinement des Forest Landscape Models. En particulier, il serait nécessaire de proposer de nouvelles approches qui intègrent mieux la simulation de processus à l'échelle des arbres qui composent les peuplements (p. ex. régénération, densité d'arbres et auto-éclaircissement, compétition et complémentarité, croissance, etc.) et celle de processus à l'échelle du paysage (p. ex. perturbations naturelles et humaines, dispersion, etc.). De tels efforts semblent être à l'œuvre dans la communauté de LANDIS-II (Fraser *et al.*, 2020), mais aussi en dehors par le développement des modèles comme iLand (Seidl *et al.*, 2012) qui combinent la simulation d'un paysage forestier entier avec celle d'arbres individuels. La manière dont l'aménagement forestier est étudié devra alors continuer de changer avec l'intégration de nouvelles connaissances, tout en profitant des capacités de calcul toujours plus importantes de nos ordinateurs.

Quoi qu'il en soit, mes recherches montrent qu'il existe des opportunités importantes d'intégrer des stratégies d'adaptation au sein stratégies d'aménagement ambitieuses et innovantes comme la TRIAD+. Elles ouvrent alors le chemin à l'exploration et au design de nouvelles stratégies qui intégreront les connaissances les plus récentes en ce qui concerne la résilience des écosystèmes forestiers. Ces stratégies novatrices pourraient cependant questionner le rôle de l'aménagement forestier en tant qu'industrie dans nos sociétés. De telles stratégies demandent en effet une coordination importante sur le long terme et à grande échelle, souvent mieux menée par des organismes gouvernementaux. Elles peuvent également impliquer des traitements forestiers coûteux pour aider les forêts face aux changements climatiques : l'aide à la régénération avec des plantations, la décommission de chemins forestiers, la suppression de perturbations naturelles, etc. Sous cet angle, il est alors possible d'imaginer que l'aménagement forestier pourrait transitionner depuis une activité industrielle vers une activité plus similaire à celle d'un service public. Il est possible que cette transition ait déjà commencé dans des régions comme le Québec, ou l'aménagement forestier est très régulé par l'état. Les conséquences économiques et sociales d'une telle transition sont difficiles à estimer, et en dehors du cadre de cette thèse. Néanmoins, il faut rappeler que la foresterie représente aujourd'hui moins de 1 % du PIB mondial (Agrawal *et al.*, 2013), et seulement 1.6 % du PIB du Québec même si les forêts recouvrent plus de la moitié du territoire de la province (MRNF, 2023). Quoi qu'il en soit, l'aménagement forestier devra se transformer drastiquement dans le futur pour réussir à satisfaire toutes les demandes qui l'entourent déjà aujourd'hui. De cet enjeu dépendra en partie le sort des forêts du monde, et de l'état dans lequel nous les laisserons pour les futures formes de vie de notre planète.

APPENDICE A

Supplementary Material for : A LANDIS-II extension for simulating forest road networks

A.1 Packages used to optimize performances

We improved the computing performance of the module by using two open-source C# NuGet packages: “Supercluster.KDTree” (Regina, 2015) and “OptimizedPriorityQueue” (BlueRaja, 2015). The first package allows to partition the recently harvested cells and the exit points into a particular binary tree data structure called a “k-d tree” (Bentley, 1975). This tree is made by repeatedly splitting a multi-dimensional space (with a number “k” of dimensions) on the position of points in that space (here, the cells centroïd). At each split, the remaining points (not used for splits) are categorized in one part of the split or the other, effectively creating a binary tree. Using a k-d tree, a fast nearest-neighbor search can be performed using the tree’s structure to find the exit point closest, in Euclidean distance, from any harvested area of interest. Once that the closet exit point is found for every harvested area, these can be ranked according to the “closest first” or “farthest first” heuristics. Hence, the k-d tree dramatically improves the computation time of the module by eliminating the need to compute the Euclidean distance for every pair of recently harvested cells and exit points at each time step.

The second package, “OptimizedPriorityQueue”, contains a priority queue object optimized for the C# language which improves the speed of the Dijkstra algorithm. The algorithm work by successively integrating cells into the “frontier” of its search space by considering the neighbors of the cells at the frontier. Each cell integrated into the frontier is given a “priority” value, which is used to determine which cell will be considered at the next iteration. In the case of the Dijkstra algorithm, this priority value is the least-cost path from the starting point of the search to this cell, with lower values being prioritized. The optimized priority queue proposed by the package takes advantage of the object-oriented structure of the C# language to reduce the number of operations needed to order a new cell (and its corresponding value). This, in turn, make the Dijkstra algorithm much faster.

A.2 Equations of the fragmentation indices

A.2.1 Clumpy

The Clumpiness Index (Clumpy; McGarigal *et al.* 2012) for the patch type (class) i is given in the following equation:

$$CLUMPY_i = \begin{bmatrix} \frac{G_i - P_i}{P_i} for G_i < P_i P_i < 0.5; else \\ \frac{G_i - P_i}{1 - P_i} \end{bmatrix}$$

where

$$G_i = \left(\frac{g_{ii}}{\left(\sum_{k=1}^{m} g_{ik}\right) - \text{min-e}_i} \right)$$

and:

1. g_{ii} is the number of like adjacencies between pixels of patch type (class) *i* based on the double-count method,
2. g_{ik} is the number of adjacencies between pixels of patch types (classes) *i* and *k* based on the double-count method. Here, *k* are the other existing patch types (class) in the landscape,
3. $\min - e_i$ is the minimum perimeter (in number of cell surfaces) of patch type (class) *i* for a maximally clumped class, and
4. P_i is the proportion of the landscape occupied by patch type (class) i.

Clumpy corresponds to the deviation of the proportion of like adjacencies between pixels of type (class) *i*, when compared with a random distribution of those pixels. It varies between -1 and 1. When Clumpy approaches 0, the pixels of type i are aggregated as they would be if they were distributed randomly. When Clumpy approaches -1, the pixels of type *i* are distributed in a way that they are less aggregated than in a random distribution. When Clumpy approaches 1, the pixels of type *i* are aggregated more than in a random distribution.

A.2.2 TCA

The Total Core Area (TCA; McGarigal *et al.* 2012), for the patch type (class) *i* is given in the following equation:

$$TCA_i = \sum_{j=1}^{n} a_{ij}^{core} * \left(\frac{1}{10000}\right)$$

where:

- a_{ij}^{core} is the core area of the patch *j* of the patch type (class) *i*. In our study, the core area pixels are defined as pixels that have no neighbours different than their own class; if not, they are considered on the edge of the patch.

As the quantity of core area of patches of type (class) *i* in the landscape increases, TCA increase too. It varies from 0 (no core area), and is only limited by the total size of the landscape.

A.2.3 PAFRAC

The Peremiter-Area Fractal Dimension (PAFRAC, McGarigal *et al.* 2012) for a patch of type (class i) is given in the following equation:

$$\frac{2}{\dfrac{\left[n_i \sum_{j=1}^{n}\left(lnp_{ij} \cdot lna_{ij}\right)\right] - \left[\left(\sum_{j=1}^{n} ln\, p_{ij}\right)\left(\sum_{j=1}^{n} ln\, a_{ij}\right)\right]}{\left(n_i \sum_{j=1}^{n} ln\, p_{ij}^2\right) - \left(\sum_{j=1}^{n} ln\, p_{ij}\right)^2}}$$

where:

- $\mathrm{a}_{\mathrm{ijis}}$ the area of patch j of the patch type (class) i,
- $\mathrm{p}_{\mathrm{ijis}}$ the perimeter of patch j of the patch type (class) i, and
- n_{i} the total number of patches of the patch type (class) i in the landscape.

PAFRAC varies from 1 to 2. As its value departs from 1 and approaches 2, the shape of the patches of type (class) *i* depart from a Euclidean geometry, and take more complex shapes. Therefore, PAFRAC approaches

1 for patches with very simple shapes (perimeter) such as squares, and approaches 2 for shapes with highly convoluted, plane-filling perimeters.

APPENDICE B

Supplementary Material for : Land sparing and sharing patterns in forestry: exploring even-aged and uneven-aged management at the landscape scale

B.1 Parameter origin

Table B-1 describes the origin of the parameters of each extension of LANDIS-II. The method for the obtention of each parameter is described in further sections.

Table B-1 : Source of the values used for our parameters for the LANDIS-II simulations for each of the five modules used.

LANDIS-II module (Appendix for detailed explanation)	Parameter(s)	Data source
Core module (A)	Species used and their life-history traits	Boulanger *et al.* (2017), Tremblay *et al.* (2018).
	Ecoregions	Boulanger *et al.* (2017), and based on climate and homogeneous soil conditions (Mansuy *et al.*, 2014).
	Initial communities	Boulanger *et al.* (2017), Tremblay *et al.* (2018) and data from the Canadian National Forest Inventory.
Biomass Succession (B)	Basic parameters (seed dispersal method, effects of shade, species traits related to biomass, effects of fire and harvest)	Boulanger *et al.* (2017), Tremblay *et al.* (2018).
	Ecoregion-specific parameters (probability of establishment, maximum net primary productivity and maximum aboveground biomass)	Boulanger *et al.* (2017), and results from PICUS simulations on a stand of 1 hectare for each tree species used, with the correct soil and climatic conditions (Mansuy *et al.*, 2014).
Base Fire (C)	Basic parameters (fire damage parameters, etc.)	Boulanger *et al.* (2017).
	Fire region-specific parameters (minimum and maximum size, ignition probability, etc.)	Boulanger *et al.* (2017), historical data for our study area, and estimations from calibration scenarios.
Biomass Harvest (D)	Management areas and stands	Current management units of Quebec and data from the 5th National Forest Inventory of Quebec (MFFP, 2018a).
	Prescriptions planning and effects	Current methodology used by of the Ministry of Forests, Wildlife and Parks of Quebec (MFFP) to assess the allowable wood

		volumes to harvest (Bureau du forestier en chef du Quebec, 2013).
	Surface to harvest	Determined by the use of a custom Python script launched using the Magic Harvest extension for LANDIS-II (Hardy, 2022).
Forest Roads Simulation (E)	Construction heuristic, skidding distance, looping behaviour	Determined using calibration scenarios to replicate characteristics of the existing road network in our study area.
	Cost raster	Forest roads cost data gathered by the MRNF and opinions of governmental infrastructure experts.
	Thresholds for the quantity of timber to transport during a time step by road types	Parameterized so that the proportion of each road type in a management strategy simulation is similar to the proportion used in a current management strategy in the study area.
	Multiplicative cost values	Data gathered by the MRNF.
	Max age before destruction	Road duration by type determined by the MRNF.

B.2 Core Parameters

B.2.1 Species used and their life history traits

The species and their associated life-history traits that we used are the same as those in the study by Tremblay et al. (2018). This is detailed in Table B-2. These were derived from various sources (e.g., Burns *et al.*, 1990 ; Farrar, 1995) and from expert judgment when empirical sources did not exist (Tremblay *et al.*, 2018).

Table B-2 : Life-history traits parameters of the 17 tree species used in our simulations.

Real name	Code name	Longevity (year)	Age of sexual maturity	Shade tolerance	Fire tolerance	Seed dispersal distance		Vegetative reproduction probability	Sprout age		Post fire regeneration
						Effective	Max		Min	Max	
Abies balsamea	ABIE.BAL	150	30	5	1	25	160	0	0	0	none
Acer rubrum	ACER.RUB	150	10	3	2	100	200	0.5	10	100	resprout
Acer saccharum	ACER.SAH	300	40	5	2	100	200	0.1	10	60	resprout
Betula alleghaniensis	BETU.ALL	300	40	3	1	100	400	0.1	10	180	resprout
Betula papyrifera	BETU.PAP	150	20	2	1	200	5000	0.5	10	70	resprout
Fagus grandifolia	FAGU.GRA	250	40	5	1	30	3000	0.5	10	30	none
Larix laricina	LARI.LAR	150	40	1	1	50	200	0	0	0	none
Picea glauca	PICE.GLA	200	30	3	2	100	303	0	0	0	none
Picea mariana	PICE.MAR	200	30	4	2	80	200	0	0	0	serotiny
Picea rubens	PICE.RUB	300	30	4	1	100	303	0	0	0	none
Pinus banksiana	PINU.BAN	150	20	1	2	30	100	0	0	0	serotiny
Pinus resinosa	PINU.RES	200	40	2	3	12	275	0	0	0	none
Pinus strobus	PINU.STR	300	20	3	3	100	250	0	0	0	none
Populus tremuloides	POPU.TRE	150	20	1	2	1000	5000	0.9	10	150	resprout
Quercus rubra	QUER.RUB	250	30	3	3	30	3000	0.75	20	200	resprout
Thuja occidentalis	THUJ.SPP.ALL	300	30	5	1	45	60	0.1	10	60	none
Tsuga canadensis	TSUG.CAN	300	60	5	1	30	100	0	0	0	none

All of these species are among the most present species in our simulated area, as shown in Figure B-1. The species on the far right of the figure corresponds to the red oak, included in our simulations because of its status as a very valuable species in Quebec despite its relatively low abundance.

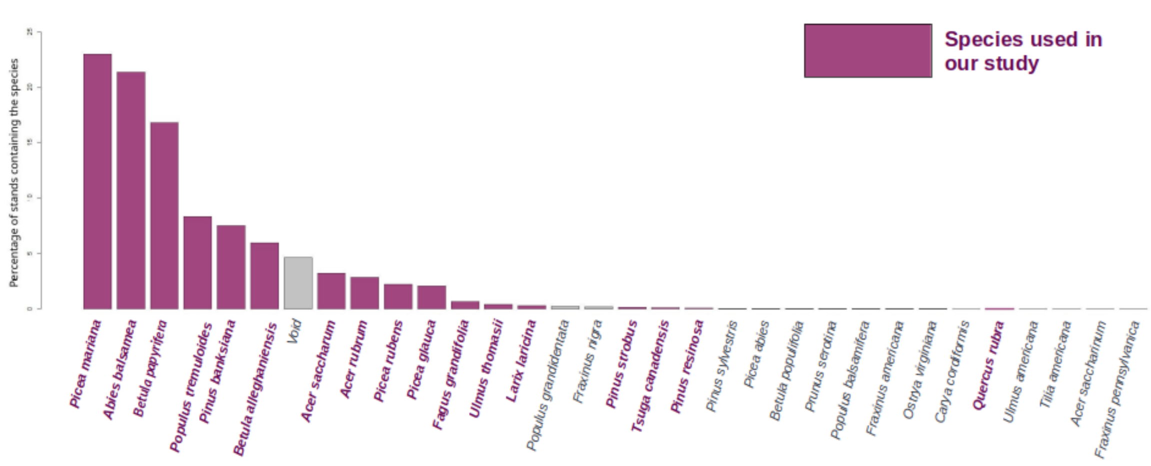

Figure B-1 : Histogram of the percentage of the forest stands identified by the 5th National Forest Inventory in our landscape where the tree species are predicted to be present. The three-letter codes are those used for the tree species by the Ministry of forests of Quebec. The purple rectangles indicate the codes corresponding to the 17 tree species used in this study.

B.2.2 Ecoregions

Our ecoregions were defined in the same way as those of Boulanger et al. (2017) and Tremblay et al. (2018) based on two datasets: the extent of the “ecodistricts” of Quebec (Ecological Stratification Working Group *et al.*, 1996); and soil data from Mansuy *et al.* (2014). We used the k-means method (Forgy, 1965) to define ecoregions that minimized the variance for variables of these two datasets (soil and climate), but also other factors influencing tree growth such as slope and exposure. The limits of the resulting ecoregions can be seen in Figure B-2.

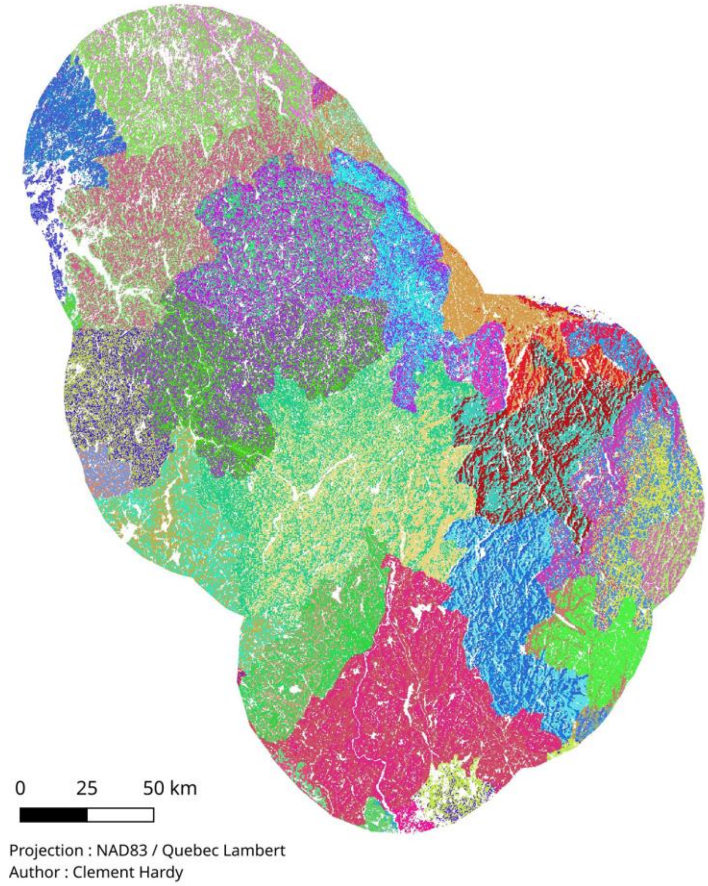

Figure B-2 : Raster containing the spatial extent of the different ecoregions in our simulated area. Each shade of colour on the map represents a different ecoregion, where soil and climate conditions are considered homogeneous. White pixels represent areas where growth is impossible, i.e., urban areas, lakes or rivers.

B.2.3 Initial communities

LANDIS-II requires a raster indicating the initial communities of tree species present in each cell of the simulated area at the beginning of the simulation. This raster contains “map codes” that are linked to a particular community of age-cohort of different species, each code being referenced in another parameter text file. To obtain this information, we used the same protocol as Boulanger et al. (2017) and Tremblay et al. (2018). Hence, we used the biomass maps for all of our main tree species from the National Forest Inventory (NFI) of Canada and several climatic attributes (e.g., yearly precipitation) to assign the tree communities of the sample plots of the government of Quebec to the cells of our initial landscape using the k Nearest Neighbor assignment (k-NN) method (Beaudoin *et al.*, 2014). Several imputations were performed to isolate cells where the forest hadn’t been disturbed for a certain time so that we could link them to the sample plots of the corresponding age, allowing for a more robust imputation in the end.

After the imputation, the data was then treated with scripts that transformed it into LANDIS-II map codes and initial communities. As seen in Figure B-3, the resulting map of initial conditions is complex, and does not show any clear pattern or repetition of a particular code of initial community. This comes from the fact that the number of sample plots in Quebec used for the assignment is very high (> 100 000), resulting in a complex and realistic initial landscape.

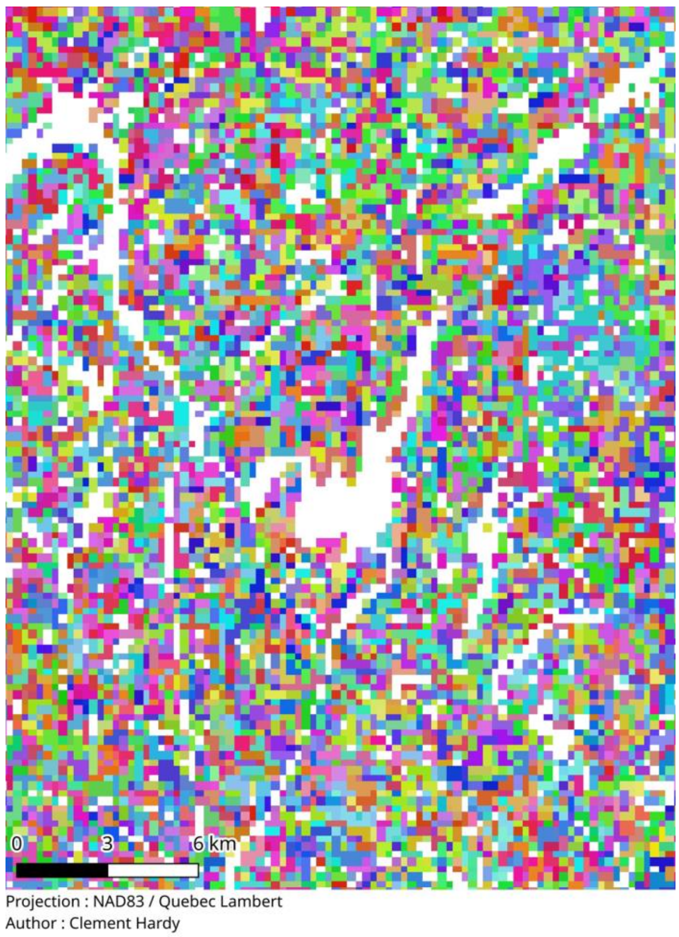

Figure B-3 : Extract of the initial conditions map. Each colour corresponds to a different and unique initial community, which is an ensemble of age cohorts of different species. White indicates inactive pixels where there are no forests.

The fit of the results of such a method was tested by Boulanger et al. (2017) (see "Supplementary Material S6" of their appendices). The test was a comparison of the initial estimates (t = 0) of the biomass in each cell made by LANDIS-II (that is based on the age cohorts that have been given as initial conditions) and the biomass maps of the NFI (that were used during the imputation by the k-NN method). As the great majority of the cells of LANDIS-II had an initial content not very different from the NFI maps, it was thus accepted that the method had produced satisfactory results

B.3 Biomass succession parameters

B.3.1 Basic parameters

B.3.1.1 Seed dispersal algorithm

We chose the Ward algorithm, currently the most realistic of the three proposed by LANDIS-II, as the distance of dispersal for each species is regulated by two negative exponential distributions that calculate the probability of a seed landing in a nearby site, and parameters can vary according to each species (see Table B-1).

B.3.1.2 Configuration file

With Biomass Succession, we used no climate configuration file as our simulations did not consider a changing climate among scenarios.

B.3.1.3 Spin up mortality

The spin-up mortality parameter was fixed to the low value of 0.01, as our parametrization of both the initial conditions and the parameters regulating the biomass of the age cohorts were precise enough so that a significant correction in the initial biomass estimates due to mortality was not required.

B.3.1.4 Minimum relative biomass for each shade class

Simulations of vegetation dynamics in LANDIS-II were made on the scale of a millennium, and simulations of minimum relative biomass on the scale of a single pixel for each shade class empirically (see “Supplementary Material S5” of Boulanger et al. 2017). The parameters were thus tweaked until satisfactory vegetation dynamics were obtained for the main forest compositions in our area: temperate forest, mixed-temperate forest, and softwood boreal forest. More information on these tests is available in the documents on the Github repository of Dominic Cyr (https://github.com/dcyr/LandisPixelLevelSimulations).

B.3.1.5 Probability of establishment given light conditions

The probabilities of establishment of the species given light conditions and tolerance to shade corresponded to the default values of LANDIS-II as those produced acceptable dynamics in conjunction with the defined species life-history traits.

B.3.1.6 Species-specific biomass parameters

These parameters were all derived from existing literature on the 17 tree species, with the exception of the shape parameters of the growth curve and the mortality curve for each species. Those were derived from both trial and error to produce realistic vegetation dynamics and the position of the different species in the ecological succession after a disturbance. A special accommodation was made for the red spruce (*Picea rubens*), which tended to generate too much biomass during the simulations in comparison to other species when the same logic was applied to its parameters.

B.3.1.7 Evapotranspiration parameterization

The parameter was fixed at the same value of 600 for all of our ecoregions as it was linked to the decay rate for the decomposition of the leaves to dead biomass, a process which we were not interested in simulating for our research question.

Dynamic input file parameters

Each of the three parameters in this file (establishment probability, maximum annual net primary productivity and maximum biomass) were derived from PICUS simulations, as was the case for the studies of Boulanger et al. (2017) and Tremblay et al. (2018). A complete account of how PICUS was used for our study can be found in the appendices of Boulanger et al. (2017) (see “Supplementary Material S3”). Each PICUS simulation was thus parameterized to simulate stands of one hectare containing one of our 17 tree species, with climate and soil conditions corresponding to a given ecoregion. The dynamic of the stands was then simulated for 300 years.

The scripts used to transform the outputs of PICUS into the set of parameters needed for LANDIS-II can be found on the GitHub repository of Dominic Cyr who was in charge of the task (https://github.com/dcyr/PicusToLandisIIBiomassSuccession). The protocol of transferring the outputs of

PICUS into the parameters needed by LANDIS-II is also described in the appendices of Boulanger et al. (2017) (see "Supplementary Material S4").

B.4 Base fire parameters

B.4.1 Parameters specific to the fire regime

These were determined using the values associated with the homogeneous fire regions of Boulanger *et al.* (2014). Hence, the minimum, maximum and mean fire sizes of our fire regions (see section B.4.2) corresponded to those of the associated homogeneous fire regions of Boulanger *et al.* (2014).

Meanwhile, the ignition probability and the "k" that influences both the probabilities of fire ignition and spreading (see Scheller et Domingo 2018, equation 1) were estimated empirically using calibration scenarios to reproduce the Percentage of Annual Area Burned (PAAB) of our two fire regions based on the homogeneous fire regions. To that end, we ran simulations of 30 years (3 time steps) on our landscape with the correct initial conditions, and no disturbance other than fire. We expected 30 years to be enough as the fire regime of the homogeneous fire regions of Boulanger *et al.* (2014) would not necessarily associate correctly with the forest communities simulated by LANDIS-II if the simulation lasted too long.

B.4.2 Fire regions

The fire regions were defined using the LANDIS-II ecoregions and the homogeneous fire regions, to avoid the geometric-like boundaries of the fire regions of Boulanger *et al.* (2014). When the majority of a LANDIS-II ecoregion was composed of a homogeneous fire zone, then the whole LANDIS-II ecoregion was designated as this homogeneous fire zone. This resulted in the map in Figure B-4.

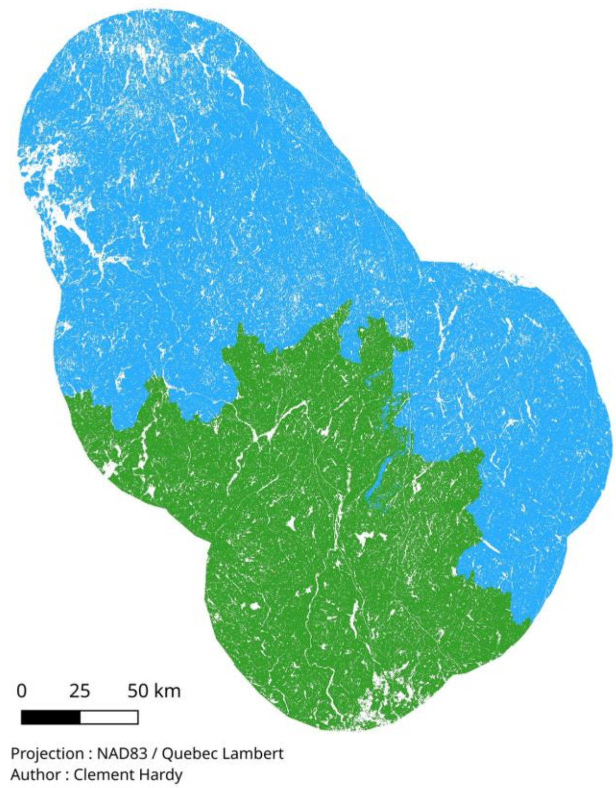

Figure B-4 : Map showing the two fire regions in our simulated area. The boreal fire zone is in the north (blue), and the mixed-temperate fire zone is in the south (green).

B.4.3 Fire severity and damage

As crown fires are the most frequent type of fire in the study region (Jayen *et al.*, 2006), we parametrized the Base Fire module to only such fires using a single severity class in the parameter file. This severity class was activated when the cell hadn't burned for more than 10 years: if less than 10 years had passed, no fire would happen on the cell. This severity of fire then damaged the tree species depending on their fire tolerance: the more tolerant a tree species, the less damage its age cohorts take from fire.

B.5 Biomass Harvest parameters

B.5.1 Management areas

The management areas used for the Biomass Harvest module were defined to replicate the existing management areas in our landscape. Their spatial limits corresponds to the different management areas currently existing in our simulated area, as defined by the Ministry of Natural Ressources and Forests of Quebec (MRNF, formerly the MFFP) (Figure B-5A).

In addition to these, the Python script used with the Magic Harvest extension read a raster indicating the distance of a cell to the main road network of the landscape, with 5 categories of distance (Figure B-5B). This was used to simulate a progressive "invasion" of the landscape by roads in the scenarios with no initial road network.

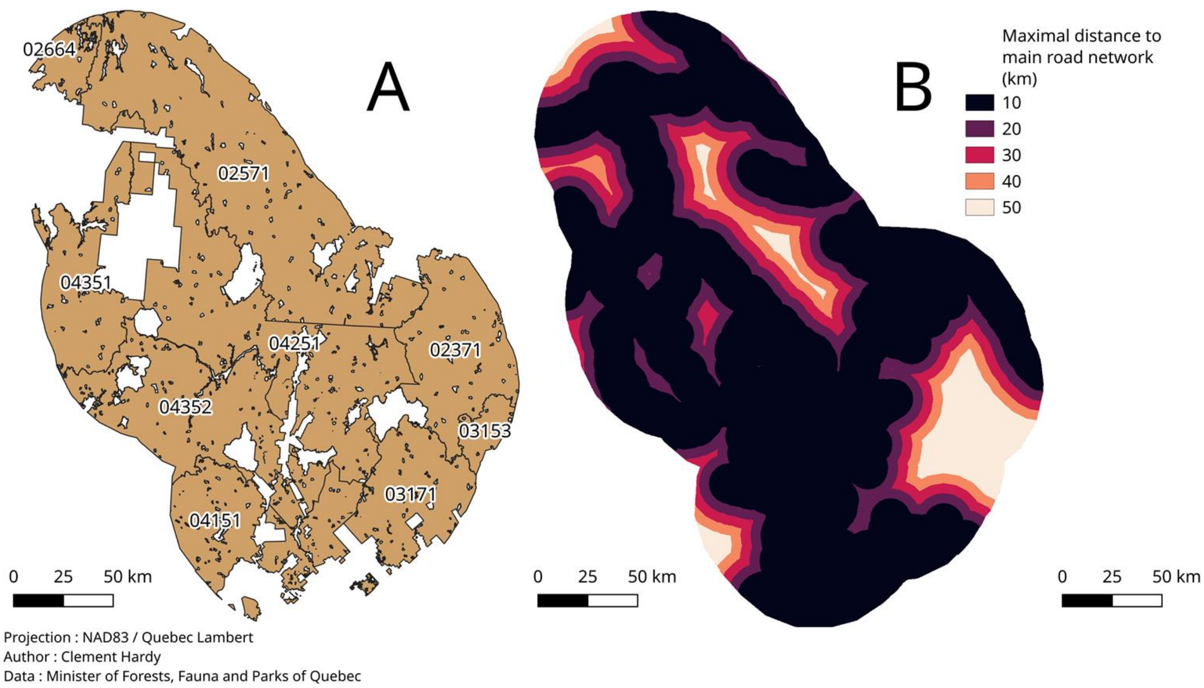

Figure B-5 : Side by side maps of A) the nine management areas present in our simulated landscape; B) the raster indicating the buffer areas corresponding to five categories of distance from the main paved roads: Less than 10 km (red), between 10 and 15 km (orange), between 15 and 20 km (yellow), between 20 and 25 km (light blue), and more than 25 km (dark blue) from the paved roads

B.5.2 Forest stands

The map describing the forest stands across our simulated landscape was simply derived from the forest stands identified by the latest provincial forest inventory carried out by the MRNF of Quebec (MFFP, 2018a).

B.5.3 Prescriptions

We programmed three harvest prescriptions corresponding to the three prescriptions that were mostly used in our simulated area according to the documents of public consultation about the forest management done in the area (e.g., MFFP 2018b). Those prescription included clear cutting with protection of the regeneration and the soil (with the acronym *CPRS* in French); the shelterwood system (with the acronym *CPR* in French); and continuous cover forestry, or irregular shelterwood (with the acronym *CPI-CP* in French).

Usually, prescriptions in LANDIS-II are associated with a stand-ranking method, a stand qualification criteria, and a site selection method inside the harvest stands. However, we bypassed all of these by using a Python script launched by the extension "Magic Harvest" that we developed for LANDIS-II. In essence, the script took care of establishing where in the landscape would the prescriptions be applied, making the use of the stand ranking, qualification and site selection of LANDIS-II not relevant to our purposes. Therefore, the only remaining parameters of importance in each of these prescriptions were related to the cohort removal, which is already given in the main text of the article (see section 2.2.3.4).

For more details about the Python script that we use, see the next section. The complete parameter files for Biomass Harvest, and the Python scripts used are all available in the online repository containing the files used for our simulation (https://figshare.com/s/1e84862cf4114b336a7f).

B.5.4 The Python script used to determine the stands to harvest

The Python script launched by the "Magic Harvest" extension was composed of three big sections :

1. Reading the current state of the landscape

2. Determining the stands to harvest

3. Exporting the management area map that will be read by Biomass Harvest

In the first section, the script detects which scenario is currently ongoing (level of uneven-aged management, aggregation, etc.); it then reads the stand maps, and different information about each stand : their age, their total biomass, their ecoregion, their area, the neighbouring stands, etc., based on the different output raster made by LANDIS-II for the previous time step.

In the second section, the stands to harvest for each prescription are determined one prescription at a time, starting with clearcutting, then shelterwood cutting, and then the uneven-aged prescription (CPI-

CP). For each prescription, stands are selected until the estimated biomass harvested reaches the biomass target for this prescription, starting with stands that have the oldest age cohorts. Only stands older than 30 years (i.e., with age cohorts older than 30 years) are available for harvest. In the case of uneven-aged management, stands are reserved for future uneven-aged cuts every 30 years in the next 90 years once selected, and cannot be used by an even-aged prescription until then. However, the delay of 30 years between uneven-aged cuts can be overridden in cases where the biomass target for the uneven-aged prescription cannot be reached unless it is so. When a stand is selected for a prescription, an algorithm propagates the cut area to surrounding stands until no neighbouring stands are available for further propagation, or until the maximum possible cut area (determined by the level of aggregation for the scenario and the prescription) is reached.

A raster map is then exported in the third section. In every pixel of the map, a code indicates which prescription should be executed in the pixel. The map is then given to Biomass Harvest as a "management areas" map. The "Magic Harvest" extension then forces the Biomass Harvest extension to reload its parameters, and thus read this management area map that was exported. Then, Biomass Harvest is given as instruction to harvest 100% of each management area defined in the map with the prescription corresponding to its code. In effect, this forces Biomass Harvest to harvest the stands determined by the Python script with the prescriptions defined in section B.5.3.

B.5.5 Biomass target to harvest for each management area

In order to set a credible biomass target to harvest in LANDIS-II, we used the documents of public consultation by the Ministry of Forests of Quebec where the allowable volumes of wood to be harvested are determined for five-year periods for each management area of Quebec (e.g., MFFP 2018b). We transformed the volumes of wood (expressed as m3) into biomass units (in tons) for each of the management areas in our simulated area by consulting the documents of public consultation for the period 2018-2023. As each of these documents contained the allowable volume of wood to harvest for several categories of trees (e.g., pine, poplar, birch, etc.), we used wood density data from Gonzalez (1990) to transform wood volumes into biomass. Our protocol was to use the mean values of wood density that were measured in Quebec, or in Canada if values from Quebec were not available for a given species. When several tree species were in a single category, we first calculated a mean wood density value for each of the species, and then a mean of those values for the whole category of species.

Once the allowable volumes to harvest were transformed into allowable biomass targets, we simply summed the biomass targets of each category of tree species into a unique biomass target to harvest for the whole management area. This was done as LANDIS-II did not allow us to consider biomass targets for individual species. A second simplification was done to adapt the biomass target obtained for each management area to the surface of the management area that was present inside our simulated area. To that end, we simply assumed a proportional relation between the biomass target for the management area, and the percentage of the surface of the management area that was in our simulated area. As an example, if a management area had a biomass target of 1 300 000 tons of wood to harvest every time step with 50% of its surface inside our simulated area, then we assumed that the biomass target for this area during our simulation would be 650 000 tons of wood for each time step, or 50% of 1 300 000. The same logic was applied to compute the biomass targets for the smaller management areas that were created when the 5 buffer zones were defined around the main paved roads (see section B.5.1 and Figure B-5).

B.6 Forest Roads Simulation (FRS) extension parameters

B.6.1 Basic parameters

B.6.1.1 Time step

The time step that we use for the FRS module during our simulation was of 10 years, as was the case for the Biomass Harvest module of LANDIS-II that we used. Ideally, the time step of both of these extensions should be the same so that the FRS module is always called just after the harvest module in LANDIS-II.

B.6.1.2 Heuristic

The heuristic that we chose to order the recently harvested cells was the “closest first” heuristic. This is due to the fact that this heuristic is both the most understandable, as it is expected from forestry companies to focus first on the forest stands that are closest to existing routes or sawmills to reduce their operational costs. However, we also did test the heuristics with an alternative version of the model developed for the open source GIS software QGIS that took the form of a plugin called “Forest Roads Network” (Hardy, 2019) . The Forest Roads Network QGIS plugin works in the exact same way as the FRS module for LANDIS-II expect that it is not dynamic in time. We tested the heuristics on five different harvesting sites where forest roads had been built in the forests of Quebec that were identified by the MRNF of Quebec. For each one of these harvesting sites, the “closest first” heuristic had the closest results to the existing road network on those sites in terms of roads generated and the cover of the harvested areas (Figure B-6).

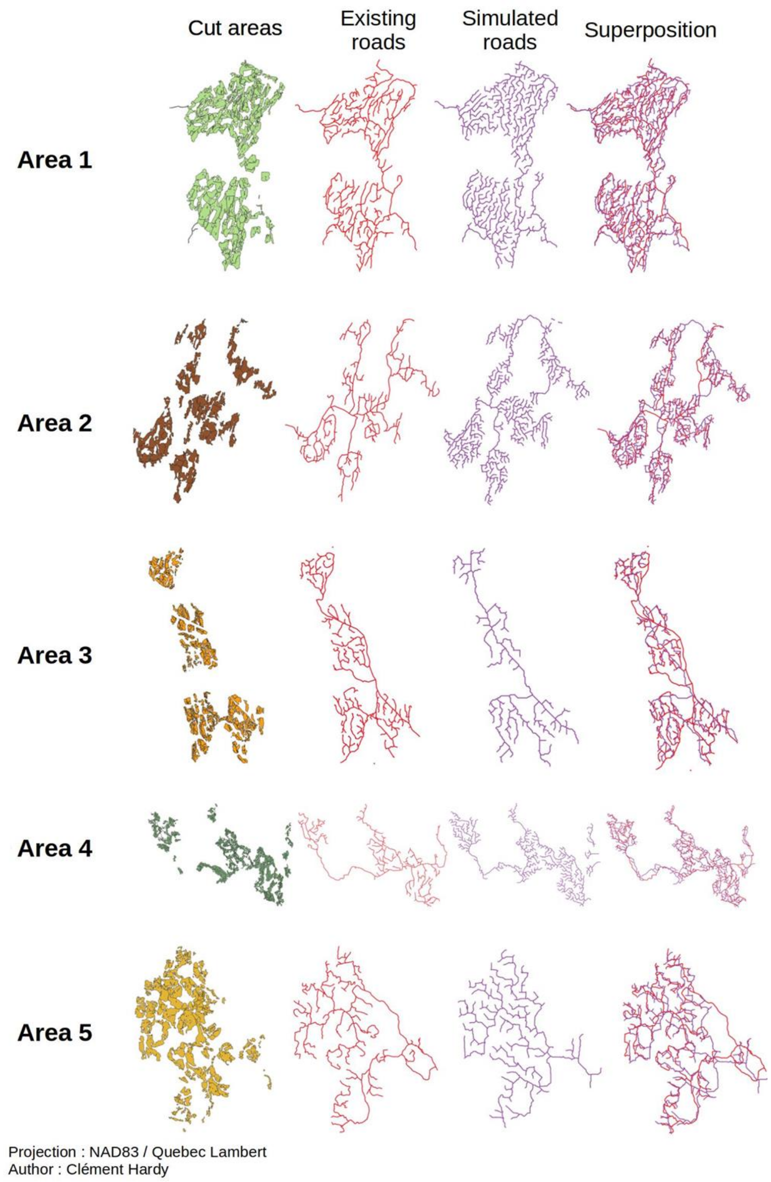

Figure B-6 : Extracts of maps showing the tests of the “Closest first” heuristic that were made with the Forest Roads Network plugin for QGIS. Areas 1-5 correspond to the different cut areas identified by the MRNF for testing, while each column corresponds to the focus of the test: cut areas, existing roads, simulated roads, or superposition.

B.6.1.3 Skidding distance

The skidding distance of 250 m was chosen as it seemed to represent a good compromise between the opinion that experts at the MRNF gave us, and information from that measured skidding distance in cuts located in the boreal forest, that were between 50 m and 150 m.

B.6.1.4 Looping algorithm parameters

The five parameters ruling the functioning of the looping algorithm were determined arbitrarily, and tested in an article introducing the FRS module (Hardy *et al.*, 2023b ; Roa Cea, 2011) so as to replicate the fragmentation of the existing, initial road network. The parameters are presented in Table B-3.

Table B-3 : Parameters used for the looping algorithm of the FRS module during our simulations.

Parameter	**Value**
LoopingBehavior	Yes
LoopingMinDistance	500
LoopingMaxDistance	1200
LoopingMaxPercentageOfRoads	10
LoopingMaxCost	10
LoopingProbability	16

B.6.2 Input rasters and cost parameters

The input rasters and the cost parameters that we used were all derived from public and private data made available by the MRNF of Quebec. Indeed, the basic distance cost, the cost linked to the coarse elevation (slope) and the costs linked to the soils were all derived from information given by the Ministry. As this information took the form of a matrix of potential costs for different categories of forest roads (i.e., primary, secondary, tertiary, etc.), different types of soil and different slopes, we used a linear model based on the data concerning a type of road that we chose as a reference (winter roads according to the nomenclature of the ministries of Quebec) to extract the costs. The linear model had the following form:

Total cost of construction = Basic distance cost + Cost due to soil + Cost due to slope

The adjusted R2 of the model was 0.88, allowing us to believe that it was acceptable to use its coefficients for our parametrization. We used a reference road type as the FRS module uses multiplication values to adapt the cost of construction determined for a cell for the reference type to the other types of roads that are used by the user (see section B.6.3.2).

B.6.2.1 Basic distance cost

The basic distance cost, which corresponded to the intercept of the linear model, was of $674.17 CA for 100 m of winter roads.

B.6.2.2 Coarse elevation costs

The costs due to the slope varied according to several ranges of elevation difference (for the coarse elevation), that are measured in the module by looking at the difference in elevation between cells, and the distance between their centroids. These ranges of elevation for the coarse elevation simply corresponded to the classification used by the MRNF of Quebec in their spatial data, which was also used in the cost matrix from which the linear model was derived. The additional costs due to elevation are presented in Table B-4, with the addition of one last cost for the highest categories of slope that was added to prevent the module from building roads on slopes greater than 41%.

Table B-4 : Additional cost of constructing a road on a cell 100 m in size that is due to the difference in elevation between two cells (slope). Each range of slope is categorized according to classification of the MRNF of Quebec, that equates to a certain percentage of slope.

	Lower elevation threshold (% of slope)	Upper elevation threshold (% of slope)	Additional cost value due to elevation ($CA)
Slope category A and B	0	9	0
Slope category C	9	16	365
Slope category D and E	16	41	778
Slope category F and higher	41	100	100 000

B.6.2.3 Fine elevation costs

The additional cost due to what we called the “fine elevation” varied according to different ranges of values corresponding to the quantity of topographic obstacles in the cells. For example, the fine elevation raster that we used describes how many elevation lines were present in a given cell as the sum of their length (see section B.6.2.10). Thus, we applied the following logic:

- If the length of all elevation lines in the cell is less than or equal to the side of a cell, because our elevations lines represent 10 m of elevation (see section B.6.2.10), that means that there is a high probability that such elevation (10 m) will have to be crossed in the cell when a road is constructed in it. Because of the size of the cells, we consider that this slope is low enough (10%) so that no detour would have to be made.
- If the length of all elevation lines in the cell is equal to more than 3 times the side of a cell, a detour would have to be made to dodge some topographic obstacles as there is a high probability that more than 30 m of slope would have to be dealt with in the cell. This detour would result in a longer road being constructed for the cell, resulting in an increase in the normal cost of construction associated with the cell.
- This detour will be even greater if the length of all elevation lines in the cell is greater than 6 times the side of the cell.
- If the length of all elevation lines in the cell is greater than 10 times the side of the cell, it will then be extremely difficult to cross the cell, resulting in exorbitantly high construction costs for the module.

The result of this logic leads to the Table B-5. While not perfect, we estimate that taking the variation of fine-scale topography inside the cell into account in such a way allows the module to detect and properly avoid areas with cliffs or ravines. Indeed, such obstacles might be too small to be detected from a digital elevation model like the one used to create the “coarse” elevation raster (see section B.6.2.9), but they still could make it impossible for forest roads to be built (Figure B-8).

Table B-5 : Multiplicative values for the costs of construction of a road in a cell that are due to the topographic obstacles (e.g., cliffs or ravines) in the cell. Each range is categorized by the total length of all of the topographic lines present in the cells which indicates a variation of around 10 m of elevation in our case.

Lower threshold of topography (metres of elevation lines in a cell)	Upper threshold of topography (metres of elevation lines in a cell)	Multiplicative value for the costs of construction in the cell
0	300	1
300	600	1.5
600	1000	2
1000	Infinite	10

B.6.2.4 Coarse water cost

The cost related to the coarse water raster corresponds to the price of building a bridge in a cell of 100 m in size was $1 400 000 CA, based on advice of experts at the MRNF. This was imperfect, as these costs can vary with the type of road that is constructed (a larger road necessitating a larger bridge); we thought that it would be enough to influence the path of the roads so that their construction would avoid crossing large bodies of water.

B.6.2.5 Fine water cost

As the cost related to the fine water raster had to correspond to the mean price of installing a culvert, we used the value of $20 000 CA based on advice from experts at the MRNF of Quebec. While imperfect again, we kept this value for the same reasons described in section B.6.2.4.

B.6.2.6 Costs rasters

All of the different rasters that we used for the parametrization of the FRS module were relatively easy to obtain thanks to the public access of many different sets of spatial data made available by the MRNF in Quebec. The most important rasters are shown in Figure B-7.

B.6.2.7 Raster of buildable zones

The raster of buildable zones simply corresponded to the extent of our simulated area as we considered that forest roads could be constructed anywhere within the space as long as the cost was taken into account.

B.6.2.8 Raster of the initial road network

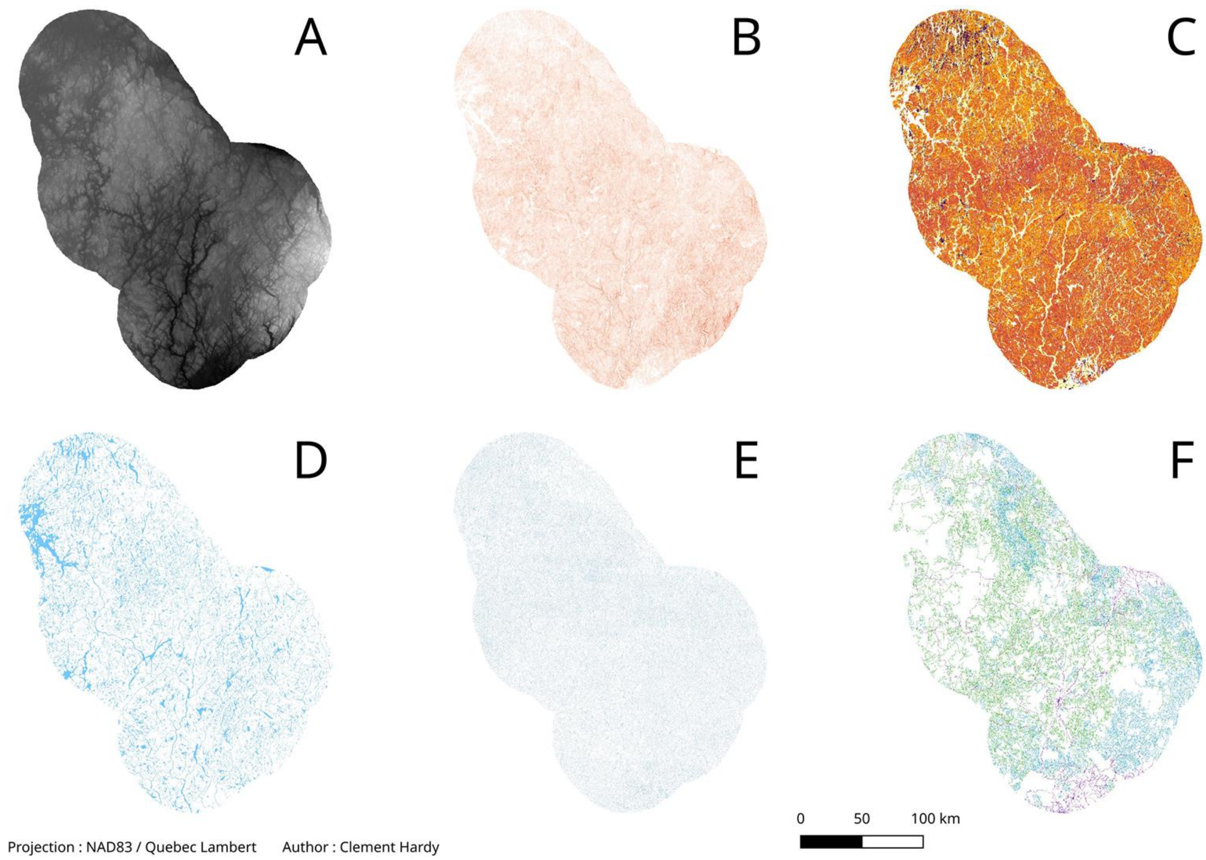

Figure B-7 : Cost rasters used for parameterizing the FRS module: A) The coarse elevation raster (derived from the Digital Elevation Model of Canada; the darker the pixel, the lower the elevation), B) the fine elevation raster (derived from the elevation lines of Quebec; the redder the pixel, the more elevation lines there are), C) the soil cost raster (includes the additional cost of construction due to soil derived from our linear model; the darker the pixel, the higher the cost), D) the coarse water raster (the water bodies in our simulated area), E) the fine water raster (the length of streams in each cell; the darker blue the pixel, the more streams there are), and F) the initial road network raster (main paved roads in purple, primary roads in red, secondary roads in green and tertiary roads in blue).

The initial road network raster (Figure B-7F) contained only the main paved roads within the simulated area, or those and the current existing forest roads according to the presence or absence of an initial road network in the scenario. All of the roads were taken from the AQReseau+ database of Quebec, which compiles all of the existing main paved roads as well as the forest roads from the ROUTARD database which is a compilation of the existing forest roads in Quebec from different sets of data.

B.6.2.9 Coarse elevation raster

The coarse elevation raster contained the data of the Canadian Digital Elevation Model for 1945-2011, edited by Natural Resources Canada.

B.6.2.10 Fine elevation raster

The fine elevation raster was made using a dataset containing elevation lines corresponding to 10 m differences in elevation for all of Quebec, made by the Ministry of Energy and Natural Resources of Quebec. This dataset was then converted into a raster containing values corresponding to the total length of the elevation lines in each cell using the "Sum Line Length" tool of the QGIS software.

B.6.2.11 Coarse water raster

The coarse water raster corresponded to all of the polygons of the 5th national forest inventory of Quebec that were classified as a body of water (via the CO_TER attribute of the polygons).

B.6.2.12 Fine water raster

The fine water raster was made using a vector data set from the Ministry of Energy and Natural Resources (MERN) of Quebec containing the predicted location of all streams in Quebec based on topography. In a manner similar to the fine elevation raster, the vector data set of the streams was transformed into a raster containing the length of all the streams crossing a cell via the "Sum Line Length" tool of QGIS.

The module then automatically deduced an average number of culverts that would have to be installed to cross the cell according to the size of the cell and the length of the streams present. To that end, the length of the streams in the cell was divided by the length of the diagonal of the cell, and the result was rounded off to obtain the number of culverts that would have to be installed. For example, if the cell contains 350 m of streams and is 100 m × 100 m large, then 2 culverts would be required to cross the cell.

B.6.2.13 Soil costs raster

The raster containing the additional costs due to the soil was made by combining the coefficients of our linear model for every type of soil with the types of soil identified for every polygon of the 5th national forest inventory of Quebec (DEP_SUR attribute). Using the QGIS attribute calculator for vector layers, we were able to associate the appropriate cost to every polygon, and to transform those polygons into a raster layer where each cell contained the value of the additional cost for the soil.

B.6.2.14 Resulting cost raster

The combination of all of the cost rasters and of the cost parameters results in a “cost map”, produced during the initialization of the module (Figure B-8). This cost map allowed us to check for artifacts or outlier values that might influence the functioning of the module. We found no such artifacts or outliers as the cost map correctly represented streams, cliffs, bodies of waters and variations in soil and elevation.

B.6.3 Road thresholds and multiplication values

B.6.3.1 Road aging, wood flux, and types of roads

As we decided to simulate road aging and the influence of the flux of harvested wood going through the forest road network on the categories of the forest roads, we had to parameterize the FRS module accordingly.

In the end, we decided to use 9 different types of roads, based on the ones existing in our landscape, and on the classification used by the different ministries of Quebec that deal with the roads of the province. 7 Of these 9 types of roads, 7 were considered forest roads that could be built by the FRS module, including winter roads, roads of class 5 to class 1, and “out of norms” roads (Gouvernement du Québec, 2015). We added two types that were considered as exit points: sawmills and the main, paved road network. As no sawmills were identified in our landscape, only the main paved road network was used.

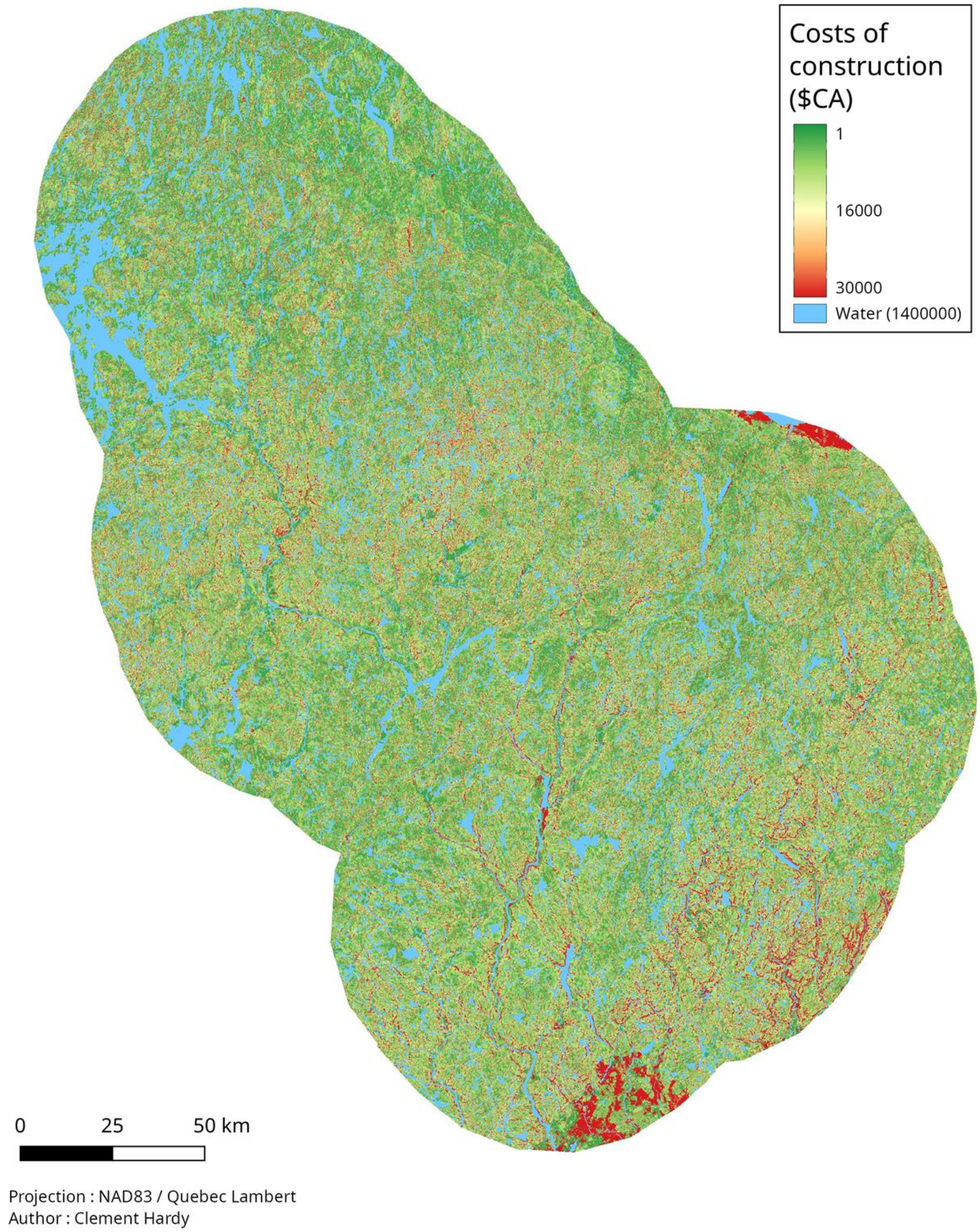

Figure B-8 : Cost map of our simulated area generated by the FRS module during its initialization phase, and based on the cost rasters and the cost parameters that we provided. The range in the costs of construction within each pixel is indicated by the variation in colour.

B.6.3.2 Multiplicative cost values

For each of the 7 types of forest roads, a multiplicative cost value had to be given. This value is used by the module when a road must be upgraded to a higher road type (as forest roads are always initially constructed with the lowest road type by the FRS module, and then upgraded if needed according to wood fluxes) so that the correct cost of an upgrade can be calculated by the FRS module.

We determined the multiplicative cost value for each type of forest road using the table of costs from which we derived our linear model (see section B.6.2). For each type of road, we calculated the mean cost of construction at the scale of the side of a cell for every type of soil and slope. We then divided this mean cost of construction by the mean cost of construction of the reference type of road that we used when creating the linear model, which was winter roads. The result of this division gave us the multiplicative cost values for a given category of road; those values ranged from 1 for winter roads (our reference type) to 15.06 for "out of norm" roads (Table B-6).

B.6.3.3 Maximum age before destruction

The maximum age for each road type before their destruction was determined from documentation from the MRNF of Quebec (MFFP, 2020). No age had to be given to road types that were not constructed by the module (i.e., the sawmill and main paved road type), as those were not affected by senescence in the current functioning of the FRS module.

B.6.3.4 Wood flux thresholds

For each of the 7 types of forest roads, two thresholds had to be given to express the wood flux as a number of age cohorts harvested. Later versions of the FRS module are expected to allow this number to be expressed in biomass harvested. However, whether expressed in biomass or in another unit of harvested timber, we found no information for linking the quantity of wood flux on a forest road to its upgrade to a larger road type. We expect that such sources are non-existent as the decision-making for such a process is complex, dependent on the country and whether the forest is publicly owned or privately owned, as well as the fact that forest road networks are often planned many years in advance.

To compensate for this lack of information, we decided to use an empirical parametrization based on the current proportions of the different categories of roads in our landscape. To that end, we launched a calibration simulation of our landscape lasting 150 years, with harvesting planned for the period 2018-2023, and used the same parametrization of the base fire module that we used in our main simulations. The FRS module was parameterized with all of the parameters that we described above, except for the wood flux thresholds which were arbitrary. We then gathered the values of the wood flux on the roads during the simulation that were greater than 0 for each road cell in the landscape for the entire simulation. Then, we visualized the distribution of the values of wood flux on a histogram, and compared it to a histogram showing the frequencies of the road types in the initial road landscape when all forest roads were taken into account. Consequently, we tweaked the threshold of the bins of the first histogram to obtain the same proportions by column as for the second histogram (Figure B-9). This way, the wood flux thresholds for each road type were made so that under current management conditions, a simulation would produce the same proportion of each type of road as the proportion existing in the initial road landscape.

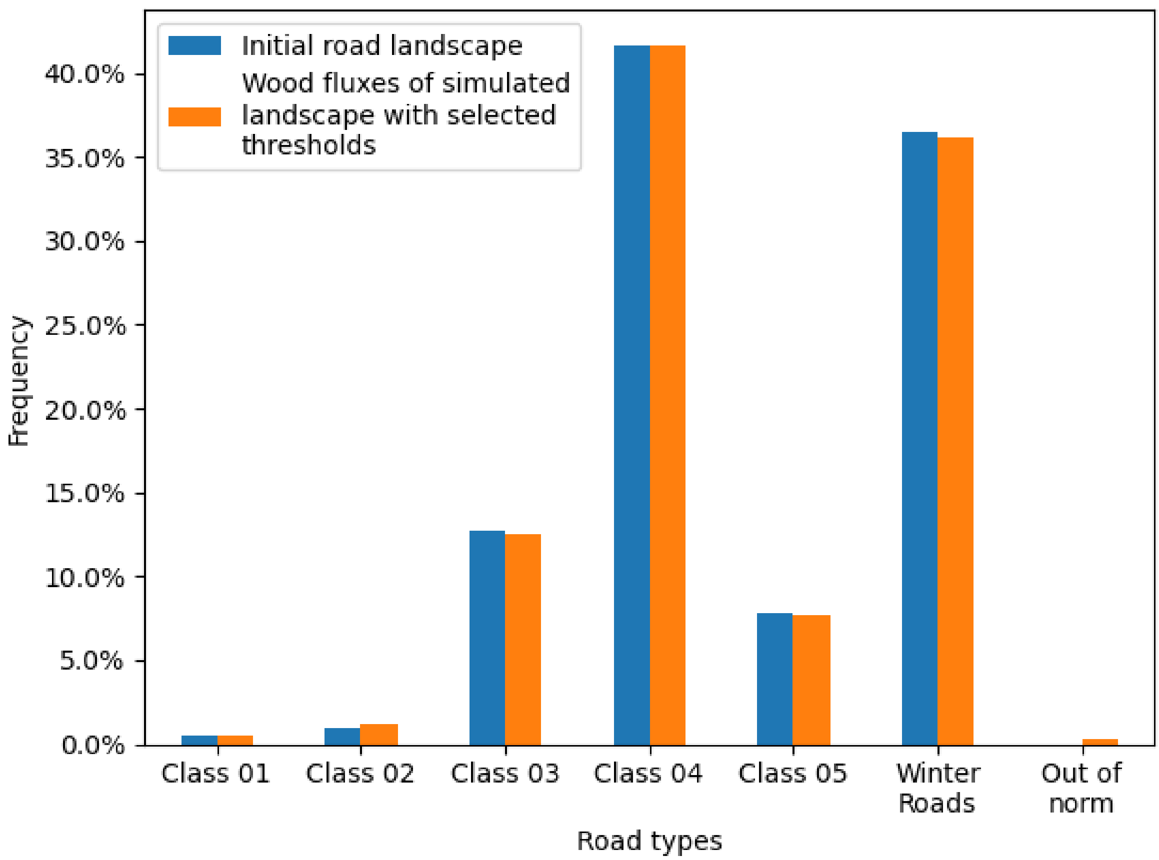

Figure B-9 : Histogram showing the seven road types that we used and the difference in frequency between the roads present in the initial road landscape (all forest roads included), and the transformation of a set of wood flux values into the road types, according to the thresholds of wood flux for each road type that we empirically determined. The wood flux was measured at each time step during a 150-year simulation with current forest management conditions.

B.6.3.5 Resulting parameters

The values resulting from our parameterization of the road types and wood flux are presented in Table B-6.

Table B-6 : Parameters used in our simulations to classify the roads into different types in the FRS module, how much they cost, and how long these different types of roads lasted during the simulations.

Name of the road type	Lower wood flux threshold	Upper wood flux threshold	Road type ID	Multiplicative cost value	Maximum age before destruction
Winter roads	0	70	6	1	1
Class 05	70	105	5	1.5	3
Class 04	105	3000	4	2.15	10
Class 03	3000	40000	3	3.57	15
Class 02	40000	70000	2	9.19	25
Class 01	70000	100000	1	13.38	25
Out of norms	100000	Inf.	7	15.06	50

B.7 Manual correction of the initial construction costs in scenarios with an initial road network

The current functioning of the FRS extension led to an artifact in our results, in the scenarios with the presence of an initial road network. This artifact comes from the fact that when the FRS extension creates a new road to reach a newly harvested area, its search algorithm stops at existing roads; then, the wood harvested in the cell is considered to be transported throughout the rest of the existing road network toward an exit point (Hardy *et al.*, 2023b). During this process, every road in the existing road network is upgraded to fit the transportation of wood, with roads with more passage becoming upgraded to larger roads.

However, in the initial road network, many of the existing roads being very near bodies of water (e.g., rivers), they would be considered as bridges by the model as their location superimposed with locations on the coarse water raster (see section B.6.2.11), as is shown in Figure B-10. In these cases, when enough wood was transported through these roads, they were upgraded to larger roads; but as they were considered as bridges, the upgrades were extremely costly. This resulted in an initial spike in road costs in scenarios with an initial road network that was an order of magnitude superior to the road costs in the rest of the simulation.

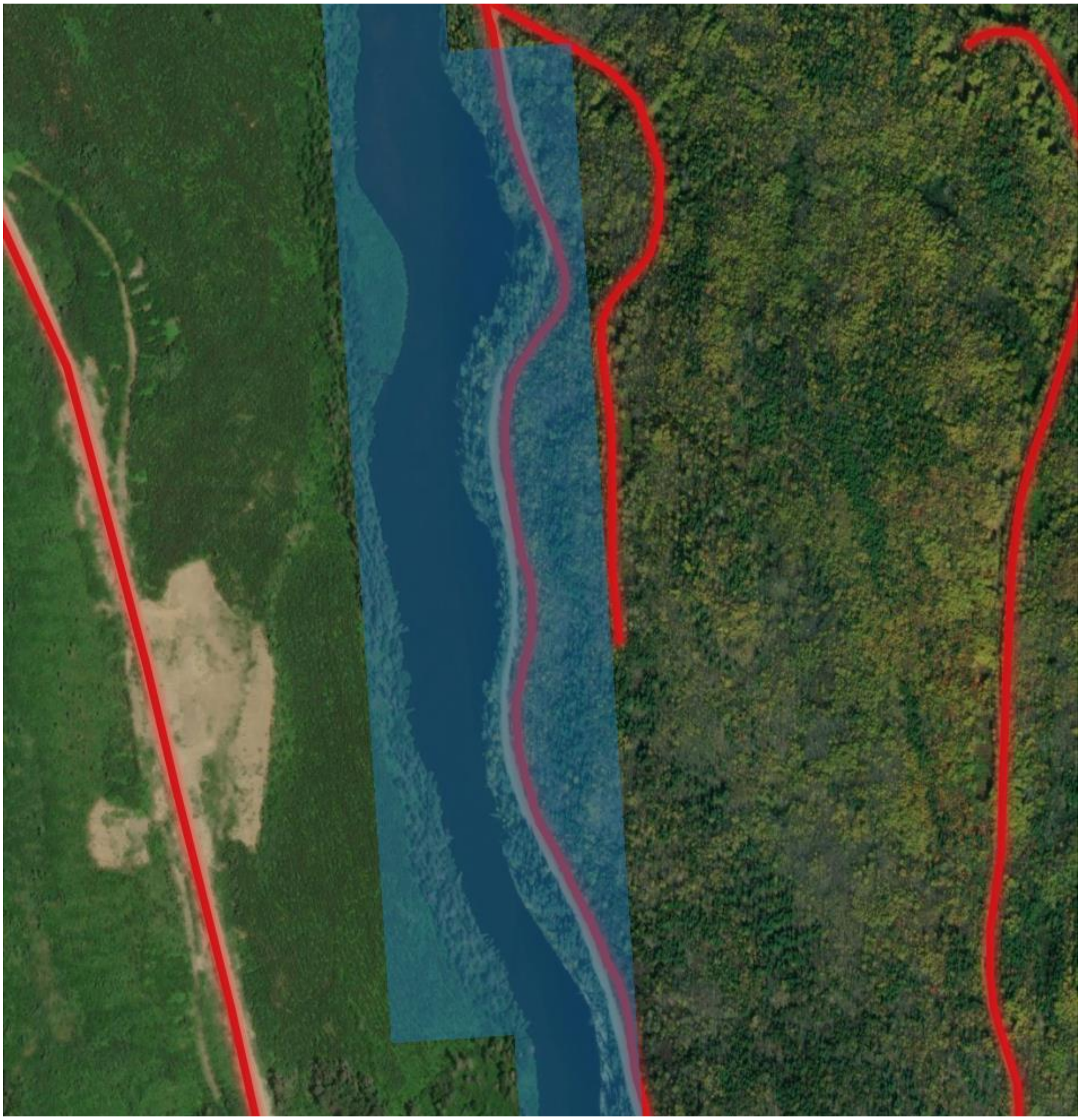

Figure B-10 : Aerial photography from Bing Maps superimposed with our initial road data (red lines) and coarse water raster (blue pixels; see section F.2.11).

To fix this artifact, we computed the road costs manually through a python Script. In the case of the first time step for scenarios with an initial road network, and in the case of road upgrades, we checked if this upgrade concerned a bridge. When that was the case, we used a raster that indicated the position of real bridges (created by finding roads that intersected with a shapefile layer for the bodies of water) to check if the bridge did exist in reality. If it did not, the cost of upgrade was computed as if the road was not a bridge, solving the problem. The consequent results obtained with this method are shown in the main article.

B.8 Surface harvested, biomass harvested and surface burned

The measures of the surface harvested (Figure B-11) and the biomass harvested (Figure B-12) using the “Biomass Harvest” module during our simulations, as well as the surface burned using the “Base Fire” module (Figure B-13) are displayed below.

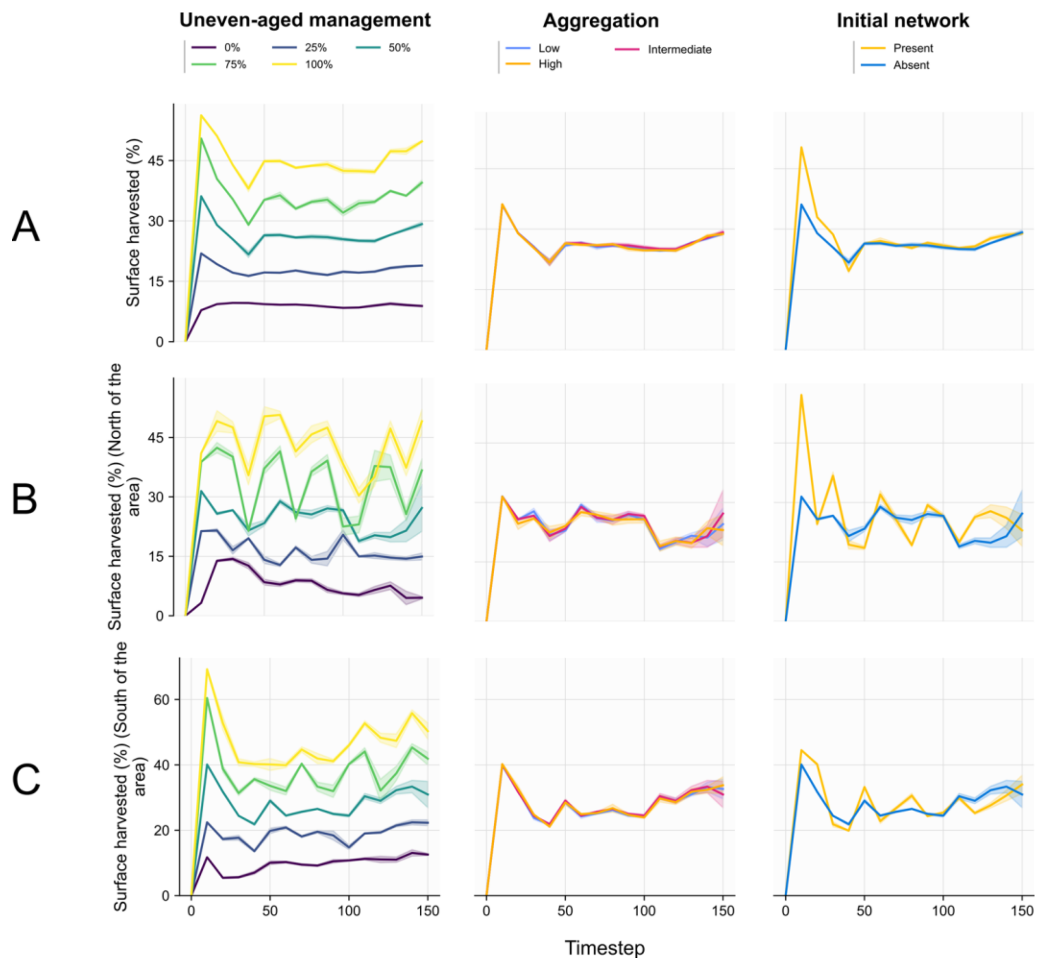

Figure B-11 : Time dynamics of the surface harvested using the “Biomass Harvest” module during our simulation for A) the entire study area, B) the northern region of our study area, and C) the southern region of our study area. Each factor is varied while keeping the other factors at their intermediate value. Curves correspond to the average over 5 simulation runs, and shaded areas correspond to the standard deviation between replicates.

As can be seen in Figure B-11, there are periodic variations in surface harvested by the module that are due to the repetition of uneven-aged prescription in the same stands every 30 years. As stands are "reserved" for the prescription once they start being harvested with it (see section B.5.3), and because stands that are older and contain more biomass are reserved in priority, this creates a sometimes periodic pattern in the surface harvested as older stands are harvested at some time steps, and younger stands at others.

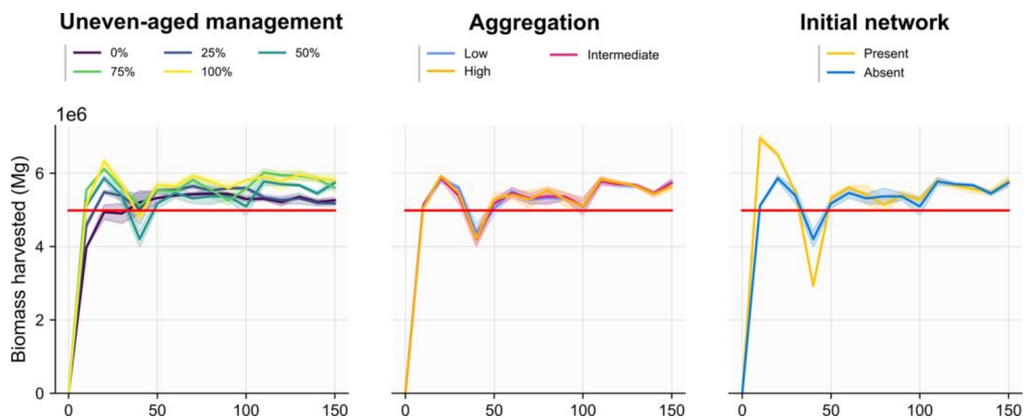

Figure B-12 : Time dynamics of the biomass harvested using the “Biomass Harvest” module during our simulation for the entire study area. The red line corresponds to the biomass target derived from the public consultation documents for the different management areas of our landscape, determined by the MRNF of Quebec for the period 2018-2023 and extended to a 10-year time step. Each factor is varied while keeping the other factors at their intermediate value. Curves correspond to the average over 5 simulation runs, and shaded areas correspond to the standard deviation between replicates.

Figure B-12 shows that the biomass harvested is relatively constant and near the target, although small variations in time and between scenarios do exist. The variations in time can be seen in almost any scenario as the initial spike in biomass harvested at time step 20, followed by a drop at time step 40. These are due to the way that the Python script activated by magic harvest reads the state of the landscape; the script only knows the amount of total tree biomass in a stand, but doesn't know how this biomass is partitioned between the different age cohorts (see section B.5.5). As most prescriptions do not harvest the younger age cohorts, the Python script can over- or under-estimate the biomass that will be harvested in a stand according to how this partitioning is in the stands of the landscape at different time steps, resulting in these variations. In the case of the scenarios with an initial network (Figure B-12, right), this artifact of our methodology is made even more flagrant as the model starts harvesting the whole of the landscape at the first time step, while the other scenario progressively harvest more and more of the landscape on the first 50 years of the simulation (see section B.5.1).

In Figure B-13, it is apparent that the northern region displays a much higher surface burned than the southern region (around 10 times higher), which makes sense given the ecological reality of our study area: the northern region is composed of boreal forests that have a more intensive fire regime than the southern region which is composed of mixed forests.

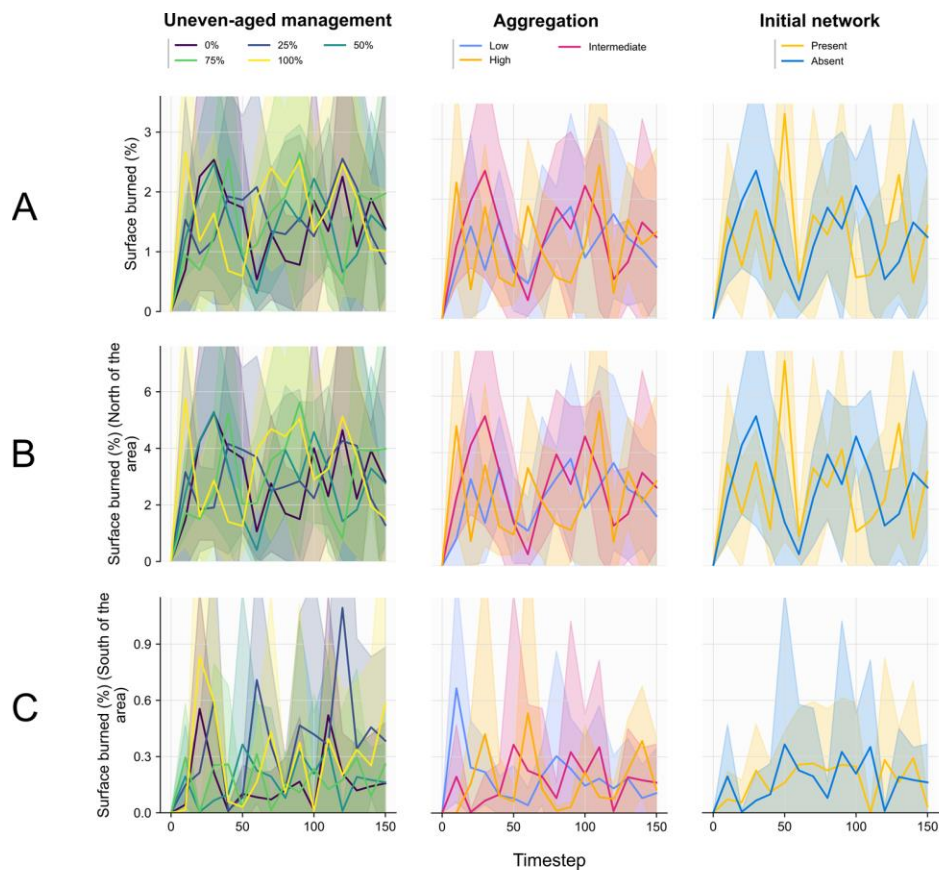

Figure B-13 : Time dynamics of the surface burned using the “Base Fire” module during our simulation for A) the entire study area, B) the northern region of our study area, and C) the southern region of our study area. Each factor is varied while keeping the other factors at their intermediate value. Curves correspond to the average over 5 simulation runs, and shaded areas correspond to the standard deviation between replicates

In every scenario and across replicates, forest fires tend to show a very high variability in the amount of surface burned at every time step, which is coherent with the way that the Base Fire extension functions (i.e., via ignition events and concurrent spreading; see main article). We believe that this variability is captured sufficiently with the use of 5 replicates simulations for each of our measures.

B.9 Changes in old-growth forest amount and fragmentation for the whole study area

The variation in the amount and fragmentation of the old-growth forest for the three factors that were varied in every simulation scenario over the whole study area is shown in Figure B-14. The tendency observed for the curves with a varying level of use of uneven-aged management and those related to different levels of aggregation are already described in detail in the main article.

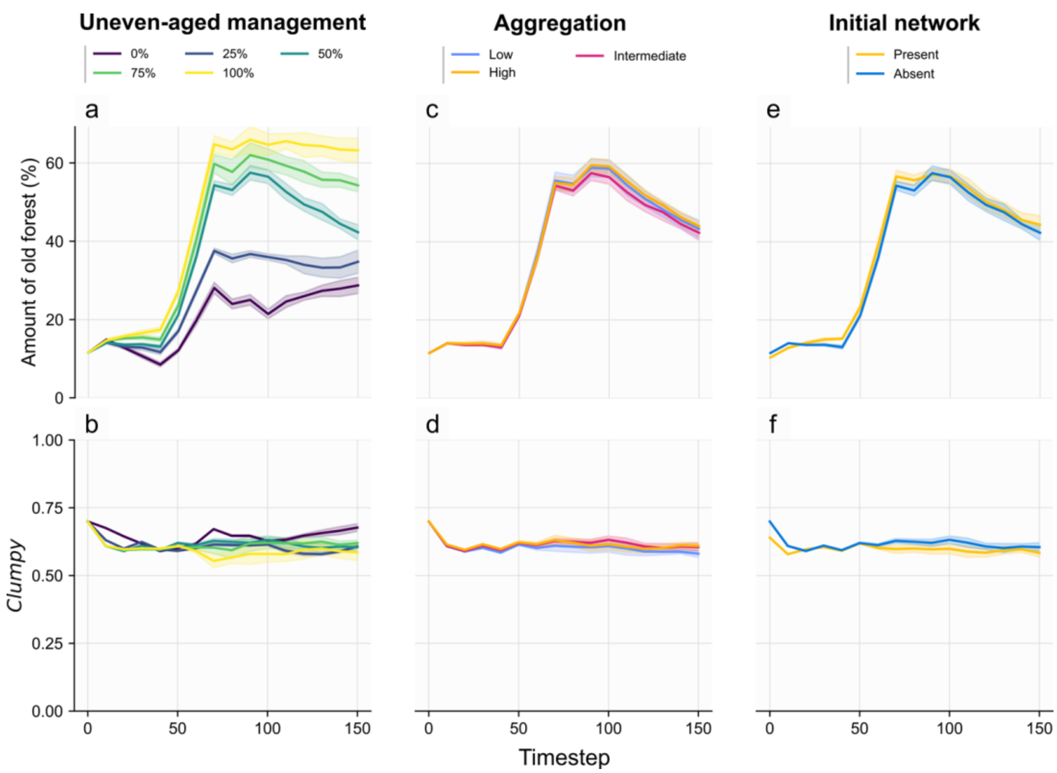

Figure B-14 : Time dynamics of the amount of old-growth forest (with age cohorts > 91 years) and the fragmentation index Clumpy over the whole study area for each investigated factor: percentage of the biomass target harvested using uneven-aged management (a, b), level of aggregation of harvested areas (c, d), and presence of an initial forest road network (e, f). Each factor is varied while keeping the other factors at their intermediate value. Curves correspond to the average over 5 simulation runs, and shaded areas correspond to the standard deviation between replicates.

B.10 Effect of aggregation on road density for scenarios without uneven-aged management

As said in the main article, no effect of aggregation on road density was observable when the reference level for uneven-aged management was fixed at 50%, as is the case for the figures of the main article. However, such an effect was visible if the reference level was fixed at 0% of uneven-aged management, as shown in Figure B-15. Hence, in scenarios with 0% if uneven-aged management, high levels of aggregation were able to reduce the quantity of roads in the landscape by about 10% during most of the simulation.

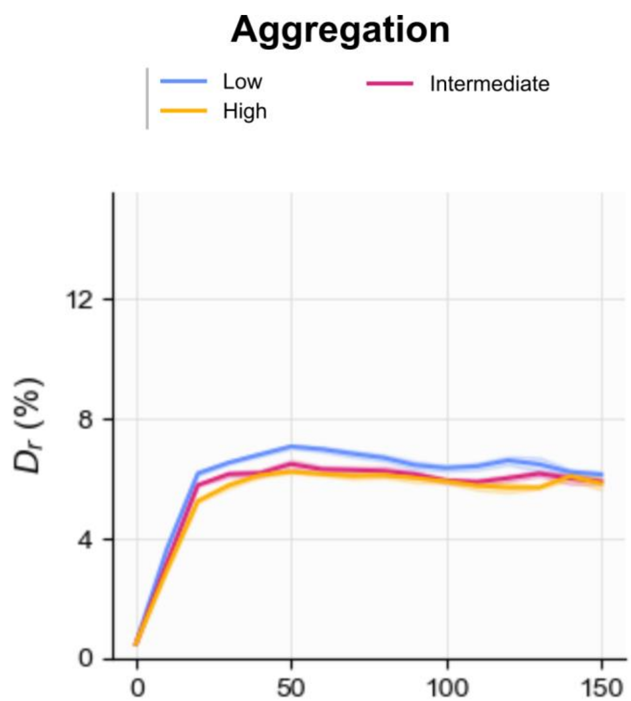

Figure B-15 : Effect of the aggregation levels on the density of roads in the landscape for scenarios with no uneven-aged management, and only even-aged management.

B.11 Results related to the productivity and age of the cohorts in the landscape scale

The variation of the mean age of all of the tree cohorts in the landscape (A) and the mean annual net primary productivity (P) of all of the cells of the landscape, for the three factors that were varied in every simulation scenario are shown in Figure B-16. While the aggregation of the cuts and the presence or absence of an initial network had no effect on either measure (Figure B-16c,d,e,f), uneven-aged management resulted in older age cohorts (Figure B-16a) and in a slightly less productive landscape at the end of the scenario (Figure B-16b). However, variations in the dynamic of P with varying levels of uneven-aged management are complex. We expect that these variations are due to the fact that in the LANDIS-II simulations, the productivity of the landscape increased as the age of the cohorts of the landscape (represented by A) decreased. Hence, A reached an inflection point around 100-120 years of simulation, after which it started to decrease (Figure B-16a) while at the same time, P started to increase. As the inflection point of A for scenarios with uneven-aged management happens later than scenarios with more even-aged management, uneven-aged management resulted in higher values of P at the beginning of the scenario; however, uneven-aged management resulted in lower values of P at the end of the scenarios, as scenarios with more even-aged management reached the inflection point earlier, and thus took the lead in the value of P (Figure B-16b).

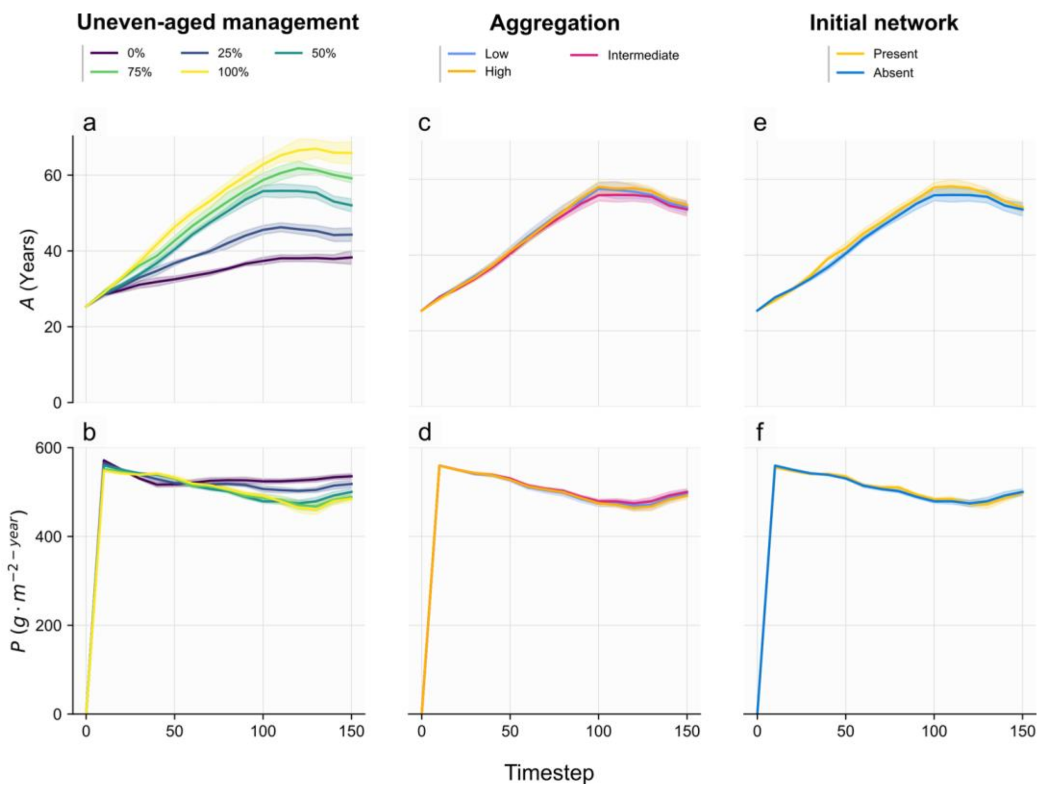

Figure B-16 : Time dynamics of the mean age of the age cohort and of the mean annual primary productivity of the forest in our study area for each investigated factor: percentage of the biomass target harvested using uneven-aged management (a-c), level of aggregation of harvested areas (d-f), percentage of the biomass target (g-i), and presence of an initial forest road network (j-l). Each factor is varied while keeping the other factors at their intermediate value. Curves correspond to the average over 5 simulation runs, and shaded areas correspond to the standard deviation.

B.12 Formula of the Clumpy fragmentation index

Clumpy is the name of the Clumpiness Index (McGarigal *et al.*, 2012), whose formula for the patch type (class) i is given in the following equation:

$$\text{Given}G_i = \left(\frac{g_{ii}}{\left(\sum_{k=1}^{m} g_{ik}\right) - \text{min-e}_i}\right)$$

$$CLUMPY_i = \left[\begin{array}{c} \frac{G_i - P_i}{P_i} for G_i < P_i P_i < 0.5; else \\ \frac{G_i - P_i}{1 - P_i} \end{array} \right]$$

With the terms:

- g_{ii} being the number of like adjacencies between pixels of patch type (class) i based on the double-count method.
- g_{ik} being the number of adjacencies between pixels of patch types (classes) i and k based on the double-count method. Here, k are the other existing patch types (class) in the landscape.
- $\min - e_i$ being the minimum perimeter (in number of cell surfaces) of patch type (class) i for a maximally clumped class
- P_i being the proportion of the landscape occupied by patch type (class) i

APPENDICE C

Supplementary Material for : Climate is stronger than you think: Exploring functional planting and TRIAD zoning for increased forest resilience to extreme disturbances

C.1 Species used and their life history traits

The species and their associated life-history traits that we used are the same as those in the study by Tremblay et al. (2018) and are detailed in Table 1, except for two hybrid species that we added in our simulation. These parameters were derived from various sources (e.g., Burns *et al.* 1990 ; Farrar 1995) and from expert judgment when empirical sources did not exist (Tremblay *et al.*, 2018). The parameters of the two hybrid species (the hybrid poplar POPU.HYB and hybrid larch LARI.HYB) were copied from their non-hybrid counterparts (POPY.TRE and LARI.LAR respectively), but were given a smaller longevity and were incapable of dispersing by themselves. This was done based on expert opinion to imitate the fast-growing nature of both hybrids, and the fact that their dispersion would only be done through human planting and heavy fertilization.

Tableau C-1 : Life-history traits parameters of the 19 tree species used in our simulations.

Real name	Code name	Longevity (year)	Age of sexual maturity	Shade tolerance	Fire tolerance	Seed dispersal distance		Vegetative reproduction probability	Sprout age		Post fire regeneration
						Effective	Max		Min	Max	
Abies balsamea	ABIE.BAL	150	30	5	1	25	160	0	0	0	none
Acer rubrum	ACER.RUB	150	10	3	2	100	200	0.5	10	100	resprout
Acer saccharum	ACER.SAH	300	40	5	2	100	200	0.1	10	60	resprout
Betula alleghaniensis	BETU.ALL	300	40	3	1	100	400	0.1	10	180	resprout
Betula papyrifera	BETU.PAP	150	20	2	1	200	5000	0.5	10	70	resprout
Fagus grandifolia	FAGU.GRA	250	40	5	1	30	3000	0.5	10	30	none
Larix laricina	LARI.LAR	150	40	1	1	50	200	0	0	0	none
Larix x marschlinsii Coaz, (hybrid)	LARI.HYB	80	80	1	1	1	2	0	0	0	none

Real name	Code name	Longevity (year)	Age of sexual maturity	Shade tolerance	Fire tolerance	Seed dispersal distance		Vegetative reproduction probability	Sprout age		Post fire regeneration
						Effective	Max		Min	Max	
Picea glauca	PICE.GLA	200	30	3	2	100	303	0	0	0	none
Picea mariana	PICE.MAR	200	30	4	2	80	200	0	0	0	serotiny
Picea rubens	PICE.RUB	300	30	4	1	100	303	0	0	0	none
Pinus banksiana	PINU.BAN	150	20	1	2	30	100	0	0	0	serotiny
Pinus resinosa	PINU.RES	200	40	2	3	12	275	0	0	0	none
Pinus strobus	PINU.STR	300	20	3	3	100	250	0	0	0	none
Populus tremuloides	POPU.TRE	150	20	1	2	1000	5000	0.9	10	150	resprout
Populus spp. (hybrid)	POPU.HYB	60	60	1	2	1	2	0.9	10	60	resprout
Quercus rubra	QUER.RUB	250	30	3	3	30	3000	0.75	20	200	resprout
Thuja occidentalis	THUJ.SPP.ALL	300	30	5	1	45	60	0.1	10	60	none
Tsuga canadensis	TSUG.CAN	300	60	5	1	30	100	0	0	0	none

C.2 Details of the prescriptions implemented in Biomass Harvest

We defined 6 types of harvest prescriptions to use with the Biomass Harvest extension of LANDIS-II. Here, we show how they were implemented in the format of the parameters for Biomass Harvest, along with the details of their spatial and temporal distribution controlled through the Magic Harvest extension.

C.2.1 Control of each prescriptions spatial distribution through the python script

Each prescription was ultimately implemented at each time step through the use of the Magic Harvest extension that we created (Hardy, 2022) and a python script called by Magic Harvest. Therefore, all prescriptions had a similar stand ranking, stand qualification and site selection parameters in the format of Biomass Harvest (see below), but these 3 parameters are not representative of how the prescriptions were distributed. Indeed, these parameters were only chosen so that the python script could control in which pixels each prescription would precisely be used. We did this because the Biomass Harvest extension of LANDIS-II was incapable of planting several species in a given forest stand. Controlling prescriptions through the python script thus allowed us to apply different prescriptions in a single stand to plant different species in the different pixels of the stand, which would be necessary for our use of functional planting.

As such, the python script created a raster map of prescriptions at each time step, where numbers 1-19 indicated that a particular prescription be used in this pixel and 0 indicated no prescription. This raster was then indicated as both a management area raster and a stand raster to Biomass Harvest, which was then forced to reload its parameters. Then, Biomass Succession would harvest 100% of the pixels indicated with the prescription corresponding to the number in the pixels. Here, the stand ranking, stand qualification and site selection parameters that we chose for all of our prescriptions insured that all of the pixels with numbers 1-19 in the raster map would be harvested without any exception.

For all prescriptions, the python script selected stands by prioritizing them according to a criterion (e.g., those with the most biomass first, or those with the lowest functional diversity first). All prescriptions were applicable only to stands with at least one age cohort of 30 years or older. When a stand was selected for harvesting, the biomass that would be harvested by Biomass Harvest was estimated by a function in the python script, based on the content of the stand. The estimated harvested biomass were summed as stands were selected to be harvested, and the selection stopped when the sum reached the biomass target to harvest with the given prescription for a time step. As such, the biomass harvested by each prescription remained constant through time.

C.2.2 Clear cuts (CC-PlantFunct, CC-PlantIntens, CC-NormalPlant and CC-NoPlant)

Clear cuts removed all or almost all of the trees presents in the stand, and could be followed by the plantation of one or two species in the cell.

C.2.2.1 CC-PlantFunct

CC-PlantFunct was the type of clear-cuts used for functional planting. It removed 90% of the biomass of all tree cohorts older than 10 years. As such, it did not remove any age cohort in the pixels of stand, which kept trees of the existing species in place in the pixel. Then, it planted a new age cohort for a given species (see below for the choice of the species). Therefore, we created 17 variation of CC-PlantFunct in Biomass Harvest, one for each species to plant (except the hybrid species, which were not considered useful to the goals of functional planting).

The 17 CC-PlantFunct prescriptions took the following form in the parameters of Biomass Harvest:

```
Prescription CC-PlantFunct-SPECIES_TO_PLANT

>> STAND RANKING:
StandRanking MaxCohortAge
>> STAND QUALIFICATION FOR CUTTING:
>> None.
>> SITE SELECTION:
SiteSelection Complete
>> COHORT REMOVAL METHOD:
CohortsRemoved SpeciesList
>> Species      Cohorts removed
>> -------      ---------------
   ABIE.BAL    11-999 (90%)
   ACER.RUB    11-999 (90%)
   ACER.SAH    11-999 (90%)
   BETU.ALL    11-999 (90%)
   BETU.PAP    11-999 (90%)
   FAGU.GRA    11-999 (90%)
   LARI.LAR    11-999 (90%)
   LARI.HYB    11-999 (90%)
   PICE.GLA    11-999 (90%)
   PICE.MAR    11-999 (90%)
   PICE.RUB    11-999 (90%)
   PINU.BAN    11-999 (90%)
   PINU.RES    11-999 (90%)
   PINU.STR    11-999 (90%)
   POPU.TRE    11-999 (90%)
   POPU.HYB    11-999 (90%)
   QUER.RUB    11-999 (90%)
   THUJ.SPP.ALL     11-999 (90%)
   TSUG.CAN    11-999 (90%)
>> PLANTING:
Plant SPECIES_TO_PLANT
```

The stands affected by CC-PlantFunct were chosen as follows: the functional diversity of every harvestable stands (i.e., not in a protected area) was computed at each time step in the python script that controlled the prescriptions. Then, stands were selected by prioritizing those with the lowest functional diversity. The selected stands had to be located in the extensive zones of the TRIAD zoning system if the prescription was used during a TRIAD scenario.

Once stands were selected, the python script identified which species to plant inside of it for functional planting purposes. To that end, the script identified which functional groups were missing in the stand (i.e., no age cohorts of the species of these groups were present in the cell). If all functional groups were present, the script identified the rarest present group. For each missing functional group (or for the rarest), the script then identified the species that had the highest Probability of Establishment parameter from the Biomass Succession extension for the ecoregion in which the stand was. This ensured that species as adapted as possible to the ecoregion were to be planted. Then, the script randomly applied a CC-PlantFunct prescription for one of the species to plant in one of the pixels of the stand. In this way, the species to plant were randomly planted in the pixels of the stand in equal proportion pixel-wise.

C.2.2.2 CC-PlantIntens

CC-PlantIntens was used to remove all of the present trees in a given stand, and replace them with an even-aged mixed plantation of a hybrid species (hybrid poplar or hybrid larch, depending on the ecoregion) and a commercial species, the white spruce. We therefore created two variations of the prescription for the two hybrid species.

CC-PlantIntens took the following form in the parameters of Biomass Harvest:

```
Prescription CC-PlantIntens-HYBRID_SPECIES/PICE.GLA

>> STAND RANKING:
StandRanking MaxCohortAge
>> STAND QUALIFICATION FOR CUTTING:
>> None.
>> SITE SELECTION:
SiteSelection Complete
>> COHORT REMOVAL METHOD:
CohortsRemoved SpeciesList
>> Species      Cohorts removed
>> -------      ---------------
   ABIE.BAL    All
   ACER.RUB    All
   ACER.SAH    All
   BETU.ALL    All
   BETU.PAP    All
   FAGU.GRA    All
   LARI.LAR    All
   LARI.HYB    All
   PICE.GLA    All
   PICE.MAR    All
   PICE.RUB    All
   PINU.BAN    All
   PINU.RES    All
   PINU.STR    All
   POPU.TRE    All
   POPU.HYB    All
   QUER.RUB    All
   THUJ.SPP.ALL     All
   TSUG.CAN    All
>> PLANTING:
Plant HYBRID_SPECIES PICE.GLA
```

CC-PlantIntens was only used in the intensive zones of the TRIAD scenarios, in order to simulate an intensive form of forestry focused on wood production alone as is typical of the TRIAD zoning system.

Stands to harvest were selected by prioritizing those with the largest amount of biomass. CC-PlantIntens was then applied uniformly to all pixels in the selected stands.

C.2.2.3 CC-NormalPlant

The goal of CC-NormalPlant was to imitate the fact that a plantation of species present before a clear-cut (often commercial species) are regularly done in Quebec to help the regeneration of the stand (Bureau du forestier en chef du Quebec, 2013).

The CC-NormalPlant prescription took the same form as CC-FunctPlant in the parameter files of Biomass Harvest (see above). It selected stands by prioritizing those with the largest amount of biomass in the area where it was applied. Once stands were selected, the python script identified the two dominant species in the stand in terms of biomass. These two dominant species were then replanted after the cut by randomly assigning their planting in the pixels of the stands. However, based on expert opinions, we made sure that the balsam fir was never replanted, even if it was dominant in the stand before the cut. Instead, it was replaced by either white or black spruce, depending on the other dominant species in the stand. This was done as the balsam fir is often seen as an undesirable tree species by the industry that is often too present in the regenerating stands, and thus not replanted.

C.2.2.4 CC-NoPlant

CC-NoPlant was the simplest form of clear-cut. It selected stands by prioritizing those with the largest amount of biomass, and applied uniformly to all pixels of the selected stands. It took the following form in the parameter file of Biomass Harvest:

```
Prescription CPRS-NOPLANT

>> STAND RANKING:
StandRanking MaxCohortAge
>> STAND QUALIFICATION FOR CUTTING:
>> None.
>> SITE SELECTION:
SiteSelection Complete
>> COHORT REMOVAL METHOD:
CohortsRemoved SpeciesList
>> Species      Cohorts removed
>> -------      ---------------
   ABIE.BAL    11-999 (90%)
   ACER.RUB    11-999 (90%)
   ACER.SAH    11-999 (90%)
   BETU.ALL    11-999 (90%)
   BETU.PAP    11-999 (90%)
   FAGU.GRA    11-999 (90%)
```

```
   LARI.LAR    11-999 (90%)
   LARI.HYB    11-999 (90%)
   PICE.GLA    11-999 (90%)
   PICE.MAR    11-999 (90%)
   PICE.RUB    11-999 (90%)
   PINU.BAN    11-999 (90%)
   PINU.RES    11-999 (90%)
   PINU.STR    11-999 (90%)
   POPU.TRE    11-999 (90%)
   POPU.HYB    11-999 (90%)
   QUER.RUB    11-999 (90%)
   THUJ.SPP.ALL     11-999 (90%)
   TSUG.CAN    11-999 (90%)
```

C.2.2.5 Selection cuts (SC)

Selection cuts were implemented to imitate an irregular shelterwood repeated other multiple years in the same stand as a form of uneven-aged management often practiced in Quebec. It harvested 30% of the biomass of all age cohorts of 30 years of age or older, and applied every 30 years to the same stand for 90 years after its first application in the stand.

SC took the following form in the parameter files of Biomass Harvest:

```
Prescription SC

>> STAND RANKING:
StandRanking MaxCohortAge
>> STAND QUALIFICATION FOR CUTTING:
>> None.
>> SITE SELECTION:
SiteSelection Complete
>> COHORT REMOVAL METHOD:
CohortsRemoved SpeciesList
>> Species       Cohorts removed
>> -------       ---------------
   ABIE.BAL    30-999(30%)
   ACER.RUB    30-999(30%)
   ACER.SAH    30-999(30%)
   BETU.ALL    30-999(30%)
   BETU.PAP    30-999(30%)
   FAGU.GRA    30-999(30%)
   LARI.LAR    30-999(30%)
   LARI.HYB    30-999(30%)
   PICE.GLA    30-999(30%)
   PICE.MAR    30-999(30%)
   PICE.RUB    30-999(30%)
   PINU.BAN    30-999(30%)
   PINU.RES    30-999(30%)
   PINU.STR    30-999(30%)
   POPU.TRE    30-999(30%)
   POPU.HYB    30-999(30%)
   QUER.RUB    30-999(30%)
   THUJ.SPP.ALL     30-999(30%)
   TSUG.CAN    30-999(30%)
```

SC prioritized stands with the most biomass. New stands to harvest with SC were selected if the biomass target to harvest with SC at each time step wasn't reached by harvesting the stands selected in previous years for harvesting with SC, to which the repeated prescriptions of SC applied every 30 years. Once a stand was selected to be harvested with SC for the next 90 years, it became unavailable to other prescriptions until the end of those 90 years. SC applied uniformly to every pixel in a stand and was not followed by any plantation. In TRIAD scenarios, SC was only applied to stands contained in the extensive zones.

C.2.2.6 Commercial Thinning (CT)

Commercial Thinning was implemented to imitate its current use in Quebec. As such, it harvested a large amount of biomass from younger age cohorts in a stand, and a smaller amount for older cohorts. It was used with two variations: an "intensive" and a "non-intensive" form. The intensive form was used in the intensive areas of the TRIAD scenarios, while the non-intensive form was used in the BAU scenarios.

CT was implemented in the following way in the parameter file of Biomass Harvest:

```
Prescription Thinning

>> STAND RANKING:
StandRanking MaxCohortAge
>> STAND QUALIFICATION FOR CUTTING:
>> None.
>> SITE SELECTION:
SiteSelection Complete
>> COHORT REMOVAL METHOD:
CohortsRemoved SpeciesList
>> Species        Cohorts removed
>> -------        ---------------
   ABIE.BAL      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   ACER.RUB      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   ACER.SAH      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   BETU.ALL      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   BETU.PAP      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   FAGU.GRA      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   LARI.LAR      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   LARI.HYB      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   PICE.GLA      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   PICE.MAR      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   PICE.RUB      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   PINU.BAN      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   PINU.RES      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   PINU.STR      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   POPU.TRE      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   POPU.HYB      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   QUER.RUB      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   THUJ.SPP.ALL  1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
   TSUG.CAN      1-30 (80%) 31-50 (66%) 51-70 (60%) 71-90 (60%) 91-100 (40%) 101-120 (5%)
```

CT prioritized stands with the largest biomass. In its intensive form, CT was applied 3 times to a given stand, with a 30-year period between application. As for SC, new stands were added to the intensive CT if the target to harvest with CT for the time step wasn't reached by the repeated CT at this time step. Once intensive CT was applied to a stand, the stand became unavailable to other prescriptions until the end of its 60 years of repeated application. In its non-intensive form, CT was only applied one time, with no repeated application. CT applied uniformly to every pixel in a stand and was not followed by any plantation.

C.3 Details of the catastrophic events – Distribution and effects

The "catastrophic" disturbance events that we triggered at t = 100 in most of our simulations were simulated as harvest prescriptions by the use of Biomass Harvest and Magic Harvest (see section C.2 for more). In this way, we were able to precisely control where and when these events triggered, as well as their impacts on the forest stands. Here, we explain how their distribution and effects were defined.

C.3.1 Large Fire

The large fire touched 70% of the pixel of forests of the landscape, with the 30% remaining being left untouched as fire refugias (see main article). To that end, every stand was put into the list of stands impacted by the fire, and fire refugias were progressively created by removing stands from the list. This process is done in the python script magicHarvestFunctions_v3.py available in the files associated to the article (see function "megaFireCatastrophy"). We represent it here in pseudocode:

```
while surfaceOfRefugias < 0.3 * surfaceOfForests:
      # We take a number from a power law distribution
      # Going from 1 to 100, with an alpha parameter of 0.07
      # Will generate a lot of small values close to 1, and a few large values
      surfaceOfNextRefugia = powerLawDistributionRandomNumber(1, 100, 0.07)
      # Next, we choose a random stand and propagate the refugia
      # from neighbouring stand to neighbouring stand until the surface
      # (in hectares) chosen with the power law is reached.
      standToStartRefugia = chooseARandomStand(listOfStandsID)
      listOfStandsInRefugia = list(standToStartRefugia)
      while area(listOfStandsInRefugia) < surfaceOfNextRefugia:
            propagateRefugia(listOfstandsInRefugia)
```

Once the refugias had been defined in that way, special harvest prescriptions were applied to all of the other (burned) stands. These prescriptions were made so that the loss of biomass for each age cohort changed according to the species, with fire tolerant species losing less biomass. This is easily made in Biomass Harvest, with the prescriptions imitating the effects of the large fire taking the following form:

```
Prescription MegaFire-100%Effect

>> STAND RANKING:
StandRanking MaxCohortAge
>> STAND QUALIFICATION FOR CUTTING:
>> None.
>> SITE SELECTION:
SiteSelection Complete
>> COHORT REMOVAL METHOD:
CohortsRemoved SpeciesList
>> Species      Cohorts removed
>> -------      ---------------
   ABIE.BAL  1-999  (100%)
   ACER.RUB  1-999  (90%)
   ACER.SAH  1-999  (90%)
   BETU.ALL  1-999  (100%)
   BETU.PAP  1-999  (100%)
   FAGU.GRA  1-999  (100%)
   LARI.LAR  1-999  (100%)
   LARI.HYB  1-999  (100%)
   PICE.GLA  1-999  (90%)
   PICE.MAR  1-999  (90%)
   PICE.RUB  1-999  (100%)
   PINU.BAN  1-999  (90%)
   PINU.RES  1-999  (80%)
   PINU.STR  1-999  (80%)
   POPU.TRE  1-999  (90%)
   POPU.HYB  1-999  (90%)
   QUER.RUB  1-999  (80%)
   THUJ.SPP.ALL     1-999  (100%)
   TSUG.CAN  1-999  (100%)
```

However, as indicated in the main text of the article, we further varied the effect of the fire from stand to stand according to the Community Weighted Mean (CWM) of the fire tolerance traits for all tree species in the stand, based on the biomass of their respective age cohorts. As such, we further created 5 prescriptions for the large fire, each of them further reducing the biomass lost for all species as indicated in Table 2 of the main article. As an example, here is the prescription for stands suffering only 90% of the "full" effect of the large fire (as indicated in the prescription above):

```
Prescription MegaFire-90%Effect

>> STAND RANKING:
StandRanking MaxCohortAge
>> STAND QUALIFICATION FOR CUTTING:
>> None.
>> SITE SELECTION:
SiteSelection Complete
>> COHORT REMOVAL METHOD:
CohortsRemoved SpeciesList
>> Species      Cohorts removed
>> -------      ---------------
   ABIE.BAL  1-999  (90%)
   ACER.RUB  1-999  (81%)
   ACER.SAH  1-999  (81%)
```

```
BETU.ALL  1-999 (90%)
BETU.PAP  1-999 (90%)
FAGU.GRA  1-999 (90%)
LARI.LAR  1-999 (90%)
LARI.HYB  1-999 (90%)
PICE.GLA  1-999 (81%)
PICE.MAR  1-999 (81%)
PICE.RUB  1-999 (90%)
PINU.BAN  1-999 (81%)
PINU.RES  1-999 (72%)
PINU.STR  1-999 (72%)
POPU.TRE  1-999 (81%)
POPU.HYB  1-999 (81%)
QUER.RUB  1-999 (72%)
THUJ.SPP.ALL     1-999 (90%)
TSUG.CAN  1-999 (90%)
```

As indicated in the main article, the relation between the age-cohort level and stand-level values of biomass loss and the fire tolerance trait of the species and of the stand were chosen arbitrarily, based on expert opinion.

C.3.2 Severe Drought

The severe drought was implemented in a way similar to the large fire. However, it impacted every stand of the landscape, with no refugias being created. The age-cohort level and stand-level effects were dealt with in the same way than with the large fire (see above).

C.3.3 Mountain Pine Beetle epidemic

The Mountain Pine Beetle (MPB) epidemic impacted only the stands of the landscape that contained one of its potential host species: the jack pine (*Pinus banksiana*), the eastern white pine (*Pinus strobus*) and the red pine (*Pinus resinosa*). All age cohorts of these three species lost 80% of their biomass across the entire landscape, with the other species left untouched. However, this effect changed from stand to stand according to their ratio of biomass between the host species and non-host species. We based the modulation of this effect on the data from the Figure 1b of Jactel *et al.* (2021). The figure shows the log response ratio of tree diversity (i.e., if the stand studied is "mixed" or "pure" as to the hosts species of the MPB) on the abundance or damages of borer insects among 45 studies. The mean of this log ratio was around -0.4557, which can be interpreted in the way that "mixed" stands had only 57% of the abundance of borer insects or damage from these insects as compared to "pure" stands. We implemented this information in the following way: Pure stands (with 100% - 90% of their biomass composed of the three host species) suffered 100% of the impact of the MPB in our simulations (i.e., 80% reduction in the biomass of the cohort of each host species). In contrast, stands with 50—40% of their biomass being composed of non-host species suffered 57% of the MPB (based on the log ratio of Jactel *et al.* 2021). The remaining 8 categories of stands that we defined according to the ratio of host species (90 – 80%, 80% - 70%, 70 – 60%, 60 – 50%, 40 – 30%, etc.) had a MPB impact that varied according to a linear relation defined by the two previous categories (100% effect for 100 – 90% host abundance, 57% effect for 50 – 40% host abundance). As such, the effect for each category was defined by the equation:

$$MPBeffectforcategory = 0.86 * Hostabundanceoftheupperboundofthecategory + 0.14$$

This variation of effect was finally implemented through 10 different prescriptions defined in the Biomass Harvest parameter file, in a way similar to the large fire and severe drought (see above).

C.4 Clustering to create the functional groups

We clustered our 17 species (without taking into account the two hybrid species) into 5 functional groups based on a database of 10 functional traits. We chose traits related to the resistance, resilience and growth of trees facing a disturbance: post fire regeneration (PFR), seed mass (SM), specific leaf area (SLA), wood density (WD), leaf nitrogen content (Nmass), bark thickness coefficient (BTcoef), maximum height (maxH), fire tolerance (FirT), drought tolerance (DT) and shade tolerance (ST). Values for each traits came from the Try Database, Paquette et Messier (2011), and from the core parameters that we used in LANDIS-II (see sections above). As we wanted the clustering to produce biologically meaningful functional groups (meaning groups that displayed species of different biological categories or life strategies), we added the characteristic of being an angiosperm or a gymnosperm as an additional trait for the clustering.

Having chosen our traits, we calculated a dissimilarity matrix based on our traits using the Gower's distance between each species (Gower, 1971). During this calculation, we ponderated the different traits according to the correlation between them, with correlated/redundant traits being ponderated less. The correlation between traits was assessed by a statistical test using Spearman’s rank-order correlation (ρ). When *x* traits were found to be significantly correlated together, we ponderated them with the value $1/x$. Ponderating the traits in that way allowed us to reduce the importance of a redundant information contained in the correlated traits during the following clustering. In the end, the SLA, WD and Nmass traits were found to be correlated together, and FirT and BTcoef with one another. This resulted in a ponderation of 0.33 for

SLA, WD and Nmass, and a ponderation of 0.5 for FirT and BTcoeff. The correlation between traits is shown in Figure C-1.

With the dissimilarity matrix calculated, we performed a hierarchical clustering with Ward's method to

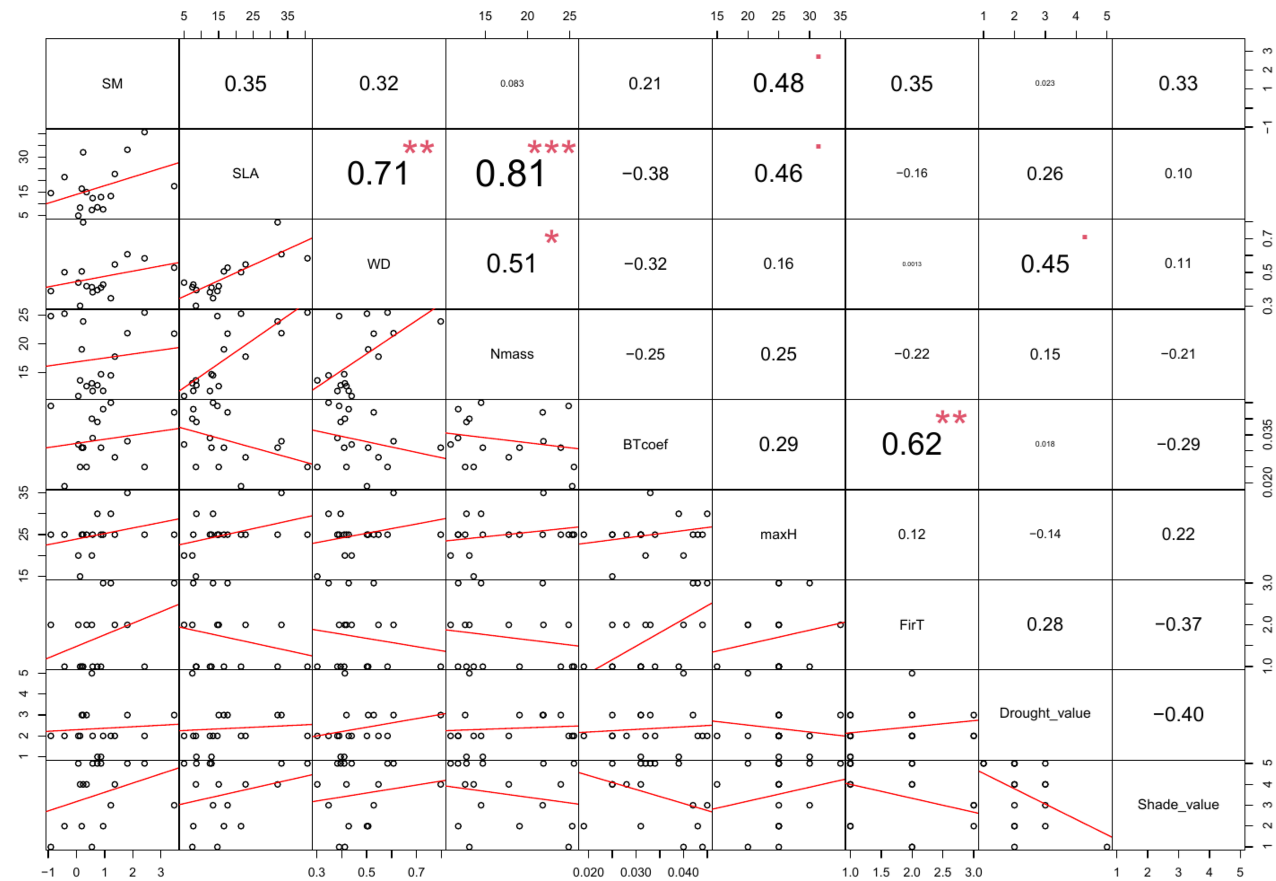

Figure C-1 : Correlogram of the trait values for our 17 different species to cluster in functional groups. Only numerical traits are represented.

aggregate species with smaller distances between them into functional groups. We then validated the number of functional groups through the analysis of the silhouette width. The dendrogram of the clustering is given below, with the final functional groups kept indicated by the blue envelopes in Figure C-2.

Dendrogram of agnes(x = gow_perso, diss = TRUE, method = "ward")

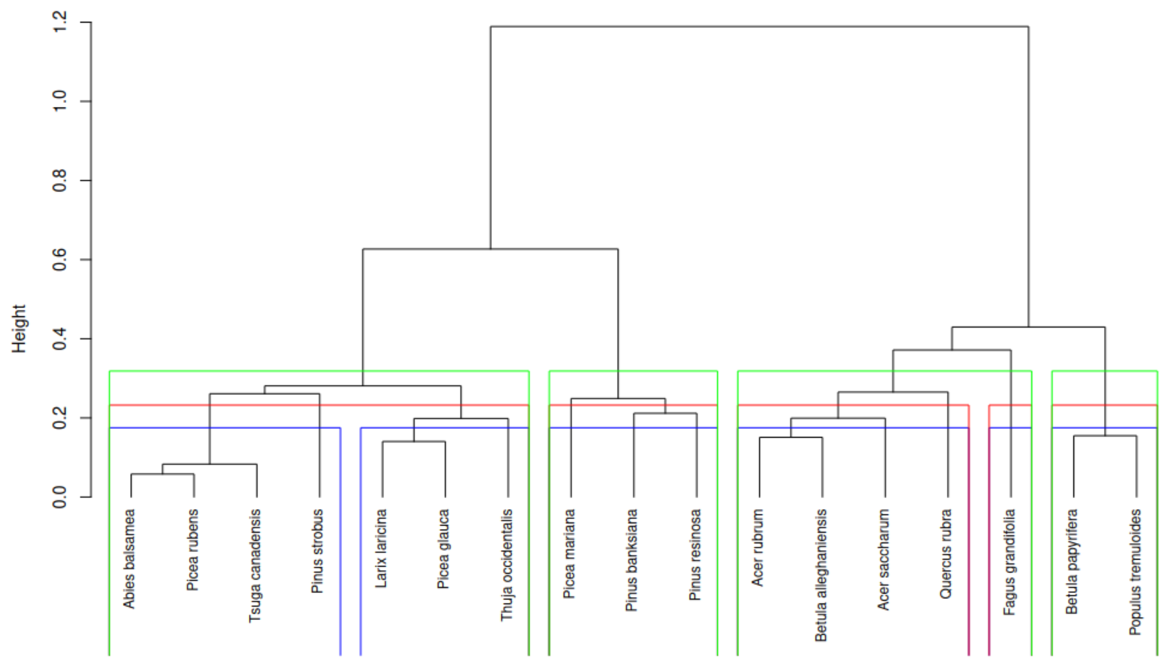

Figure C-2 : Dendrogram showing the clustering of our 17 species of interest into 5 functional groups (blue envelopes). Note that Fagus grandifolia has been included in the last group on the right of the dendrogram, with Betulla papyrifera and Populus tremuloides (see text).

The final 5 functional groups were numbered 1 to 5 (from left to right on the dendrogram), and interpreted in the following way:

1 – Softwood trees with high tolerance to shade

2 – Softwood trees with low tolerance to shade and low resistance to fire

3 – Softwood trees with low tolerance to shade and high resistance to fire

4 – Late succession hardwoods trees

5 – Pionner hardwood trees

As *Fagus grandifolia* was between group 4 and 5, we decided to include it in group 5 as it is often found as a pioneer species.

The resulting radar plots for the mean value of each trait for the different groups are shown in Figure C-3.

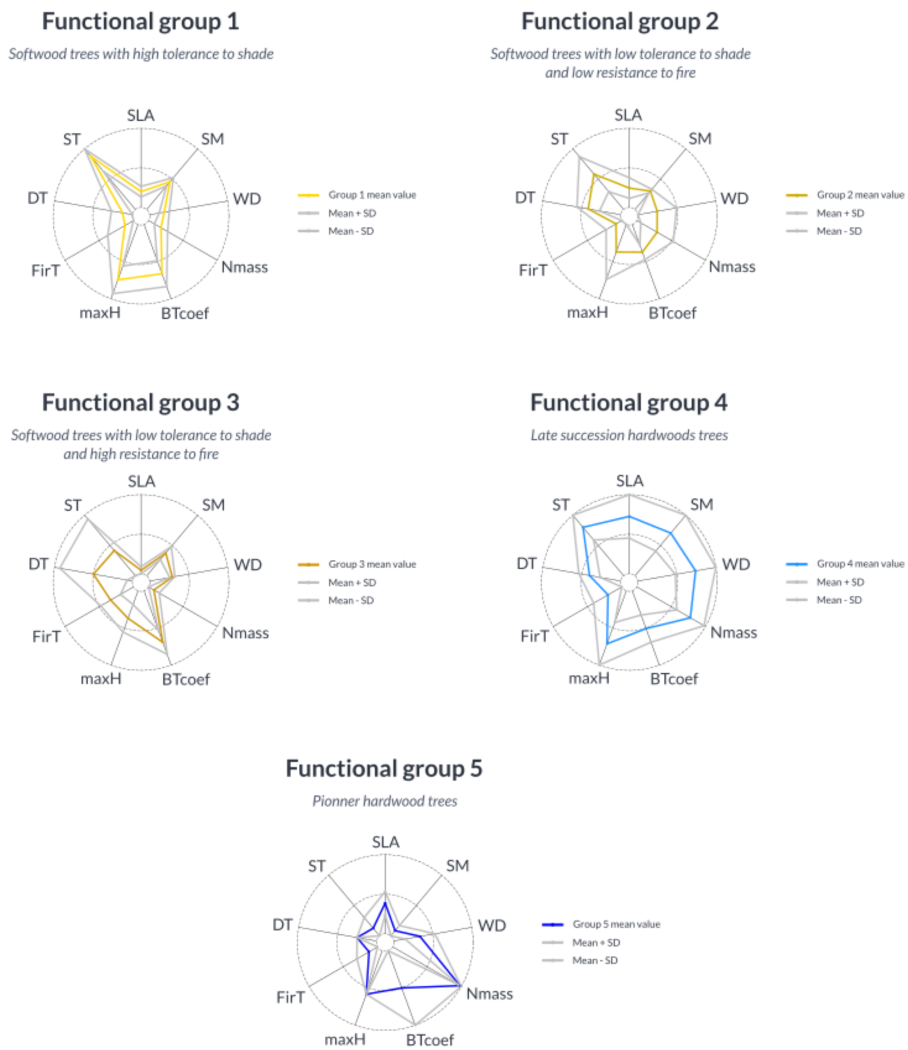

Figure C-3 : Radar plots of the mean values and standard deviation for the 5 functional groups resulting from the clustering, and for the 9 quantitative traits in our database.

C.5 Surface burned in the different scenarios

As stated in the main text of the article, the forest fires simulated by the Base Fire extension of LANDIS-II were stochastic in nature. As such, the surface burned at every time step presented a large amount of variability between replicates and scenarios. In addition, the surface burned tended to increase through time in scenarios with climate change (RCP 4.5 and RCP 8.5). Moreover, the north of our simulated area tended to have much more forest fires (and more surface burned) as it was situated in a different homogeneous fire zone (as defined by Boulanger *et al.* 2014) than the south of the area. Consequently, the south of the area had less surface burned by time step when compared to the north (by a factor of almost ten). We display these tendencies in Figure C-4, Figure C-5 and Figure C-6, which show the surface burned in the whole of the landscape, the north and the south respectively.

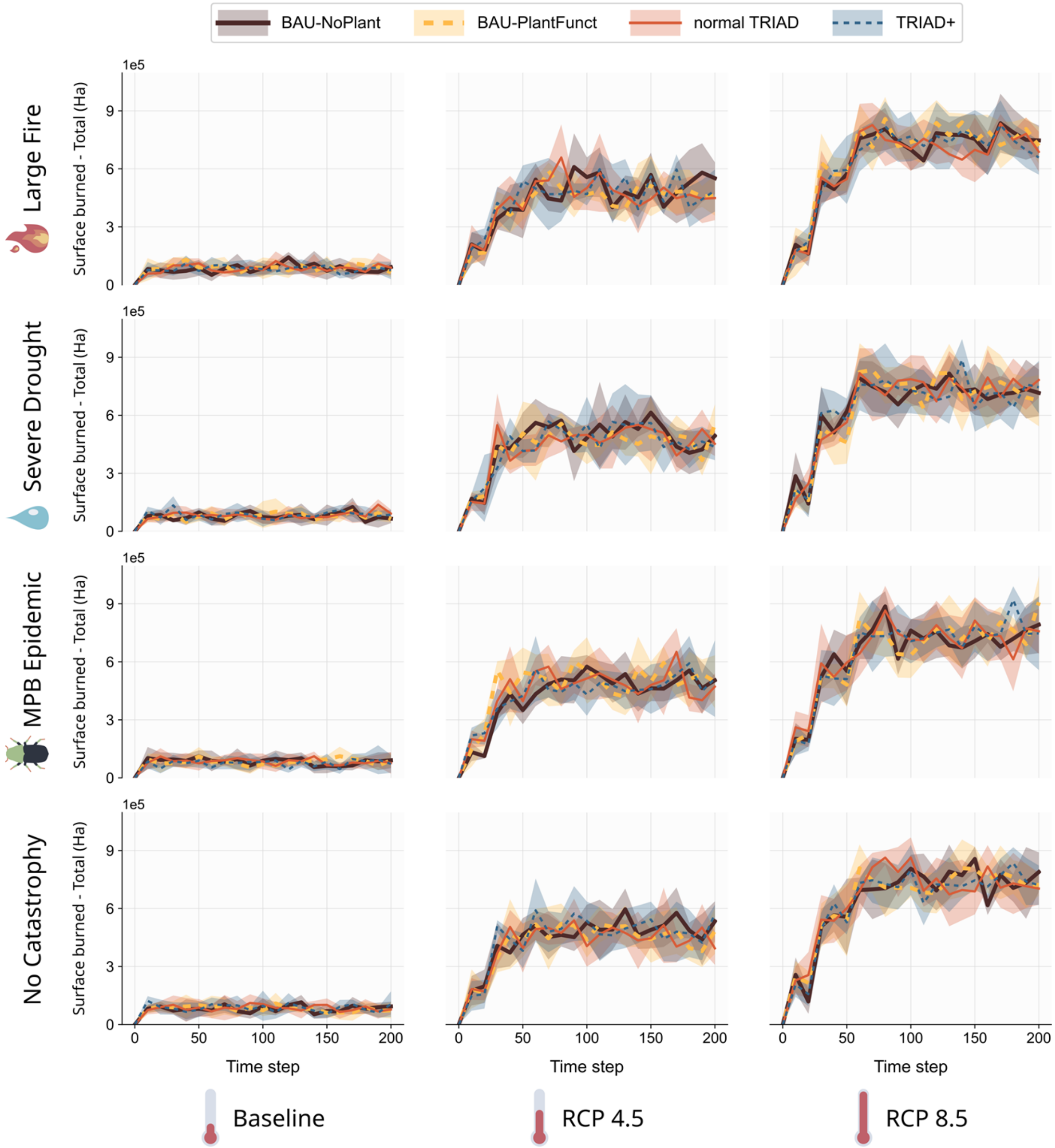

Figure C-4 : Temporal variation of the surface burned in the whole of the landscape for each combination of management, climate and catastrophe scenario. Solid lines are mean values and envelopes are standard deviation across 5 simulation replicates.

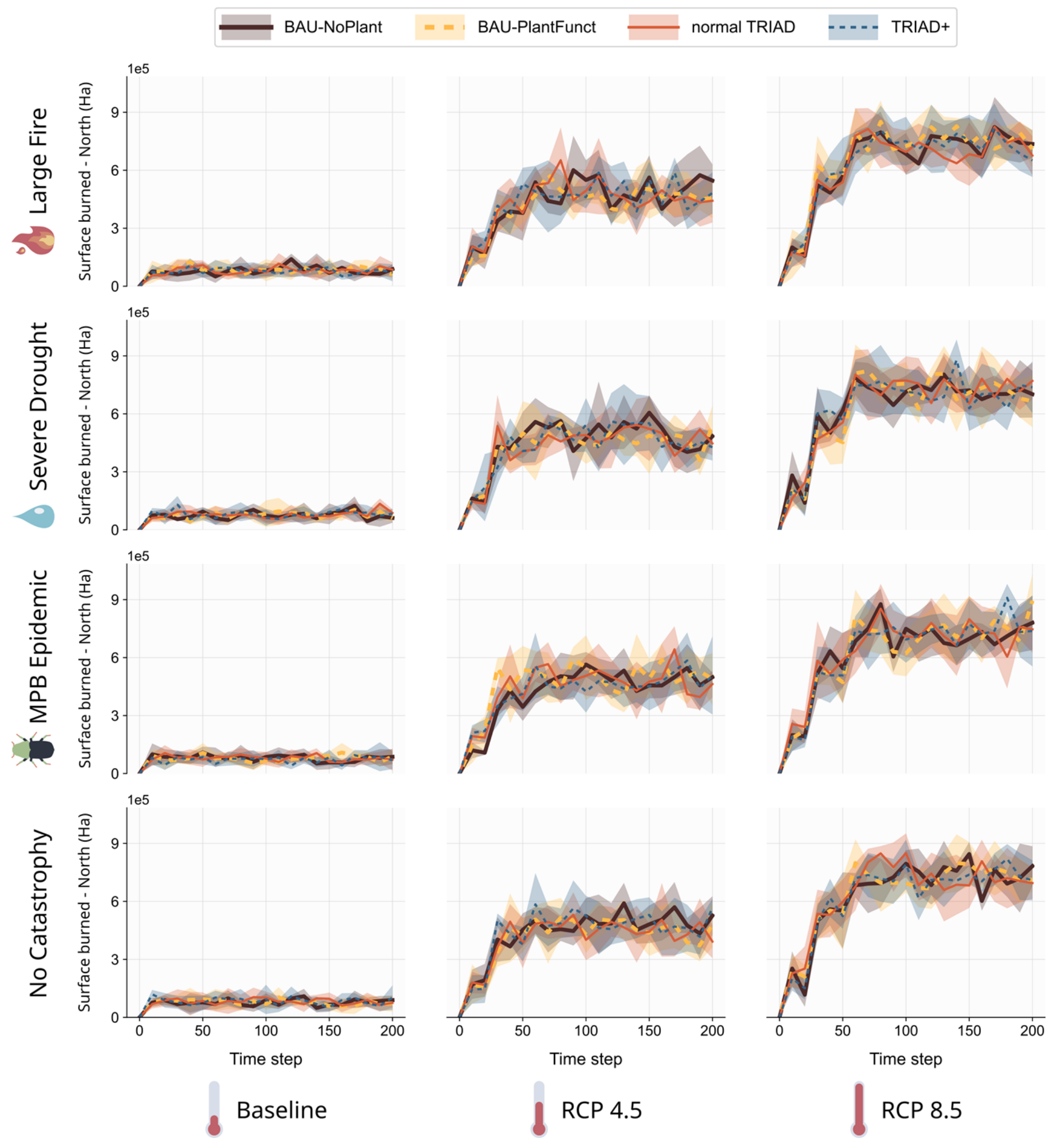

Figure C-5 : Temporal variation of the surface burned in the north of the landscape for each combination of management, climate and catastrophe scenario. Solid lines are mean values and envelopes are standard deviation across 5 simulation replicates.

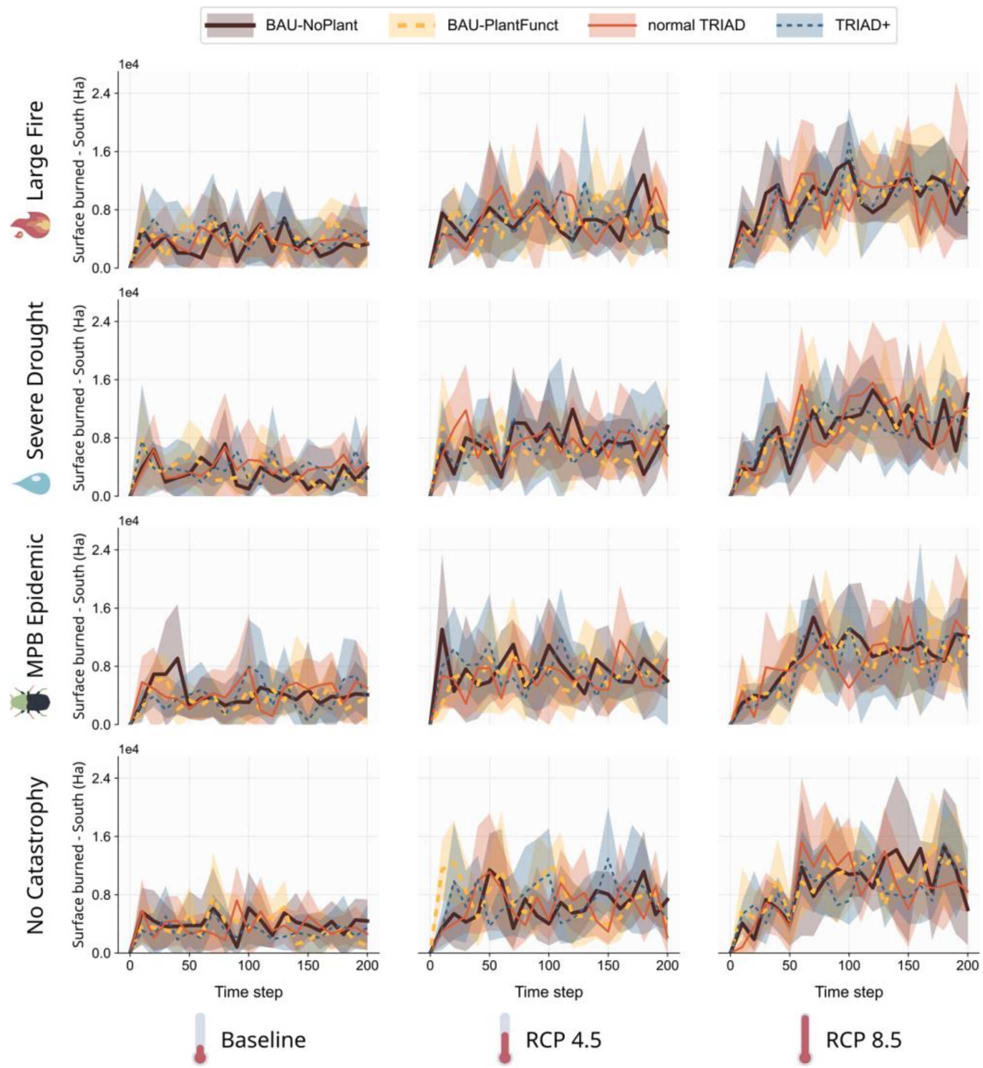

Figure C-6 : Temporal variation of the surface burned in the south of the landscape for each combination of management, climate and catastrophe scenario. Solid lines are mean values and envelopes are standard deviation across 5 simulation replicates.

C.6 Evaluating the size of the differences in the net change of the different management scenarios

In the article, we show that the TRIAD+ and BAU-PlantFunct scenario tended to improve the resilience of the mature biomass of stands impacted by a catastrophic event according to three different resilience measures. However, it can be difficult to convey the practical importance of these improvements for management decisions. In order to give a sense of scale to these differences, we propose an additional figure (Figure C-7) showing two types of values side by side. The first is the raw net change (in Mg) of the sum of the mature biomass of all forest stands impacted by a catastrophic event, varying from scenario to scenario. This is slightly different than the net change value described inFigure 3-7, which is a relative net change (in % of the value of mature biomass in the stand before the catastrophic event). In addition, Figure 3-7 shows a relative net change at the stand scale, with bars representing the mean value for all stands, while we show here a sum across all stands. The second value shown on this additional figure is the biomass target to harvest at each time step across all scenarios. In this way, we can compare the magnitude of the difference in the biomass "preserved" after the catastrophic event by improvement in forest resilience in some management scenarios to the practical reality of the biomass harvested regularly in the landscape. As shown in Figure C-7, the differences between scenario are small compared to the target biomass to harvest. This suggests that while the TRIAD+ and BAU-PlantFunct scenarios are able to increase forest resilience and to help the landscape recover more biomass than the normal TRIAD and BAU-NoPlant scenarios, their effect is not large enough to have a high influence on management decisions.

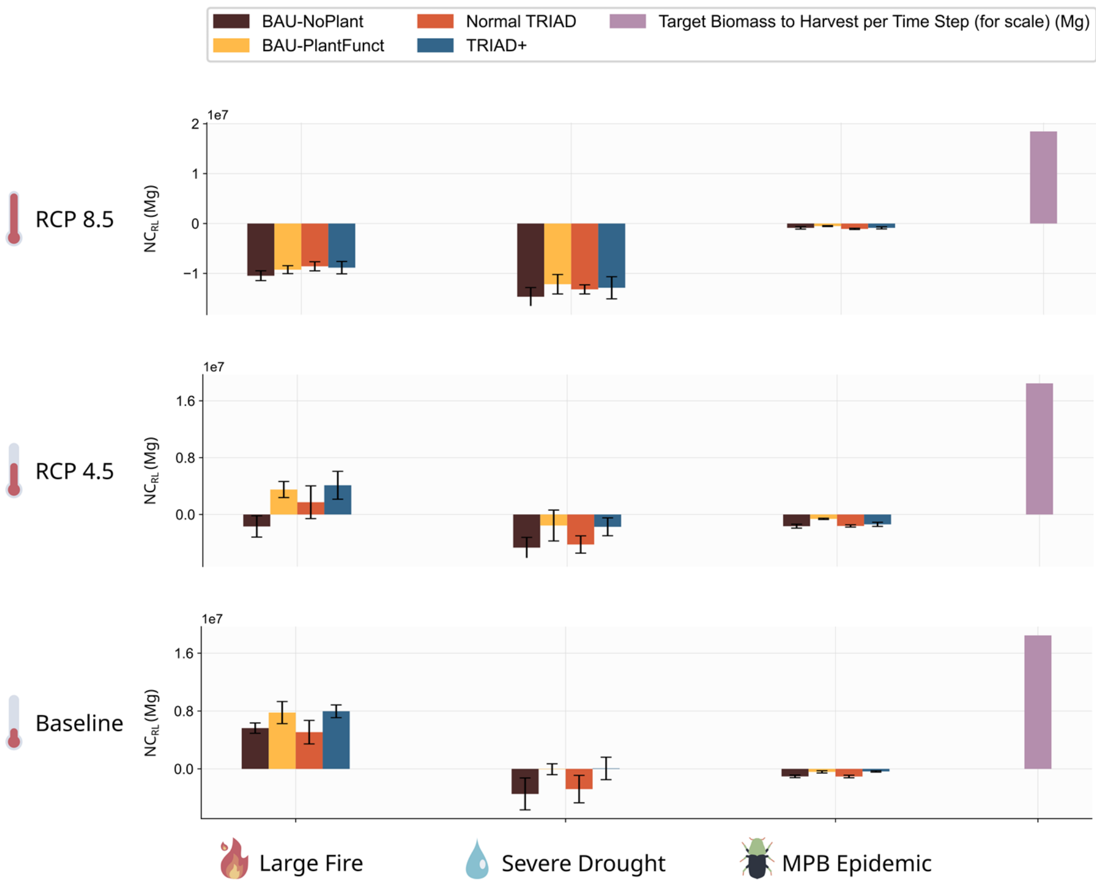

Figure C-7 : Bar plots showing the raw net change of the mature biomass of all stands impacted by a catastrophic event (NCRL). Each colored bar represents the mean and the black vertical lines the standard deviation over the 5 simulation replicates for a given combination of climate, management strategy and catastrophic event. The purple bars on the right-hand side represent the target biomass to harvest per time step in all of the simulations, independent of the management strategy used. It gives a sense of scale as to how much mature biomass can be "protected" by a given management strategy by increasing the resilience of the stands of the landscape.

RÉFÉRENCES

Abdi, E., Samdaliry, H., Ghalandarayeshi, S., Khoramizadeh, A., Sohrabi, H., Deljouei, A., Kvist Johannsen, V. et Etemad, V. (2020). Modeling wind-driven tree mortality: The effects of forest roads. *Austrian J. For. Sci*, *137*, 1-21.

Acharya, R. P., Maraseni, T. et Cockfield, G. (2019). Global trend of forest ecosystem services valuation – An analysis of publications. *Ecosystem Services*, *39*, 100979. https://doi.org/10.1016/j.ecoser.2019.100979

Adam, M.-C., Kneeshaw, D. et Beckley, T. M. (2012). Forestry and road development: Direct and indirect impacts from an aboriginal perspective. *Ecology and Society*, *17*(4).

Adamic, M., Diaci, J., Rozman, A. et Hladnik, D. (2017). Long-term use of uneven-aged silviculture in mixed mountain Dinaric forests: a comparison of old-growth and managed stands. *Forestry: An International Journal of Forest Research*, *90*(2), 279-291. https://doi.org/10.1093/forestry/cpw052

Agnoletti, M. (2006). Man, forestry, and forest landscapes. Trends and perspectives in the evolution of forestry and woodland history research. *Schweizerische Zeitschrift fur Forstwesen*, *157*(9), 384-392. https://doi.org/10.3188/szf.2006.0384

Agrawal, A., Cashore, B., Hardin, R., Shepherd, G., Benson, C. et Miller, D. (2013). Economic contributions of forests. *Background paper*, *1*, 1-127.

Ahrends, A., Burgess, N. D., Milledge, S. A. H., Bulling, M. T., Fisher, B., Smart, J. C. R., Clarke, G. P., Mhoro, B. E. et Lewis, S. L. (2010). Predictable waves of sequential forest degradation and biodiversity loss spreading from an African city. *Proceedings of the National Academy of Sciences*, *107*(33), 14556-14561. https://doi.org/10.1073/pnas.0914471107

Akbarimehr, M. et Naghdi, R. (2012). Reducing erosion from forest roads and skid trails by management practices. *Journal of Forest Science*, *58 (2012)*(No. 4), 165-169. https://doi.org/10.17221/136/2010-JFS

Alain, D., Boudreau, J.-F., Philibert, Y., Pomerleau, I., Québec (Province) et Direction des inventaires forestiers. (2016a). *Placettes-échantillons permanentes: normes techniques*. http://collections.banq.qc.ca/ark:/52327/2748265

Alain, D., Boudreau, J.-F., Philibert, Y., Pomerleau, I., Québec (Province) et Direction des inventaires forestiers. (2016b). *Placettes-échantillons temporaires: normes techniques*. http://collections.banq.qc.ca/ark:/52327/2748267

Albrich, K., Rammer, W. et Seidl, R. (2020a). Climate change causes critical transitions and irreversible alterations of mountain forests. *Global Change Biology*, *26*(7), 4013-4027. https://doi.org/10.1111/gcb.15118

Albrich, K., Rammer, W., Turner, M. G., Ratajczak, Z., Braziunas, K. H., Hansen, W. D. et Seidl, R. (2020b). Simulating forest resilience: A review. *Global Ecology and Biogeography*, *29*(12), 2082-2096. https://doi.org/10.1111/geb.13197

Alexander, R. R. et Edminster, C. B. (1977). Uneven-aged management of old-growth spruce-fir forests: Cutting methods and stand structure goals for the initial entry. *USDA Forest Service*, 19.

Altieri, V., De Franco, S., Lombardi, F., Marziliano, P. A., Menguzzato, G. et Porto, P. (2018). The role of silvicultural systems and forest types in preventing soil erosion processes in mountain forests: a methodological approach using cesium-137 measurements. *Journal of Soils and Sediments*, *18*(12), 3378-3387. https://doi.org/10.1007/s11368-018-1957-8

Anderegg, W. R. L., Trugman, A. T., Badgley, G., Anderson, C. M., Bartuska, A., Ciais, P., Cullenward, D., Field, C. B., Freeman, J., Goetz, S. J., Hicke, J. A., Huntzinger, D., Jackson, R. B., Nickerson, J., Pacala, S. et Randerson, J. T. (2020). Climate-driven risks to the climate mitigation potential of forests. *Science*, *368*(6497), eaaz7005. https://doi.org/10.1126/science.aaz7005

Anderson, A. E. et Nelson, J. (2004). Projecting vector-based road networks with a shortest path algorithm. *Canadian Journal of Forest Research*, *34*(7), 1444-1457. https://doi.org/10.1139/x04-030

Aquilué, N., Filotas, É., Craven, D., Fortin, M., Brotons, L. et Messier, C. (2020). Evaluating forest resilience to global threats using functional response traits and network properties. *Ecological Applications*, *30*(5). https://doi.org/10.1002/eap.2095

Aquilué, N., Messier, C., Martins, K. T., Dumais-Lalonde, V. et Mina, M. (2021). A simple-to-use management approach to boost adaptive capacity of forests to global uncertainty. *Forest Ecology and Management*, *481*, 118692. https://doi.org/10.1016/j.foreco.2020.118692

Armenteras, D., Espelta, J. M., Rodríguez, N. et Retana, J. (2017). Deforestation dynamics and drivers in different forest types in Latin America: Three decades of studies (1980–2010). *Global Environmental Change*, *46*, 139-147. https://doi.org/10.1016/j.gloenvcha.2017.09.002

Arrhenius, P. S. (1896). On the influence of carbonic acid in the air upon the temperature of the ground. *The London, Edinburgh, and Dublin Philosophical Magazine and Journal of Science*, *41*(251), 237-276. https://doi.org/10.1080/14786449608620846

Au, T. F., Maxwell, J. T., Robeson, S. M., Li, J., Siani, S. M. O., Novick, K. A., Dannenberg, M. P., Phillips, R. P., Li, T., Chen, Z. et Lenoir, J. (2022). Younger trees in the upper canopy are more sensitive but also more resilient to drought. *Nature Climate Change*, *12*(12), 1168-1174. https://doi.org/10.1038/s41558-022-01528-w

Aubin, I., Munson, A. D., Cardou, F., Burton, P. J., Isabel, N., Pedlar, J. H., Paquette, A., Taylor, A. R., Delagrange, S., Kebli, H., Messier, C., Shipley, B., Valladares, F., Kattge, J., Boisvert-Marsh, L. et McKenney, D. (2016). Traits to stay, traits to move: a review of functional traits to assess sensitivity and adaptive capacity of temperate and boreal trees to climate change. *Environmental Reviews*, *24*(2), 164-186. https://doi.org/10.1139/er-2015-0072

Auger, I. et Power, H. (2019). Une nouvelle version pour les modèles de croissance forestière Artémis et Natura. *Avis de recherche forestière*, *116*, 2.

Austin, K. G., Baker, J. S., Sohngen, B. L., Wade, C. M., Daigneault, A., Ohrel, S. B., Ragnauth, S. et Bean, A. (2020). The economic costs of planting, preserving, and managing the world's forests to mitigate climate change. *Nature Communications*, *11*(1), 5946. https://doi.org/10.1038/s41467-020-19578-z

Avon, C., Bergès, L., Dumas, Y. et Dupouey, J.-L. (2010). Does the effect of forest roads extend a few meters or more into the adjacent forest? A study on understory plant diversity in managed oak stands. *Forest Ecology and Management*, *259*(8), 1546-1555. https://doi.org/10.1016/j.foreco.2010.01.031

Baccini, A., Walker, W., Carvalho, L., Farina, M., Sulla-Menashe, D. et Houghton, R. A. (2017). Tropical forests are a net carbon source based on aboveground measurements of gain and loss. *Science*, *358*(6360), 230-234. https://doi.org/10.1126/science.aam5962

Balmford, A. (2021). Concentrating vs. spreading our footprint: how to meet humanity's needs at least cost to nature. *Journal of Zoology*, *315*(2), 79-109. https://doi.org/10.1111/jzo.12920

Barnes, C., Boulanger, Y., Keeping, T., Gachon, P., Gillett, N., Boucher, J., Roberge, F., Kew, S., Haas, O., Heinrich, D., Vahlberg, M., Singh, R., Elbe, M., Sivanu, S., Arrighi, J., Van Aalst, M., Otto, F., Zachariah, M., Krikken, F., … Kimutai, J. (2023, 21 août). *Climate change more than doubled the likelihood of extreme fire weather conditions in Eastern Canada* [Report]. https://doi.org/10.25561/105981

Bar-On, Y. M., Phillips, R. et Milo, R. (2018). The biomass distribution on Earth. *Proceedings of the National Academy of Sciences*, *115*(25), 6506-6511. https://doi.org/10.1073/pnas.1711842115

Bast, H., Mehlhorn, K., Schäfer, G. et Tamaki, H. (2003). A heuristic for Dijkstra's algorithm with many targets and its use in weighted matching algorithms. *Algorithmica*, *36*(1), 75-88.

Batavia, C. et Nelson, M. P. (2017). For goodness sake! What is intrinsic value and why should we care? *Biological Conservation*, *209*, 366-376. https://doi.org/10.1016/j.biocon.2017.03.003

Beaudoin, A., Bernier, P. Y., Guindon, L., Villemaire, P., Guo, X. J., Stinson, G., Bergeron, T., Magnussen, S. et Hall, R. J. (2014). Mapping attributes of Canada's forests at moderate resolution through kNN and MODIS imagery. *Canadian Journal of Forest Research*, *44*(5), 521-532. https://doi.org/10.1139/cjfr-2013-0401

Bekessy, S. A., Runge, M. C., Kusmanoff, A. M., Keith, D. A. et Wintle, B. A. (2018). Ask not what nature can do for you: A critique of ecosystem services as a communication strategy. *Biological Conservation*, *224*, 71-74. https://doi.org/10.1016/j.biocon.2018.05.017

Bell, F. W., Parton, J., Stocker, N., Joyce, D., Reid, D., Wester, M., Stinson, A., Kayahara, G. et Towill, B. (2008). Developing a silvicultural framework and definitions for use in forest management planning and practice. *The Forestry Chronicle*, *84*(5), 678-693.

Bennett, A. C., McDowell, N. G., Allen, C. D. et Anderson-Teixeira, K. J. (2015). Larger trees suffer most during drought in forests worldwide. *Nature Plants*, *1*(10), 1-5. https://doi.org/10.1038/nplants.2015.139

Bennett, V. J. (2017). Effects of Road Density and Pattern on the Conservation of Species and Biodiversity. *Current Landscape Ecology Reports*, *2*(1), 1-11. https://doi.org/10.1007/s40823-017-0020-6

Bentley, J. L. (1975). Multidimensional binary search trees used for associative searching. *Communications of the ACM*, *18*(9), 509-517. https://doi.org/10.1145/361002.361007

Bergeron, Y. (2004). Is regulated even-aged management the right strategy for the Canadian boreal forest? *The Forestry Chronicle*, *80*, 458-462. https://doi.org/10.5558/tfc80458-4

Bergeron, Y. et Fenton, N. J. (2012). Boreal forests of eastern Canada revisited: old growth, nonfire disturbances, forest succession, and biodiversity. *Botany*, *90*(6), 509-523. https://doi.org/10.1139/b2012-034

Bergeron, Y., Flannigan, M., Gauthier, S., Leduc, A. et Lefort, P. (2004). Past, Current and Future Fire Frequency in the Canadian Boreal Forest: Implications for Sustainable Forest Management. *AMBIO: A Journal of the Human Environment*, *33*(6), 356-360. https://doi.org/10.1579/0044-7447-33.6.356

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

Bernier, P., Schoene, D. et others. (2009). *Adapting forests and their management to climate change: an overiew* (vol. 60). Information Service of FAO Rome, Italy.

Bernier, P. Y., Desjardins, R. L., Karimi-Zindashty, Y., Worth, D., Beaudoin, A., Luo, Y. et Wang, S. (2011). Boreal lichen woodlands: A possible negative feedback to climate change in eastern North America. *Agricultural and Forest Meteorology*, *151*(4), 521-528. https://doi.org/10.1016/j.agrformet.2010.12.013

Betts, M. G., Phalan, B. T., Wolf, C., Baker, S. C., Messier, C., Puettmann, K. J., Green, R., Harris, S. H., Edwards, D. P., Lindenmayer, D. B. et Balmford, A. (2021a). Producing wood at least cost to biodiversity: integrating Triad and sharing–sparing approaches to inform forest landscape management. *Biological Reviews*, *96*(4), 1301-1317. https://doi.org/10.1111/brv.12703

Betts, M., Yang, Z., Hadley, A., Rousseau, J., Northrup, J., Nocera, J., Gorelick, N. et Gerber, B. (2021b). *Forest degradation, not loss, drives widespread avian population declines*. https://doi.org/10.21203/rs.3.rs-686817/v1

Binkley, D. (2021). *Forest Ecology: An Evidence-Based Approach*. John Wiley & Sons.

Birdsey, R. A., DellaSala, D. A., Walker, W. S., Gorelik, S. R., Rose, G. et Ramírez, C. E. (2023). Assessing carbon stocks and accumulation potential of mature forests and larger trees in U.S. federal lands. *Frontiers in Forests and Global Change*, *5*. https://www.frontiersin.org/articles/10.3389/ffgc.2022.1074508

Blattert, C., Mönkkönen, M., Burgas, D., Di Fulvio, F., Toraño Caicoya, A., Vergarechea, M., Klein, J., Hartikainen, M., Antón-Fernández, C., Astrup, R., Emmerich, M., Forsell, N., Lukkarinen, J., Lundström, J., Pitzén, S., Poschenrieder, W., Primmer, E., Snäll, T. et Eyvindson, K. (2023). Climate targets in European timber-producing countries conflict with goals on forest ecosystem services and biodiversity. *Communications Earth & Environment*, *4*(1), 1-12. https://doi.org/10.1038/s43247-023-00771-z

Bliss, J. C. (2000). Public Perceptions of Clearcutting. *Journal of Forestry*, *98*(12), 4-9. https://doi.org/10.1093/jof/98.12.4

BlueRaja. (2015). *OptimizedPriorityQueue C# Nuget Package*. https://github.com/BlueRaja/High-Speed-Priority-Queue-for-C-Sharp#readme

Bolte, A., Ammer, C., Löf, M., Madsen, P., Nabuurs, G.-J., Schall, P., Spathelf, P. et Rock, J. (2009). Adaptive forest management in central Europe: Climate change impacts, strategies and integrative concept. *Scandinavian Journal of Forest Research*, *24*(6), 473-482. https://doi.org/10.1080/02827580903418224

Bolton, A. V., Montag, D. et Gallo, V. (2022). Global forestry areas, deforestation and mental health: A worldwide ecological study. *The Journal of Climate Change and Health*, *6*, 100109. https://doi.org/10.1016/j.joclim.2021.100109

Bonan, G. B. (2008). Forests and Climate Change: Forcings, Feedbacks, and the Climate Benefits of Forests. *Science*, *320*(5882), 1444-1449. https://doi.org/10.1126/science.1155121

Boncina, A. (2011). History, current status and future prospects of uneven-aged forest management in the Dinaric region: an overview. *Forestry: An International Journal of Forest Research*, *84*(5), 467-478. https://doi.org/10.1093/forestry/cpr023

Bont, L. G., Heinimann, H. R. et Church, R. L. (2015). Concurrent optimization of harvesting and road network layouts under steep terrain. *Annals of Operations Research*, *232*(1), 41-64. https://doi.org/10.1007/s10479-012-1273-4

Boston, K. (2016). The Potential Effects of Forest Roads on the Environment and Mitigating their Impacts. *Current Forestry Reports*, *2*(4), 215-222. https://doi.org/10.1007/s40725-016-0044-x

Bouchard, M., Aquilué, N., Périé, C. et Lambert, M.-C. (2019). Tree species persistence under warming conditions: A key driver of forest response to climate change. *Forest Ecology and Management*, *442*, 96-104. https://doi.org/10.1016/j.foreco.2019.03.040

Boudreault, C., Zouaoui, S., Drapeau, P., Bergeron, Y. et Stevenson, S. (2013). Canopy openings created by partial cutting increase growth rates and maintain the cover of three Cladonia species in the Canadian boreal forest. *Forest Ecology and Management*, *304*, 473-481. https://doi.org/10.1016/j.foreco.2013.05.043

Boulanger, Y., Arseneault, D., Boucher, Y., Gauthier, S., Cyr, D., Taylor, A. R., Price, D. T. et 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. et 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 Forests. *Canadian Journal of Forest Research*, *42*(7), 1264‑1276. https://doi.org/10.1139/x2012-069

Boulanger, Y., Gauthier, S. et 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. https://doi.org/10.1139/cjfr-2013-0372

Boulanger, Y. et Pascual Puigdevall, 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., Puigdevall, J. P., Bélisle, A. C., Bergeron, Y., Brice, M.-H., Cyr, D., De Grandpre, L., Fortin, D., Gauthier, S., Grondin, P., Labadie, G., Leblond, M., Marchand, M., Splawinski, T. B., St-Laurent, M.-H., Thiffault, E., Tremblay, J. et 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*, cjfr-2022-0282. https://doi.org/10.1139/cjfr-2022-0282

Boulanger, Y., Taylor, A. R., Price, D. T., Cyr, D., McGarrigle, E., Rammer, W., Sainte-Marie, G., Beaudoin, A., Guindon, L. et 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

Boulanger, Y., Taylor, A. R., Price, D. T., Cyr, D. et Sainte-Marie, G. (2018). Stand-level drivers most important in determining boreal forest response to climate change. *Journal of Ecology*, *106*(3), 977‑990. https://doi.org/10.1111/1365-2745.12892

Bourgeois, L., Kneeshaw, D. et Boisseau, G. (2005). Les routes forestières au Québec : Les impacts environnementaux, sociaux et économiques. *VertigO - la revue électronique en sciences de l'environnement*, (Volume 6 Numéro 2). https://doi.org/10.4000/vertigo.4352

Bradford, J. B., Jensen, N. R., Domke, G. M. et D'Amato, A. W. (2013). Potential increases in natural disturbance rates could offset forest management impacts on ecosystem carbon stocks. *Forest Ecology and Management*, *308*, 178‑187. https://doi.org/10.1016/j.foreco.2013.07.042

Brang, P., Spathelf, P., Larsen, J. B., Bauhus, J., Boncčina, A., Chauvin, C., Drössler, L., García-Güemes, C., Heiri, C., Kerr, G. et others. (2014). Suitability of close-to-nature silviculture for adapting temperate European forests to climate change. *Forestry: An International Journal of Forest Research*, *87*(4), 492‑503.

Brecka, A. F. J., Boulanger, Y., Searle, E. B., Taylor, A. R., Price, D. T., Zhu, Y., Shahi, C. et 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/10.1016/j.foreco.2020.118352

Breckling, B., Jopp, F. et Reuter, H. (2011). Historical Background of Ecological Modelling and Its Importance for Modern Ecology. Dans *Modelling Complex Ecological Dynamics* (p. 29-40). Springer, Berlin, Heidelberg. https://doi.org/10.1007/978-3-642-05029-9_3

Brice, M.-H. (2023). *Effet des changements climatiques sur la composition des forêts du Québec méridional.* https://mffp.gouv.qc.ca/les-forets/carrefour-forets/colloques/mesurer-vulnerabilite-forets-changements-climatiques-strategie-amenagement/

Bright, R. M., Bogren, W., Bernier, P. et Astrup, R. (2016). Carbon-equivalent metrics for albedo changes in land management contexts: relevance of the time dimension. *Ecological Applications*, *26*(6), 1868-1880. https://doi.org/10.1890/15-1597.1

Broadbent, E. N., Asner, G. P., Keller, M., Knapp, D. E., Oliveira, P. J. C. et Silva, J. N. (2008). Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. *Biological Conservation*, *141*(7), 1745-1757. https://doi.org/10.1016/j.biocon.2008.04.024

Brockerhoff, E. G., Barbaro, L., Castagneyrol, B., Forrester, D. I., Gardiner, B., González-Olabarria, J. R., Lyver, P. O., Meurisse, N., Oxbrough, A., Taki, H., Thompson, I. D., van der Plas, F. et Jactel, H. (2017). Forest biodiversity, ecosystem functioning and the provision of ecosystem services. *Biodiversity and Conservation*, *26*(13), 3005-3035. https://doi.org/10.1007/s10531-017-1453-2

Brockerhoff, E. G., Jactel, H., Parrotta, J. A., Quine, C. P. et Sayer, J. (2008). Plantation forests and biodiversity: oxymoron or opportunity? *Biodiversity and Conservation*, *17*(5), 925-951. https://doi.org/10.1007/s10531-008-9380-x

Brunette, M., Bourke, R., Hanewinkel, M. et Yousefpour, R. (2018). Adaptation to Climate Change in Forestry: A Multiple Correspondence Analysis (MCA). *Forests*, *9*(1), 20. https://doi.org/10.3390/f9010020

Buma, B. et Wessman, C. A. (2013). Forest resilience, climate change, and opportunities for adaptation: A specific case of a general problem. *Forest Ecology and Management*, *306*, 216-225. https://doi.org/10.1016/j.foreco.2013.06.044

Bureau du forestier en chef du Quebec/ Quebec's Office of the Chief Forester. (2013). *Manuel de détermination des possibilités forestières, 2013-2018*. Bureau du forestier en chef.

Burivalova, Z., Şekercioğlu, Ç. H. et Koh, L. P. (2014). Thresholds of Logging Intensity to Maintain Tropical Forest Biodiversity. *Current Biology*, *24*(16), 1893-1898. https://doi.org/10.1016/j.cub.2014.06.065

Burns, R. M., Honkala, B. H. et others. (1990). Silvics of North America: 1. Conifers; 2. Hardwoods Agriculture Handbook 654. *US Department of Agriculture, Forest Service, Washington, DC*.

Burrows, M. T., Schoeman, D. S., Buckley, L. B., Moore, P., Poloczanska, E. S., Brander, K. M., Brown, C., Bruno, J. F., Duarte, C. M., Halpern, B. S. et others. (2011). The pace of shifting climate in marine and terrestrial ecosystems. *Science (New York, N.Y.)*, *334*(6056), 652-655.

Burton, P. J. et Canada, N. R. C. (2003). *Towards Sustainable Management of the Boreal Forest*. NRC Research Press.

Canelles, Q., Aquilué, N., James, P. M. A., Lawler, J. et Brotons, L. (2021). Global review on interactions between insect pests and other forest disturbances. *Landscape Ecology*, *36*(4), 945-972. https://doi.org/10.1007/s10980-021-01209-7

Cantarello, E., Newton, A. C., Martin, P. A., Evans, P. M., Gosal, A. et Lucash, M. S. (2017). Quantifying resilience of multiple ecosystem services and biodiversity in a temperate forest landscape. *Ecology and Evolution*, *7*(22), 9661-9675. https://doi.org/10.1002/ece3.3491

Carles, S. (2023). *Les nouveaux défis de la production de plants dans un contexte de changements climatiques.* https://mffp.gouv.qc.ca/les-forets/carrefour-forets/colloques/mesurer-vulnerabilite-forets-changements-climatiques-strategie-amenagement/

Carleton, T. J. et Maclellan, P. (1994). Woody vegetation responses to fire versus clear-cutting logging: A comparative survey in the central Canadian boreal forest. *Écoscience*, *1*(2), 141-152. https://doi.org/10.1080/11956860.1994.11682238

Carlson, M. et Kurz, W. A. (2007). Approximating natural landscape pattern using aggregated harvest. *Canadian Journal of Forest Research*, *37*(10), 1846-1853. https://doi.org/10.1139/X07-059

Carpenter, S., Walker, B., Anderies, J. M. et Abel, N. (2001). From Metaphor to Measurement: Resilience of What to What? *Ecosystems*, *4*(8), 765-781. https://doi.org/10.1007/s10021-001-0045-9

Carvalho Jr, E. A. R., Nienow, S. S., Bonavigo, P. H. et Haugaasen, T. (2021). Mammal responses to reduced-impact logging in Amazonian forest concessions. *Forest Ecology and Management*, *496*, 119401. https://doi.org/10.1016/j.foreco.2021.119401

Cavanagh, J.-A. E. et Clemons, J. (2006). Do Urban Forests Enhance Air Quality? *Australasian Journal of Environmental Management*, *13*(2), 120-130. https://doi.org/10.1080/14486563.2006.9725122

Chaudhary, A., Burivalova, Z., Koh, L. P. et Hellweg, S. (2016). Impact of Forest Management on Species Richness: Global Meta-Analysis and Economic Trade-Offs. *Scientific Reports*, *6*(1), 23954. https://doi.org/10.1038/srep23954

Chung, W., Sessions, J. et Heinimann, H. R. (2004). An Application of a Heuristic Network Algorithm to Cable Logging Layout Design. *International Journal of Forest Engineering*, *15*(1), 11-24. https://doi.org/10.1080/14942119.2004.10702485

Chung, W., Stückelberger, J., Aruga, K. et Cundy, T. W. (2008). Forest road network design using a trade-off analysis between skidding and road construction costs. *Canadian Journal of Forest Research*, *38*(3), 439-448. https://doi.org/10.1139/X07-170

Churkina, G., Organschi, A., Reyer, C. P. O., Ruff, A., Vinke, K., Liu, Z., Reck, B. K., Graedel, T. E. et Schellnhuber, H. J. (2020). Buildings as a global carbon sink. *Nature Sustainability*, *3*(4), 269-276. https://doi.org/10.1038/s41893-019-0462-4

Civitello, D. J., Cohen, J., Fatima, H., Halstead, N. T., Liriano, J., McMahon, T. A., Ortega, C. N., Sauer, E. L., Sehgal, T., Young, S. et Rohr, J. R. (2015). Biodiversity inhibits parasites: Broad evidence for the dilution effect. *Proceedings of the National Academy of Sciences*, *112*(28), 8667-8671, world. https://doi.org/10.1073/pnas.1506279112

Clark, W. A. (2011). Clarifying the spiritual value of forests and their role in sustainable forest management. *Journal for the Study of Religion, Nature & Culture*, *5*(1).

Coggins, C. et Chen, B. (2022). *Sacred Forests of Asia: Spiritual Ecology and the Politics of Nature Conservation*. New York: Routledge.

Commission indépendante sur les caribous forestiers et montagnards. (2022). *Rapport Final de la Commission Indépendante sur les Caribous Forestiers et Montagnards*. Gouvernement du Québec.

Cooke, B. J. et Carroll, A. L. (2017). Predicting the risk of mountain pine beetle spread to eastern pine forests: Considering uncertainty in uncertain times. *Forest Ecology and Management*, *396*, 11-25. https://doi.org/10.1016/j.foreco.2017.04.008

Cosofret, C. et Bouriaud, L. (2019). Which Silvicultural Measures Are Recommended to Adapt Forests to Climate Change? A Literature Review. *Bulletin of the Transilvania University of Brasov. Series II: Forestry • Wood Industry • Agricultural Food Engineering*, 13-34. https://doi.org/10.31926/but.fwiafe.2019.12.61.1.2

Côté, P., Tittler, R., Messier, C., Kneeshaw, D. D., Fall, A. et Fortin, M.-J. (2010). Comparing different forest zoning options for landscape-scale management of the boreal forest: Possible benefits of the TRIAD. *Forest Ecology and Management*, *259*(3), 418-427. https://doi.org/10.1016/j.foreco.2009.10.038

Couillard, P.-L., Bouchard, M., Laflamme, J. et Hébert, F. (2022). *Zonage des régimes de feux du Québec méridional* (189). Gouvernement du Québec, ministère des Forêts, de la Faune et des Parcs, Direction de la recherche forestière.

Craven, D., Filotas, E., Angers, V. A. et Messier, C. (2016). Evaluating resilience of tree communities in fragmented landscapes: linking functional response diversity with landscape connectivity. *Diversity and Distributions*, *22*(5), 505-518. https://doi.org/10.1111/ddi.12423

Cuddington, K., Fortin, M.-J., Gerber, L. R., Hastings, A., Liebhold, A., O'connor, M. et Ray, C. (2013). Process-based models are required to manage ecological systems in a changing world. *Ecosphere*, *4*(2), 1-12.

Curtis, P. G., Slay, C. M., Harris, N. L., Tyukavina, A. et Hansen, M. C. (2018). Classifying drivers of global forest loss. *Science*, *361*(6407), 1108-1111. https://doi.org/10.1126/science.aau3445

Cyr, D., Gauthier, S., Bergeron, Y. et Carcaillet, C. (2009). Forest management is driving the eastern North American boreal forest outside its natural range of variability. *Frontiers in Ecology and the Environment*, *7*(10), 519-524. https://doi.org/10.1890/080088

Cyr, D., Splawinski, T. B., Pascual Puigdevall, J., Valeria, O., Leduc, A., Thiffault, N., Bergeron, Y. et Gauthier, S. (2022). 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.

Dale, V. H., Joyce, L. A., Mcnulty, S., Neilson, R. P., Ayres, M. P., Flannigan, M. D., Hanson, P. J., Irland, L. C., Lugo, A. E., Peterson, C. J., Simberloff, D., Swanson, F. J., Stocks, B. J. et Michael Wotton, B. (2001). Climate change and forest disturbances. *BioScience*, *51*(9), 723. https://doi.org/10.1641/0006-3568(2001)051[0723:CCAFD]2.0.CO;2

D'Amato, A. W., Bradford, J. B., Fraver, S. et Palik, B. J. (2011). Forest management for mitigation and adaptation to climate change: Insights from long-term silviculture experiments. *Forest Ecology and Management*, *262*(5), 803-816.

D'Amato, A. W., Bradford, J. B., Fraver, S. et Palik, B. J. (2013). Effects of thinning on drought vulnerability and climate response in north temperate forest ecosystems. *Ecological Applications*, *23*(8), 1735-1742. https://doi.org/10.1890/13-0677.1

da Rosa, C. M. et Marques, M. C. M. (2021). How are biodiversity and carbon stock recovered during tropical forest restoration? Supporting the ecological paradigms and political context involved. *Journal for Nature Conservation*, 126115. https://doi.org/10.1016/j.jnc.2021.126115

Davis, E. C., Sohngen, B. et Lewis, D. J. (2022). The effect of carbon fertilization on naturally regenerated and planted US forests. *Nature Communications*, *13*(1), 5490. https://doi.org/10.1038/s41467-022-33196-x

de Bello, F., Lavorel, S., Hallett, L. M., Valencia, E., Garnier, E., Roscher, C., Conti, L., Galland, T., Goberna, M., Májeková, M., Montesinos-Navarro, A., Pausas, J. G., Verdú, M., E-Vojtkó, A., Götzenberger, L. et Lepš, J. (2021). Functional trait effects on ecosystem stability: assembling the jigsaw puzzle. *Trends in Ecology & Evolution*, *36*(9), 822-836. https://doi.org/10.1016/j.tree.2021.05.001

Deschênes, J.-F. (2022, 1er novembre). Québec fermera d'autres chemins forestiers pour protéger le caribou. *Radio Canada*. https://ici.radio-canada.ca/nouvelle/1929161/chemin-foret-reserve-faunique-matane-destruction-demantelement

Dhiaulhaq, A. et McCarthy, J. F. (2020). Indigenous Rights and Agrarian Justice Framings in Forest Land Conflicts in Indonesia. *The Asia Pacific Journal of Anthropology*, *21*(1), 34-54. https://doi.org/10.1080/14442213.2019.1670243

Diaci, J., Kerr, G. et O'hara, K. (2011). Twenty-first century forestry: integrating ecologically based, uneven-aged silviculture with increased demands on forests. *Forestry: An International Journal of Forest Research*, *84*(5), 463-465. https://doi.org/10.1093/forestry/cpr053

Díaz, S., Hector, A. et Wardle, D. A. (2009). Biodiversity in forest carbon sequestration initiatives: not just a side benefit. *Current Opinion in Environmental Sustainability*, *1*(1), 55-60. https://doi.org/10.1016/j.cosust.2009.08.001

Díaz, S., Purvis, A., Cornelissen, J. H. C., Mace, G. M., Donoghue, M. J., Ewers, R. M., Jordano, P. et Pearse, W. D. (2013). Functional traits, the phylogeny of function, and ecosystem service vulnerability. *Ecology and Evolution*, *3*(9), 2958-2975. https://doi.org/10.1002/ece3.601

Dieler, J., Uhl, E., Biber, P., Müller, J., Rötzer, T. et Pretzsch, H. (2017). Effect of forest stand management on species composition, structural diversity, and productivity in the temperate zone of Europe. *European Journal of Forest Research*, *136*(4), 739-766. https://doi.org/10.1007/s10342-017-1056-1

Dijkstra, E. W. (1959). A note on two problems in connexion with graphs. *Numerische Mathematik*, *1*(1), 269-271. https://doi.org/10.1007/BF01386390

Doelman, J. C., Stehfest, E., van Vuuren, D. P., Tabeau, A., Hof, A. F., Braakhekke, M. C., Gernaat, D. E. H. J., van den Berg, M., van Zeist, W.-J., Daioglou, V., van Meijl, H. et Lucas, P. L. (2020). Afforestation for climate change mitigation: Potentials, risks and trade-offs. *Global Change Biology*, *26*(3), 1576-1591. https://doi.org/10.1111/gcb.14887

D'Orangeville, L., St-Laurent, M.-H., Boisvert-Marsh, L., Zhang, X., Bastille-Rousseau, G. et Itter, M. (2023). Current Symptoms of Climate Change in Boreal Forest Trees and Wildlife. Dans M. M. Girona, H. Morin, S. Gauthier et Y. Bergeron (dir.), *Boreal Forests in the Face of Climate Change: Sustainable Management* (p. 747-771). Springer International Publishing. https://doi.org/10.1007/978-3-031-15988-6_30

Doucet, R. et Côte, M. (2009). *Manuel de foresterie* (Multimondes).

Drever, C. R., Cook-Patton, S. C., Akhter, F., Badiou, P. H., Chmura, G. L., Davidson, S. J., Desjardins, R. L., Dyk, A., Fargione, J. E., Fellows, M., Filewod, B., Hessing-Lewis, M., Jayasundara, S., Keeton, W. S., Kroeger, T., Lark, T. J., Le, E., Leavitt, S. M., LeClerc, M.-E., … Kurz, W. A. (2021). Natural climate solutions for Canada. *Science Advances*, *7*(23), eabd6034. https://doi.org/10.1126/sciadv.abd6034

Driscoll, D. A., Armenteras, D., Bennett, A. F., Brotons, L., Clarke, M. F., Doherty, T. S., Haslem, A., Kelly, L. T., Sato, C. F., Sitters, H., Aquilué, N., Bell, K., Chadid, M., Duane, A., Meza-Elizalde, M. C., Giljohann, K. M., González, T. M., Jambhekar, R., Lazzari, J., … Wevill, T. (2021). How fire interacts with habitat loss and fragmentation. *Biological Reviews*, *96*(3), 976-998. https://doi.org/10.1111/brv.12687

Eberhart, K. E. et Woodard, P. M. (1987). Distribution of residual vegetation associated with large fires in Alberta. *Canadian Journal of Forest Research*, *17*(10), 1207-1212.

Ecological Stratification Working Group, Biological Resources Research et State of the Environment Directorate of Canada. (1996). *A national ecological framework for Canada*. Centre for Land and Biological Resources Research; Hull, Quebec: State of the Environment Directorate.

Edwards, D. P., Gilroy, J. J., Woodcock, P., Edwards, F. A., Larsen, T. H., Andrews, D. J. R., Derhé, M. A., Docherty, T. D. S., Hsu, W. W., Mitchell, S. L., Ota, T., Williams, L. J., Laurance, W. F., Hamer, K. C. et Wilcove, D. S. (2014). Land-sharing versus land-sparing logging: reconciling timber extraction with biodiversity conservation. *Global Change Biology*, *20*(1), 183-191. https://doi.org/10.1111/gcb.12353

Ekroos, J., Ödman, A. M., Andersson, G. K. S., Birkhofer, K., Herbertsson, L., Klatt, B. K., Olsson, O., Olsson, P. A., Persson, A. S., Prentice, H. C., Rundlöf, M. et Smith, H. G. (2016). Sparing Land for Biodiversity at Multiple Spatial Scales. *Frontiers in Ecology and Evolution*, *3*. https://www.frontiersin.org/articles/10.3389/fevo.2015.00145

Elkin, C., Gutiérrez, A. G., Leuzinger, S., Manusch, C., Temperli, C., Rasche, L. et Bugmann, H. (2013). A 2 °C warmer world is not safe for ecosystem services in the European Alps. *Global Change Biology*, *19*(6), 1827-1840. https://doi.org/10.1111/gcb.12156

Ellison, D., Morris, C. E., Locatelli, B., Sheil, D., Cohen, J., Murdiyarso, D., Gutierrez, V., Noordwijk, M. van, Creed, I. F., Pokorny, J., Gaveau, D., Spracklen, D. V., Tobella, A. B., Ilstedt, U., Teuling, A. J., Gebrehiwot, S. G., Sands, D. C., Muys, B., Verbist, B., … Sullivan, C. A. (2017). Trees, forests and water: Cool insights for a hot world. *Global Environmental Change*, *43*, 51-61. https://doi.org/10.1016/j.gloenvcha.2017.01.002

Endicott, S., Lochhead, K., Hughes, J. et Kirby, P. (2023). *roads: Road network projection* [manual]. https://CRAN.R-project.org/package=roads

Ennos, R. (2021). *The age of wood: our most useful material and the construction of civilization*. Simon and Schuster.

Epstein, R., Weintraub, A., Sapunar, P., Nieto, E., Sessions, J. B., Sessions, J., Bustamante, F. et Musante, H. (2006). A Combinatorial Heuristic Approach for Solving Real-Size Machinery Location and Road Design Problems in Forestry Planning. *Operations Research*, *54*(6), 1017-1027. https://doi.org/10.1287/opre.1060.0331

Espelta, J. M., Cruz-Alonso, V., Alfaro-Sánchez, R., Hampe, A., Messier, C. et Pino, J. (2020). Functional diversity enhances tree growth and reduces herbivory damage in secondary broadleaf forests, but does not influence resilience to drought. *Journal of Applied Ecology*, *57*(12), 2362-2372. https://doi.org/10.1111/1365-2664.13728

Evans, A. M. et Perschel, R. (2009). A review of forestry mitigation and adaptation strategies in the Northeast U.S. *Climatic Change*, *96*(1-2), 167-183. https://doi.org/10.1007/s10584-009-9569-3

Evans, M. R. (2012). Modelling ecological systems in a changing world. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *367*(1586), 181-190. https://doi.org/10.1098/rstb.2011.0172

Fahrig, L. (2017). Ecological responses to habitat fragmentation per se. *Annual Review of Ecology, Evolution, and Systematics*, *48*, 1-23.

FAO. (2014). *State of the world's forests: enhancing the socioeconomic benefits from forests*. https://www.fao.org/3/i3710e/i3710e.pdf

FAO. (2020). *Global Forest Resources Assessment 2020*. FAO. https://doi.org/10.4060/ca9825en

FAO et OAA. (2012). *State of the world's forests 2012*. Food and Agriculture Organization of the United Nations.

FAO et UNEP. (2020). *The State of the World's Forests 2020*. https://doi.org/10.4060/ca8642en

FAOSTAT. (2019a). *FAOSTAT Database on Agriculture*. Food and Agriculture Organization of the United Nations. http://faostat.fao.org/

FAOSTAT. (2019b). *FAOSTAT Forestry Production and Trade Database*. Food and Agriculture Organization of the United Nations. http://www.fao.org/faostat/en/#data/FO

Fargione, J. E., Bassett, S., Boucher, T., Bridgham, S. D., Conant, R. T., Cook-Patton, S. C., Ellis, P. W., Falcucci, A., Fourqurean, J. W., Gopalakrishna, T., Gu, H., Henderson, B., Hurteau, M. D., Kroeger, K. D., Kroeger, T., Lark, T. J., Leavitt, S. M., Lomax, G., McDonald, R. I., … Griscom, B. W. (2018). Natural climate solutions for the United States. *Science Advances*, *4*(11), eaat1869. https://doi.org/10.1126/sciadv.aat1869

Farrar, J. (1995). Trees in Canada. Published by Fitzhenry & Whiteside Ltd. and Canadian Forest Service. *Natural Resources Canada, in cooperation with the Canada Communication Group—Publishing, Supply and Services Canada*.

Fedrowitz, K., Koricheva, J., Baker, S. C., Lindenmayer, D. B., Palik, B., Rosenvald, R., Beese, W., Franklin, J. F., Kouki, J., Macdonald, E., Messier, C., Sverdrup-Thygeson, A. et Gustafsson, L. (2014). REVIEW: Can retention forestry help conserve biodiversity? A meta-analysis. *Journal of Applied Ecology*, *51*(6), 1669-1679. https://doi.org/10.1111/1365-2664.12289

Felipe-Lucia, M. R., Soliveres, S., Penone, C., Manning, P., van der Plas, F., Boch, S., Prati, D., Ammer, C., Schall, P., Gossner, M. M., Bauhus, J., Buscot, F., Blaser, S., Blüthgen, N., de Frutos, A., Ehbrecht, M., Frank, K., Goldmann, K., Hänsel, F., … Allan, E. (2018). Multiple forest attributes underpin the supply of multiple ecosystem services. *Nature Communications*, *9*(1), 4839. https://doi.org/10.1038/s41467-018-07082-4

Feng, Y., Negrón-Juárez, R. I., Romps, D. M. et Chambers, J. Q. (2023). Amazon windthrow disturbances are likely to increase with storm frequency under global warming. *Nature Communications*, *14*(1), 101. https://doi.org/10.1038/s41467-022-35570-1

Fenton, N. J., Simard, M. et Bergeron, Y. (2009). Emulating natural disturbances: the role of silviculture in creating even-aged and complex structures in the black spruce boreal forest of eastern North America. *Journal of Forest Research*, *14*(5), 258-267. https://doi.org/10.1007/s10310-009-0134-8

Fernow, B. E. (1909). *A brief history of forestry in Europe, the United States and other countries.* University Press. https://www.biodiversitylibrary.org/bibliography/23491

Fichtner, A., Schnabel, F., Bruelheide, H., Kunz, M., Mausolf, K., Schuldt, A., Härdtle, W. et von Oheimb, G. (2020). Neighbourhood diversity mitigates drought impacts on tree growth. *Journal of Ecology*, *108*(3), 865-875. https://doi.org/10.1111/1365-2745.13353

Fischer, J., Abson, D. J., Butsic, V., Chappell, M. J., Ekroos, J., Hanspach, J., Kuemmerle, T., Smith, H. G. et Wehrden, H. von. (2014). Land Sparing Versus Land Sharing: Moving Forward. *Conservation Letters*, *7*(3), 149-157. https://doi.org/10.1111/conl.12084

Fletcher, R. J., Didham, R. K., Banks-Leite, C., Barlow, J., Ewers, R. M., Rosindell, J., Holt, R. D., Gonzalez, A., Pardini, R., Damschen, E. I., Melo, F. P. L., Ries, L., Prevedello, J. A., Tscharntke, T., Laurance,

W. F., Lovejoy, T. et Haddad, N. M. (2018). Is habitat fragmentation good for biodiversity? *Biological Conservation*, *226*, 9-15. https://doi.org/10.1016/j.biocon.2018.07.022

Flores, B. M., Staal, A., Jakovac, C. C., Hirota, M., Holmgren, M. et Oliveira, R. S. (2020). Soil erosion as a resilience drain in disturbed tropical forests. *Plant and Soil*, *450*(1-2), 11-25. https://doi.org/10.1007/s11104-019-04097-8

Folke, C. (2016). Resilience (Republished). *Ecology and Society*, *21*(4). https://www.jstor.org/stable/26269991

Forest Peoples Programme. (2014). *Securing Forests, Securing rights: Report of the International Workshop on Deforestation and the Rights of Forest Peoples*. https://doi.org/10.1163/2210-7975_HRD-5540-2014002

Forgy, E. W. (1965). Cluster analysis of multivariate data: efficiency versus interpretability of classifications. *biometrics*, *21*, 768-769.

Forman, R. T. T. (2005). *Good and bad places for roads: effects of varying road and natural pattern on habitat loss, degradation, and fragmentation*. https://escholarship.org/uc/item/8xb0z15w

Fowler, D., Coyle, M., Skiba, U., Sutton, M. A., Cape, J. N., Reis, S., Sheppard, L. J., Jenkins, A., Grizzetti, B., Galloway, J. N., Vitousek, P., Leach, A., Bouwman, A. F., Butterbach-Bahl, K., Dentener, F., Stevenson, D., Amann, M. et Voss, M. (2013). The global nitrogen cycle in the twenty-first century. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *368*(1621), 20130164. https://doi.org/10.1098/rstb.2013.0164

França, F. M., Frazão, F. S., Korasaki, V., Louzada, J. et Barlow, J. (2017). Identifying thresholds of logging intensity on dung beetle communities to improve the sustainable management of Amazonian tropical forests. *Biological Conservation*, *216*, 115-122. https://doi.org/10.1016/j.biocon.2017.10.014

Franklin, J. F. (1989). The “new forestry”. *Journal of Soil and Water Conservation*, *44*(6), 549-549.

Franklin, J. F., Johnson, K. N. et Johnson, D. L. (2018). *Ecological Forest Management*. Waveland Press.

Fraser, J. S., He, H. S. et Wang, W. J. (2020). *Density-Succession v1.0 Extension User Guide*. https://github.com/LANDIS-II-Foundation/Extension-Density-Succession/releases/download/v1.0/LANDIS-II.Density-Succession.v1.0.User.Guide.pdf

Frey, S. J. K., Hadley, A. S., Johnson, S. L., Schulze, M., Jones, J. A. et Betts, M. G. (2016). Spatial models reveal the microclimatic buffering capacity of old-growth forests. *Science Advances*, *2*(4), e1501392. https://doi.org/10.1126/sciadv.1501392

Gauthier, S., Bernier, P., Kuuluvainen, T., Shvidenko, A. Z. et Schepaschenko, D. G. (2015). Boreal forest health and global change. *Science*, *349*(6250), 819-822. https://doi.org/10.1126/science.aaa9092

Gauthier, S. et Vaillancourt, M.-A. (2008). *Aménagement écosystémique en forêt boréale*. PUQ.

Giampietro, M. et Mayumi, K. (2018). Unraveling the Complexity of the Jevons Paradox: The Link Between Innovation, Efficiency, and Sustainability. *Frontiers in Energy Research*, *6*. https://www.frontiersin.org/articles/10.3389/fenrg.2018.00026

Giles-Hansen, K. et Wei, X. (2022). Cumulative disturbance converts regional forests into a substantial carbon source. *Environmental Research Letters*, *17*(4), 044049. https://doi.org/10.1088/1748-9326/ac5e69

Gillis, A. M. (1990). The new forestry. *BioScience*, *40*(8), 558-562.

Girardin, P., Valeria, O. et Girard, F. (2022). *Measuring Spatial and Temporal Gravelled Forest Road Degradation in the Boreal Forest*, 18.

Girona, M. M., Aakala, T., Aquilué, N., Bélisle, A.-C., 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. Dans M. M. Girona, H. Morin, S. Gauthier et Y. Bergeron (dir.), *Boreal Forests in the Face of Climate Change: Sustainable Management* (p. 773-837). Springer International Publishing. https://doi.org/10.1007/978-3-031-15988-6_31

Gladstone, W. T. et Thomas Ledig, F. (1990). Reducing pressure on natural forests through high-yield forestry. *Forest Ecology and Management*, *35*(1), 69-78. https://doi.org/10.1016/0378-1127(90)90232-Z

Gonzalez, J. (1990). *Wood density of Canadian tree species.* Forestry Canada. https://cfs.nrcan.gc.ca/pubwarehouse/pdfs/11744.pdf

Götmark, F., Paltto, H., Nordén, B. et Götmark, E. (2005). Evaluating partial cutting in broadleaved temperate forest under strong experimental control: Short-term effects on herbaceous plants. *Forest Ecology and Management*, *214*(1-3), 124-141. https://doi.org/10.1016/j.foreco.2005.03.052

Gough, C. M., Vogel, C. S., Harrold, K. H., George, K. et Curtis, P. S. (2007). The legacy of harvest and fire on ecosystem carbon storage in a north temperate forest. *Global Change Biology*, *13*(9), 1935-1949. https://doi.org/10.1111/j.1365-2486.2007.01406.x

Gouvernement du Québec. (2015). Adresses Québec: Guide de l'utilisateur. *Québec (Québec): Ministère de l'Énergie et des Ressources naturelles, Direction de la cartographie topographique*, 60.

Gouvernement du Québec. *Règlement sur l'aménagement durable des forêts du domaine de l'État*. RLRQ c A-18.1, r. 0.01. http://legisquebec.gouv.qc.ca/fr/ShowDoc/cr/A-18.1,%20r.%200.01/

Government of Quebec. (s. d.). *Outil de comparaison des essences*. Récupéré le 22 décembre 2023 de http://www2.publicationsduquebec.gouv.qc.ca/essences/comparaison.php

Gower, J. C. (1971). A general coefficient of similarity and some of its properties. *Biometrics. Journal of the International Biometric Society*, 857-871.

Graham-Sauvé, L., Work, T. T., Kneeshaw, D. et Messier, C. (2013). Shelterwood and multicohort management have similar initial effects on ground beetle assemblages in boreal forests. *Forest Ecology and Management*, *306*, 266-274. https://doi.org/10.1016/j.foreco.2013.06.019

Green, R. E., Cornell, S. J., Scharlemann, J. P. W. et Balmford, A. (2005). Farming and the Fate of Wild Nature. *Science*, *307*(5709), 550-555. https://doi.org/10.1126/science.1106049

Griscom, B. W., Adams, J., Ellis, P. W., Houghton, R. A., Lomax, G., Miteva, D. A., Schlesinger, W. H., Shoch, D., Siikamäki, J. V., Smith, P., Woodbury, P., Zganjar, C., Blackman, A., Campari, J., Conant, R. T., Delgado, C., Elias, P., Gopalakrishna, T., Hamsik, M. R., … Fargione, J. (2017). Natural climate solutions. *Proceedings of the National Academy of Sciences*, *114*(44), 11645-11650. https://doi.org/10.1073/pnas.1710465114

Griscom, B. W., Goodman, R. C., Burivalova, Z. et Putz, F. E. (2018). Carbon and Biodiversity Impacts of Intensive Versus Extensive Tropical Forestry. *Conservation Letters*, *11*(1), e12362. https://doi.org/10.1111/conl.12362

Gronewold, C. A., D'Amato, A. W. et Palik, B. J. (2010). The influence of cutting cycle and stocking level on the structure and composition of managed old-growth northern hardwoods. *Forest Ecology and Management*, *259*(6), 1151-1160. https://doi.org/10.1016/j.foreco.2010.01.001

Grossiord, C. (2020). Having the right neighbors: how tree species diversity modulates drought impacts on forests. *New Phytologist*, *228*(1), 42-49. https://doi.org/10.1111/nph.15667

Grossiord, C., Granier, A., Ratcliffe, S., Bouriaud, O., Bruelheide, H., Chećko, E., Forrester, D. I., Dawud, S. M., Finér, L., Pollastrini, M., Scherer-Lorenzen, M., Valladares, F., Bonal, D. et Gessler, A. (2014). Tree diversity does not always improve resistance of forest ecosystems to drought. *Proceedings of the National Academy of Sciences*, *111*(41), 14812-14815. https://doi.org/10.1073/pnas.1411970111

Groupe DDM et MFFP du Québec. (2020). *Enquête sur les couts d'opération forestière dans les forêts du domaine de l'État ainsi que sur les couts et revenus de l'industrie du sciage du Québec 2019*.

Gucinski, H. (2001). *Forest Roads: A Synthesis of Scientific Information*. DIANE Publishing.

Guldin, J. M. (1996). The Role of Uneven-Aged Silviculture in the Context of Ecosystem Management. *Western Journal of Applied Forestry*, *11*(1), 4-12.

Gustafson, E. J. (1998). *Quantifying Landscape Spatial Pattern: What Is the State of the Art?*, 14.

Gustafson, E. J. (2007). Relative influence of the components of timber harvest strategies on landscape pattern. *Forest Science*, *53*(5), 556-561.

Gustafson, E. J., Miranda, B. R., Sturtevant, B. R. et Zhou, Z. (2023). *PnET-Succession v 5.1: Comprehensive description of an ecophysiological succession extension within the LANDIS-II forest landscape model*.

Gutsch, M., Lasch-Born, P., Kollas, C., Suckow, F. et Reyer, C. P. O. (2018). Balancing trade-offs between ecosystem services in Germany's forests under climate change. *Environmental Research Letters*, *13*(4), 045012. https://doi.org/10.1088/1748-9326/aab4e5

Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., Lovejoy, T. E., Sexton, J. O., Austin, M. P., Collins, C. D., Cook, W. M., Damschen, E. I., Ewers, R. M., Foster, B. L., Jenkins, C. N., King, A. J., Laurance, W. F., Levey, D. J., Margules, C. R., … Townshend, J. R. (2015). Habitat fragmentation and its lasting impact on Earth's ecosystems. *Science Advances*, *1*(2), e1500052-e1500052. https://doi.org/10.1126/sciadv.1500052

Hammond, W. M., Williams, A. P., Abatzoglou, J. T., Adams, H. D., Klein, T., López, R., Sáenz-Romero, C., Hartmann, H., Breshears, D. D. et Allen, C. D. (2022). Global field observations of tree die-off reveal hotter-drought fingerprint for Earth's forests. *Nature Communications*, *13*(1), 1761. https://doi.org/10.1038/s41467-022-29289-2

Hanes, C. C., Wang, X., Jain, P., Parisien, M.-A., Little, J. M. et Flannigan, M. D. (2019). Fire-regime changes in Canada over the last half century. *Canadian Journal of Forest Research*, *49*(3), 256-269. https://doi.org/10.1139/cjfr-2018-0293

Hann, D. W. et Bare, B. B. (1979). Uneven-Aged Forest Management: State of the Art (or Science?). *Forestry*, 14.

Hardy, C. (2019). *Forest Roads Network plugin for QGIS* (version 1.6). https://plugins.qgis.org/plugins/ForestRoadsNetworksUPLOAD/

Hardy, C. (2021). *Forest Roads Simulation v1.1 LANDIS-II Extension User Guide*. https://github.com/Klemet/LANDIS-II-Forest-Roads-Simulation-module/blob/master/LANDIS-II%20Forest%20Roads%20Simulation%20v1.1%20User%20Guide.pdf

Hardy, C. (2022). *Magic Harvest LANDIS-II extension* (version 1.1). https://github.com/Klemet/LANDIS-II-Magic-Harvest

Hardy, C., Messier, C., Boulanger, Y., Cyr, D. et Filotas, É. (2023a). Land sparing and sharing patterns in forestry: exploring even-aged and uneven-aged management at the landscape scale. *Landscape Ecology*, *38*(11), 2815-2838. https://doi.org/10.1007/s10980-023-01742-7

Hardy, C., Messier, C., Valeria, O. et Filotas, E. (2023b). A LANDIS-II extension for simulating forest road networks. *Canadian Journal of Forest Research*, *53*(7), 503-518. https://doi.org/10.1139/cjfr-2022-0306

Harris, S. H. et Betts, M. G. (2023). Selecting among land sparing, sharing and Triad in a temperate rainforest depends on biodiversity and timber production targets. *Journal of Applied Ecology*, *60*(4), 737-750. https://doi.org/10.1111/1365-2664.14385

Harvey, B. D., Leduc, A., Gauthier, S. et 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/10.1016/S0378-1127(01)00573-4

Haskell, D. G. (2000). Effects of Forest Roads on Macroinvertebrate Soil Fauna of the Southern Appalachian Mountains. *Conservation Biology*, *14*(1), 57-63. https://doi.org/10.1046/j.1523-1739.2000.99232.x

Haverd, V., Smith, B., Canadell, J. G., Cuntz, M., Mikaloff-Fletcher, S., Farquhar, G., Woodgate, W., Briggs, P. R. et Trudinger, C. M. (2020). Higher than expected CO2 fertilization inferred from leaf to global observations. *Global Change Biology*, *26*(4), 2390-2402. https://doi.org/10.1111/gcb.14950

Hawley, R. C. et Smith, D. M. (1962). *The practice of silviculture [by] David Martyn Smith. 7th ed. The continuation of a work published as a 1st ed. in Mar. 1921 by Ralph C. Hawley.* Wiley.

He, H. S. et Mladenoff, D. J. (1999). Spatially explicit and stochastic simulation of forest-landscape fire disturbance and succession. *Ecology*, *80*(1), 81-99.

Heinimann, H. R. (2017). Forest Road Network and Transportation Engineering – State and Perspectives. *Croat. j. for. eng.*, 19.

Hemström, K., Mahapatra, K. et Gustavsson, L. (2014). Public Perceptions and Acceptance of Intensive Forestry in Sweden. *AMBIO*, *43*(2), 196-206. https://doi.org/10.1007/s13280-013-0411-9

Hesselbarth, M. H., Sciaini, M., With, K. A., Wiegand, K. et Nowosad, J. (2019). landscapemetrics: an open-source R tool to calculate landscape metrics. *Ecography*, *42*(10), 1648-1657.

Hilbert, J. et Wiensczyk, A. (2007). Old-growth definitions and management: A literature review. *Journal of Ecosystems and Management*, *8*(1), 18.

Hiltner, U., Huth, A., Hérault, B., Holtmann, A., Bräuning, A. et Fischer, R. (2021). Climate change alters the ability of neotropical forests to provide timber and sequester carbon. *Forest Ecology and Management*, *492*, 119166. https://doi.org/10.1016/j.foreco.2021.119166

Himes, A., Bauhus, J., Adhikari, S., Barik, S. K., Brown, H., Brunner, A., Burton, P. J., Coll, L., D'Amato, A. W., Diaci, J., Dorji, Y., Foli, E. G., Ganz, D. J., Hall, J. S., Keenan, R., Lu, Y., Messier, C., Munanura, I., Piotto, D., … Puettmann, K. J. (2023). Forestry in the Face of Global Change: Results of a Global Survey of Professionals. *Current Forestry Reports*, *9*(6), 473-489. https://doi.org/10.1007/s40725-023-00205-1

Himes, A., Betts, M., Messier, C. et Seymour, R. (2022). Perspectives: Thirty years of triad forestry, a critical clarification of theory and recommendations for implementation and testing. *Forest Ecology and Management*, *510*, 120103. https://doi.org/10.1016/j.foreco.2022.120103

Hof, A. R., Dymond, C. C. et Mladenoff, D. J. (2017). Climate change mitigation through adaptation: the effectiveness of forest diversification by novel tree planting regimes. *Ecosphere*, *8*(11), e01981. https://doi.org/10.1002/ecs2.1981

Holland, E. P., Aegerter, J. N., Dytham, C. et Smith, G. C. (2007). Landscape as a model: the importance of geometry. *PLoS Computational Biology*, *3*(10), e200.

Holling, C. S. et Meffe, G. K. (1996). Command and control and the pathology of natural resource management. *Conservation biology*, *10*(2), 328-337.

Hoover, C. M. et Smith, J. E. (2023). Aboveground live tree carbon stock and change in forests of conterminous United States: influence of stand age. *Carbon Balance and Management*, *18*(1), 7. https://doi.org/10.1186/s13021-023-00227-z

Hörl, J., Keller, K. et Yousefpour, R. (2020). Reviewing the performance of adaptive forest management strategies with robustness analysis. *Forest Policy and Economics*, *119*, 102289. https://doi.org/10.1016/j.forpol.2020.102289

Houndode, D. J., Krause, C. et Morin, H. (2021). Predicting balsam fir mortality in boreal stands affected by spruce budworm. *Forest Ecology and Management*, *496*, 119408. https://doi.org/10.1016/j.foreco.2021.119408

Hua, F., Bruijnzeel, L. A., Meli, P., Martin, P. A., Zhang, J., Nakagawa, S., Miao, X., Wang, W., McEvoy, C., Peña-Arancibia, J. L., Brancalion, P. H. S., Smith, P., Edwards, D. P. et Balmford, A. (2022). The biodiversity and ecosystem service contributions and trade-offs of forest restoration approaches. *Science*, *376*(6595), 839-844. https://doi.org/10.1126/science.abl4649

Hudiburg, T. W., Law, B. E., Moomaw, W. R., Harmon, M. E. et Stenzel, J. E. (2019). Meeting GHG reduction targets requires accounting for all forest sector emissions. *Environmental Research Letters*, *14*(9), 095005. https://doi.org/10.1088/1748-9326/ab28bb

Hunt, L. M., Lemelin, R. H. et Saunders, K. C. (2009). Managing Forest Road Access on Public Lands: A Conceptual Model of Conflict. *Society & Natural Resources*, *22*(2), 128-142. https://doi.org/10.1080/08941920801922604

Irauschek, F., Rammer, W. et Lexer, M. J. (2017). Evaluating multifunctionality and adaptive capacity of mountain forest management alternatives under climate change in the Eastern Alps. *European Journal of Forest Research*, *136*(5), 1051-1069. https://doi.org/10.1007/s10342-017-1051-6

Jactel, H., Moreira, X. et Castagneyrol, B. (2021). Tree Diversity and Forest Resistance to Insect Pests: Patterns, Mechanisms, and Prospects. *Annual Review of Entomology*, *66*(1), 277-296. https://doi.org/10.1146/annurev-ento-041720-075234

Jain, P., Barber, Q. E., Taylor, S., Whitman, E., Acuna, D. C., Boulanger, Y., Chavardès, R. D., Chen, J., Englefield, P., Flannigan, M., Girardin, M. P., Hanes, C. C., Little, J., Morrison, K., Skakun, R. S., Thompson, D. K., Wang, X. et Parisien, M.-A. (2024, 28 février). *Canada Under Fire – Drivers and Impacts of the Record-Breaking 2023 Wildfire Season*. https://doi.org/10.22541/essoar.170914412.27504349/v1

Jain, P., Castellanos-Acuna, D., Coogan, S. C. P., Abatzoglou, J. T. et Flannigan, M. D. (2022). Observed increases in extreme fire weather driven by atmospheric humidity and temperature. *Nature Climate Change*, *12*(1), 63-70. https://doi.org/10.1038/s41558-021-01224-1

Jakobsson, R., Olofsson, E. et Ambrose-Oji, B. (2021). Stakeholder perceptions, management and impacts of forestry conflicts in southern Sweden. *Scandinavian Journal of Forest Research*, *36*(1), 68-82. https://doi.org/10.1080/02827581.2020.1854341

James, A. R. C. et Stuart-Smith, A. K. (2000). Distribution of Caribou and Wolves in Relation to Linear Corridors. *The Journal of Wildlife Management*, *64*(1), 154-159. https://doi.org/10.2307/3802985

Jandl, R., Spathelf, P., Bolte, A. et Prescott, C. E. (2019). Forest adaptation to climate change—is non-management an option? *Annals of Forest Science*, *76*(2), 1-13. https://doi.org/10.1007/s13595-019-0827-x

Jayen, K., Leduc, A. et Bergeron, Y. (2006). Effect of fire severity on regeneration success in the boreal forest of northwest Quebec, Canada. *Ecoscience*, *13*(2), 143-151.

Joelsson, K., Hjältén, J., Work, T., Gibb, H., Roberge, J.-M. et Löfroth, T. (2017). Uneven-aged silviculture can reduce negative effects of forest management on beetles. *Forest Ecology and Management*, *391*, 436-445. https://doi.org/10.1016/j.foreco.2017.02.006

Johnston, C. M. T. et Radeloff, V. C. (2019). Global mitigation potential of carbon stored in harvested wood products. *Proceedings of the National Academy of Sciences*, *116*(29), 14526-14531. https://doi.org/10.1073/pnas.1904231116

Jost, L. (2006). Entropy and diversity. *Oikos*, *113*(2), 363-375. https://doi.org/10.1111/j.2006.0030-1299.14714.x

Judson, O. P. (1994). The rise of the individual-based model in ecology. *Trends in Ecology & Evolution*, *9*(1), 9-14. https://doi.org/10.1016/0169-5347(94)90225-9

Jutras, S. (2022, 16 mars). *Les chemins forestiers abandonnés par Québec endommagent les milieux aquatiques*. The Conversation. http://theconversation.com/les-chemins-forestiers-abandonnes-par-quebec-endommagent-les-milieux-aquatiques-176414

Kaarakka, L., Cornett, M., Domke, G., Ontl, T. et Dee, L. E. (2021). Improved forest management as a natural climate solution: A review. *Ecological Solutions and Evidence*, *2*(3), e12090. https://doi.org/10.1002/2688-8319.12090

Kaimowitz, D. et Angelsen, A. (1998). *Economic Models of Tropical Deforestation: A Review*. CIFOR.

Kalabokidis, K. D. et Wakimoto, R. H. (1992). Prescribed burning in uneven-aged stand management of ponderosa pine/douglas fir forests. *Journal of Environmental Management*, *34*(3), 221-235. https://doi.org/10.1016/S0301-4797(05)80153-7

Kanowski, P. J. et Williams, K. J. H. (2009). The reality of imagination: Integrating the material and cultural values of old forests. *Forest Ecology and Management*, *258*(4), 341-346. https://doi.org/10.1016/j.foreco.2009.01.011

Karjalainen, E., Sarjala, T. et Raitio, H. (2010). Promoting human health through forests: overview and major challenges. *Environmental Health and Preventive Medicine*, *15*(1), 1-8. https://doi.org/10.1007/s12199-008-0069-2

Karlson, M. et Mörtberg, U. (2015). A spatial ecological assessment of fragmentation and disturbance effects of the Swedish road network. *Landscape and Urban Planning*, *134*, 53-65. https://doi.org/10.1016/j.landurbplan.2014.10.009

Karlsson, J., Rönnqvist, M. et Frisk, M. (2006). RoadOpt: A decision support system for road upgrading in forestry. *Scandinavian Journal of Forest Research*, *21*(S7), 5-15. https://doi.org/10.1080/14004080500487102

Keane, R. E., Hessburg, P. F., Landres, P. B. et Swanson, F. J. (2009). The use of historical range and variability (HRV) in landscape management. *Forest Ecology and Management*, *258*(7), 1025-1037. https://doi.org/10.1016/j.foreco.2009.05.035

Keenan, R. J. (2015). Climate change impacts and adaptation in forest management: a review. *Annals of Forest Science*, *72*(2), 145-167. https://doi.org/10.1007/s13595-014-0446-5

Keenan, R. J. (2017). Climate change and Australian production forests: impacts and adaptation. *Australian Forestry*, *80*(4), 197-207. https://doi.org/10.1080/00049158.2017.1360170

Keenan, R. J. et Kimmins, J. P. (1993). The ecological effects of clear-cutting. *Environmental Reviews*, *1*(2), 121-144. https://doi.org/10.1139/a93-010

Keenan, R. J. et Nitschke, C. (2016). Forest management options for adaptation to climate change: a case study of tall, wet eucalypt forests in Victoria's Central Highlands region. *Australian Forestry*, *79*(2), 96-107. https://doi.org/10.1080/00049158.2015.1130095

Kleinschroth, F., Laporte, N., Laurance, W. F., Goetz, S. J. et Ghazoul, J. (2019). Road expansion and persistence in forests of the Congo Basin. *Nature Sustainability*, *2*(7), 628-634. https://doi.org/10.1038/s41893-019-0310-6

Kneeshaw, D. et Gauthier, S. (2006). Accessibilité forestière accrue : panacée ou boîte de Pandore ? *Téoros. Revue de recherche en tourisme*, *25*(25-3), 36-40.

Knorn, J., Kuemmerle, T., Radeloff, V. C., Keeton, W. S., Gancz, V., Biriş, I.-A., Svoboda, M., Griffiths, P., Hagatis, A. et Hostert, P. (2013). Continued loss of temperate old-growth forests in the Romanian Carpathians despite an increasing protected area network. *Environmental Conservation*, *40*(2), 182-193.

Koivula, M. J. (2005). Effects of Forest Roads on Spatial Distribution of Boreal Carabid Beetles (Coleoptera: Carabidae). *The Coleopterists Bulletin*, *59*(4), 465-487. https://doi.org/10.1649/815.1

Krawchuk, M. A., Haire, S. L., Coop, J., Parisien, M.-A., Whitman, E., Chong, G. et Miller, C. (2016). Topographic and fire weather controls of fire refugia in forested ecosystems of northwestern North America. *Ecosphere*, *7*(12), e01632. https://doi.org/10.1002/ecs2.1632

Kreidenweis, U., Humpenöder, F., Stevanović, M., Bodirsky, B. L., Kriegler, E., Lotze-Campen, H. et Popp, A. (2016). Afforestation to mitigate climate change: impacts on food prices under consideration of albedo effects. *Environmental Research Letters*, *11*(8), 085001. https://doi.org/10.1088/1748-9326/11/8/085001

Kremen, C. (2015). Reframing the land-sparing/land-sharing debate for biodiversity conservation. *Annals of the New York Academy of Sciences*, *1355*(1), 52-76. https://doi.org/10.1111/nyas.12845

Kurz, W. A., Stinson, G. et Rampley, G. (2007). Could increased boreal forest ecosystem productivity offset carbon losses from increased disturbances? *Philosophical Transactions of the Royal Society B: Biological Sciences*, *363*(1501), 2259-2268. https://doi.org/10.1098/rstb.2007.2198

Kurz, W. A., Stinson, G., Rampley, G. J., Dymond, C. C. et Neilson, E. T. (2008). Risk of natural disturbances makes future contribution of Canada’s forests to the global carbon cycle highly uncertain. *Proceedings of the National Academy of Sciences*, *105*(5), 1551-1555. https://doi.org/10.1073/pnas.0708133105

Kuuluvainen, T., Angelstam, P., Frelich, L., Jõgiste, K., Koivula, M., Kubota, Y., Lafleur, B. et Macdonald, E. (2021). Natural Disturbance-Based Forest Management: Moving Beyond Retention and Continuous-Cover Forestry. *Frontiers in Forests and Global Change*, *4*. https://www.frontiersin.org/articles/10.3389/ffgc.2021.629020

Kuuluvainen, T., Tahvonen, O. et Aakala, T. (2012). Even-Aged and Uneven-Aged Forest Management in Boreal Fennoscandia: A Review. *Ambio*, *41*(7), 720-737. https://doi.org/10.1007/s13280-012-0289-y

Labadie, G., Hardy, C., Boulanger, Y., Vanlandeghem, V., Hebblewhite, M. et Fortin, D. (2023). Global change risks a threatened species due to alteration of predator–prey dynamics. *Ecosphere*, *14*(3), e4485. https://doi.org/10.1002/ecs2.4485

Lafleur, B., Fenton, N. J., Simard, M., Leduc, A., Paré, D., Valeria, O. et Bergeron, Y. (2018). Ecosystem management in paludified boreal forests: enhancing wood production, biodiversity, and carbon sequestration at the landscape level. *Forest Ecosystems*, *5*(1), 27. https://doi.org/10.1186/s40663-018-0145-z

LANDIS-II Foundation. (s. d.). *LANDIS-II Forest landscape model - Publications*. Récupéré le 7 septembre 2023 de https://www.landis-ii.org/publications

Larocque, G. R., DesRochers, A., Larchevêque, M., Tremblay, F., Beaulieu, J., Mosseler, A., Major, J. E., Gaussiran, S., Thomas, B. R., Sidders, D., Périnet, P., Kort, J., Labrecque, M., Savoie, P., Masse, S., Bouman, O. T., Kamelchuk, D., Benomar, L., Mamashita, T. et Gagné, P. (2013). Research on hybrid poplars and willow species for fast-growing tree plantations: Its importance for growth and yield, silviculture, policy-making and commercial applications. *The Forestry Chronicle*, *89*(01), 32-41. https://doi.org/10.5558/tfc2013-009

LaRue, E. A., Hardiman, B. S., Elliott, J. M. et Fei, S. (2019). Structural diversity as a predictor of ecosystem function. *Environmental Research Letters*, *14*(11), 114011. https://doi.org/10.1088/1748-9326/ab49bb

Law, B. E., Hudiburg, T. W., Berner, L. T., Kent, J. J., Buotte, P. C. et Harmon, M. E. (2018). Land use strategies to mitigate climate change in carbon dense temperate forests. *Proceedings of the National Academy of Sciences*, *115*(14), 3663-3668. https://doi.org/10.1073/pnas.1720064115

Lawrence, D., Coe, M., Walker, W., Verchot, L. et Vandecar, K. (2022). The Unseen Effects of Deforestation: Biophysical Effects on Climate. *Frontiers in Forests and Global Change*, *5*. https://www.frontiersin.org/articles/10.3389/ffgc.2022.756115

Lee, H., Calvin, K., Dasgupta, D., Krinner, G., Mukherji, A. et Thorne, P. (2023). Synthesis report of the IPCC sixth assessment report (AR6). *Intergovernmental Panel on Climate Change, Geneva, Switzerland*.

Légère, G. (2001). *Étude technico-économique de la stabilisation à la chaux des sols argileux en voirie forestière* [masters, École de technologie supérieure]. https://espace.etsmtl.ca/id/eprint/846/

Lewis, J. L. et Sheppard, S. R. J. (2005). Ancient Values, New Challenges: Indigenous Spiritual Perceptions of Landscapes and Forest Management. *Society & Natural Resources*, *18*(10), 907-920. https://doi.org/10.1080/08941920500205533

Lexer, M. J. et Hönninger, K. (2001). A modified 3D-patch model for spatially explicit simulation of vegetation composition in heterogeneous landscapes. *Forest Ecology and Management*, *144*(1), 43-65. https://doi.org/10.1016/S0378-1127(00)00386-8

Li, W., Guo, W.-Y., Pasgaard, M., Niu, Z., Wang, L., Chen, F., Qin, Y. et Svenning, J.-C. (2023). Human fingerprint on structural density of forests globally. *Nature Sustainability*, 1-12. https://doi.org/10.1038/s41893-022-01020-5

Li, Y., Hu, Y., Li, X. et Xiao, D. (2003). A review on road ecology. *Ying yong sheng tai xue bao= The journal of applied ecology*, *14*(3), 447-452.

Liang, J., Crowther, T. W., Picard, N., Wiser, S., Zhou, M., Alberti, G., Schulze, E.-D., McGuire, A. D., Bozzato, F., Pretzsch, H., de-Miguel, S., Paquette, A., Hérault, B., Scherer-Lorenzen, M., Barrett, C. B., Glick, H. B., Hengeveld, G. M., Nabuurs, G.-J., Pfautsch, S., … Reich, P. B. (2016). Positive biodiversity-productivity relationship predominant in global forests. *Science*, *354*(6309), aaf8957. https://doi.org/10.1126/science.aaf8957

Lindenmayer, D., Franklin, J., Lõhmus, A., Baker, S., Bauhus, J., Beese, W., Brodie, A., Kiehl, B., Kouki, J., Pastur, G. M. et others. (2012). A major shift to the retention approach for forestry can help resolve some global forest sustainability issues. *Conservation Letters*, *5*(6), 421-431.

Liu, X., Trogisch, S., He, J.-S., Niklaus, P. A., Bruelheide, H., Tang, Z., Erfmeier, A., Scherer-Lorenzen, M., Pietsch, K. A., Yang, B., Kühn, P., Scholten, T., Huang, Y., Wang, C., Staab, M., Leppert, K. N., Wirth, C., Schmid, B. et Ma, K. (2018). Tree species richness increases ecosystem carbon storage in subtropical forests. *Proceedings of the Royal Society B: Biological Sciences*, *285*(1885), 20181240. https://doi.org/10.1098/rspb.2018.1240

Liu, Y.-F., Liu, Y., Shi, Z.-H., López-Vicente, M. et Wu, G.-L. (2020). Effectiveness of re-vegetated forest and grassland on soil erosion control in the semi-arid Loess Plateau. *CATENA*, *195*, 104787. https://doi.org/10.1016/j.catena.2020.104787

Lochhead, K. et Muhly, T. B. (2018). *A demo of road simulation in CLUS*. GitHub. https://github.com/bcgov/castor/blob/main/reports/roads/draft-CLUS-roads.md

Long, J. A. et Lawrence, R. L. (2016). Mapping Percent Tree Mortality Due to Mountain Pine Beetle Damage. *Forest Science*, *62*(4), 392-402. https://doi.org/10.5849/forsci.15-046

Lopez, M. C. et Moran, E. F. (2016). The legacy of Elinor Ostrom and its relevance to issues of forest conservation. *Current Opinion in Environmental Sustainability*, *19*, 47-56. https://doi.org/10.1016/j.cosust.2015.12.001

Loreau, M., Barbier, M., Filotas, E., Gravel, D., Isbell, F., Miller, S. J., Montoya, J. M., Wang, S., Aussenac, R., Germain, R., Thompson, P. L., Gonzalez, A. et Dee, L. E. (2021). Biodiversity as insurance: from concept to measurement and application. *Biological Reviews*, *96*(5), 2333-2354. https://doi.org/10.1111/brv.12756

Lovrić, M., Da Re, R., Vidale, E., Prokofieva, I., Wong, J., Pettenella, D., Verkerk, P. J. et Mavsar, R. (2020). Non-wood forest products in Europe – A quantitative overview. *Forest Policy and Economics*, *116*, 102175. https://doi.org/10.1016/j.forpol.2020.102175

Lucash, M. S., Scheller, R. M., J. Gustafson, E. et R. Sturtevant, B. (2017). Spatial resilience of forested landscapes under climate change and management. *Landscape Ecology*, *32*(5), 953-969. https://doi.org/10.1007/s10980-017-0501-3

Lugo, A. E. et Gucinski, H. (2000). Function, effects, and management of forest roads. *Forest Ecology and Management*, *133*(3), 249-262. https://doi.org/10.1016/S0378-1127(99)00237-6

Luyssaert, S., Schulze, E.-D., Börner, A., Knohl, A., Hessenmöller, D., Law, B. E., Ciais, P. et Grace, J. (2008). Old-growth forests as global carbon sinks. *Nature*, *455*(7210), 213-215. https://doi.org/10.1038/nature07276

Lyra, A., Imbach, P., Rodriguez, D., Chou, S. C., Georgiou, S. et Garofolo, L. (2017). Projections of climate change impacts on central America tropical rainforest. *Climatic Change*, *141*(1), 93-105. https://doi.org/10.1007/s10584-016-1790-2

MacKinnon, A. (1998). Biodiversity and old-growth forests. *Conservation biology principles for forested landscapes. J. Voller and S. Harrison (editors). University of British Columbia Press, Vancouver, BC*, 146-184.

Malcolm, J. R., Holtsmark, B. et Piascik, P. W. (2020). Forest harvesting and the carbon debt in boreal east-central Canada. *Climatic Change*, *161*(3), 433-449. https://doi.org/10.1007/s10584-020-02711-8

Mansuy, N., Gauthier, S. et Bergeron, Y. (2013). Afforestation opportunities when stand productivity is driven by a high risk of natural disturbance: a review of the open lichen woodland in the eastern boreal forest of Canada. *Mitigation and Adaptation Strategies for Global Change*, *18*(2), 245-264. https://doi.org/10.1007/s11027-012-9362-x

Mansuy, N., Thiffault, E., Paré, D., Bernier, P., Guindon, L., Villemaire, P., Poirier, V. et Beaudoin, A. (2014). Digital mapping of soil properties in Canadian managed forests at 250m of resolution using the k-nearest neighbor method. *Geoderma*, *235-236*, 59-73. https://doi.org/10.1016/j.geoderma.2014.06.032

Maréchaux, I., Langerwisch, F., Huth, A., Bugmann, H., Morin, X., Reyer, C. P. O., Seidl, R., Collalti, A., Dantas de Paula, M., Fischer, R., Gutsch, M., Lexer, M. J., Lischke, H., Rammig, A., Rödig, E., Sakschewski, B., Taubert, F., Thonicke, K., Vacchiano, G. et Bohn, F. J. (2021). Tackling unresolved questions in forest ecology: The past and future role of simulation models. *Ecology and Evolution*, *11*(9), 3746-3770. https://doi.org/10.1002/ece3.7391

Marsh, D. M. et Beckman, N. G. (2004). Effects of Forest Roads on the Abundance and Activity of Terrestrial Salamanders. *Ecological Applications*, *14*(6), 1882-1891. https://doi.org/10.1890/03-5179

Marsh, D. M., Milam, G. S., Gorham, N. P. et Beckman, N. G. (2005). Forest Roads as Partial Barriers to Terrestrial Salamander Movement. *Conservation Biology*, *19*(6), 2004-2008. https://doi.org/10.1111/j.1523-1739.2005.00238.x

Martin, M., Boucher, Y., Fenton, N. J., Marchand, P. et Morin, H. (2020). Forest management has reduced the structural diversity of residual boreal old-growth forest landscapes in Eastern Canada. *Forest Ecology and Management*, *458*, 117765. https://doi.org/10.1016/j.foreco.2019.117765

Martin, M., Leduc, A., Fenton, N. J., Montoro Girona, M., Bergeron, Y. et 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/10.1111/1365-2664.14186

Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S. L., Péan, C., Berger, S., Caud, N., Chen, Y., Goldfarb, L., Gomis, M. et others. (2021). Climate change 2021: the physical science basis. *Contribution of working group I to the sixth assessment report of the intergovernmental panel on climate change*, *2*.

Matricardi, E. A. T., Skole, D. L., Costa, O. B., Pedlowski, M. A., Samek, J. H. et Miguel, E. P. (2020). Long-term forest degradation surpasses deforestation in the Brazilian Amazon. *Science*, *369*(6509), 1378-1382. https://doi.org/10.1126/science.abb3021

Matthews, J. D. (1991). *Silvicultural Systems*. Clarendon Press.

McGarigal, K., Cushman, S. A. et Ene, E. (2012). FRAGSTATS v4: spatial pattern analysis program for categorical and continuous maps. *Computer software program produced by the authors at the University of Massachusetts, Amherst. Available at the following web site: http://www. umass. edu/landeco/research/fragstats/fragstats. html*.

McMullin, R. T. et Wiersma, Y. F. (2019). Out with OLD growth, in with ecological continNEWity: new perspectives on forest conservation. *Frontiers in Ecology and the Environment*, *17*(3), 176-181. https://doi.org/10.1002/fee.2016

Merschel, A., Vora, R. S. et Spies, T. (2019). Conserving dry old-growth forest in Central Oregon, USA. *Journal of Forestry*, *117*(2), 128-135.

Messier, C. (2022). *Thirty Years of Triad Forestry: A critical clarification of theory and recommendations for implementation and testing*.

Messier, C., Bauhus, J., Doyon, F., Maure, F., Sousa-Silva, R., Nolet, P., Mina, M., Aquilué, N., Fortin, M.-J. et Puettmann, K. (2019). The functional complex network approach to foster forest resilience to global changes. *Forest Ecosystems*, *6*(1), 21. https://doi.org/10.1186/s40663-019-0166-2

Messier, C., Betts, M., Tittler, R. et Paquette, A. (2020). A novel TRIAD approach to increase resilience of the forest landscape to global change: or how to make a better omelette without cracking too many. Dans *How to balance forestry and biodiversity conservation – A view across Europe*.

Messier, C., Puettmann, K., Chazdon, R., Andersson, K. P., Angers, V. A., Brotons, L., Filotas, E., Tittler, R., Parrott, L. et Levin, S. A. (2015). From Management to Stewardship: Viewing Forests As Complex Adaptive Systems in an Uncertain World. *Conservation Letters*, *8*(5), 368-377. https://doi.org/10.1111/conl.12156

Messier, C., Puettmann, K. J. et Coates, K. D. (2013). *Managing Forests as Complex Adaptive Systems: Building Resilience to the Challenge of Global Change*. Routledge.

Messier, C., Tittler, R., Kneeshaw, D. D., Gélinas, N., Paquette, A., Berninger, K., Rheault, H., Meek, P. et Beaulieu, N. (2009). TRIAD zoning in Quebec: Experiences and results after 5 years. *The forestry chronicle*, *85*(6), 885-896.

Mestre, L. a. M., Cosset, C. C. P., Nienow, S. S., Krul, R., Rechetelo, J., Festti, L. et Edwards, D. P. (2020). Impacts of selective logging on avian phylogenetic and functional diversity in the Amazon. *Animal Conservation*, *23*(6), 725-740. https://doi.org/10.1111/acv.12592

Meunier, G. et Lavoie, C. (2012). Roads as Corridors for Invasive Plant Species: New Evidence from Smooth Bedstraw (Galium mollugo). *Invasive Plant Science and Management*, *5*(1), 92-101. https://doi.org/10.1614/IPSM-D-11-00049.1

MFFP. Le 5e inventaire écoforestier du Québec méridional, Présentation des orientations. Québec, Ministère des Forêts, de la Faune et des Parcs, secteur des forêts, Direction des inventaires forestiers. 2015.

MFFP. (2016). *Intégration des enjeux écologiques dans les plans d'aménagement forestier intégré de 2018-2023, Cahier 2.1 – Enjeu liés à la structure d'âge des forêts*. Gouvernement du Québec.

MFFP. (2018a). *Cartographie du 5e inventaire écoforestier du Québec méridional - Méthodes et données associées*. http://collections.banq.qc.ca/ark:/52327/3466439

MFFP. Plan d'aménagement forestier intégré tactique (PAFIT) 2018-2023 de l'Unité d'aménagement 042-51 de la Région de la Mauricie et du Centre-Du-Québec. https://mffp.gouv.qc.ca/wp-content/uploads/PAFIT_042-51_2018-2023.pdf 2018.

MFFP. Guide d'application du Règlement sur l'aménagement durable des forêts du domaine de l'État. https://mffp.gouv.qc.ca/RADF/guide/ 2020.

Mikoláš, M., Svitok, M., Bače, R., Meigs, G. W., Keeton, W. S., Keith, H., Buechling, A., Trotsiuk, V., Kozák, D., Bollmann, K., Begovič, K., Čada, V., Chaskovskyy, O., Ralhan, D., Dušátko, M., Ferenčík, M., Frankovič, M., Gloor, R., Hofmeister, J., … Svoboda, M. (2021). Natural disturbance impacts on trade-offs and co-benefits of forest biodiversity and carbon. *Proceedings of the Royal Society B: Biological Sciences*, *288*(1961), 20211631. https://doi.org/10.1098/rspb.2021.1631

Millar, C. I. et Stephenson, N. L. (2015). Temperate forest health in an era of emerging megadisturbance. *Science*, *349*(6250), 823-826. https://doi.org/10.1126/science.aaa9933

Mina, M., Bugmann, H., Cordonnier, T., Irauschek, F., Klopcic, M., Pardos, M. et Cailleret, M. (2017). Future ecosystem services from European mountain forests under climate change. *Journal of Applied Ecology*, *54*(2), 389-401. https://doi.org/10.1111/1365-2664.12772

Mina, M., Messier, C., Duveneck, M., Fortin, M. et Aquilué, N. (2021). Network analysis can guide resilience-based management in forest landscapes under global change. *Ecological Applications*, *31*(1). https://doi.org/10.1002/eap.2221

Mina, M., Messier, C., Duveneck, M., Fortin, M.-J. et Aquilué, N. (2020). Network analysis can guide resilience-based management in forest landscapes under global change. *Ecological Applications*, e2221.

Mina, M., Messier, C., Duveneck, M., Fortin, M.-J. et Aquilué, N. (2022). Managing for the unexpected: building resilient forest landscapes to cope with global change. *Global Change Biology*, *n/a*(n/a). https://doi.org/10.1111/gcb.16197

Mitchard, E. T. A. (2018). The tropical forest carbon cycle and climate change. *Nature*, *559*(7715), 527-534. https://doi.org/10.1038/s41586-018-0300-2

Molina, E., Valeria, O., De Grandpre, L., Ramirez, J. A., Cyr, D. et 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. https://doi.org/10.1016/j.foreco.2021.119016

Montejo-Kovacevich, G., Hethcoat, M. G., Lim, F. K. S., Marsh, C. J., Bonfantti, D., Peres, C. A. et Edwards, D. P. (2018). Impacts of selective logging management on butterflies in the Amazon. *Biological Conservation*, *225*, 1-9. https://doi.org/10.1016/j.biocon.2018.06.012

Morán-Ordóñez, A., Ameztegui, A., De Cáceres, M., de-Miguel, S., Lefèvre, F., Brotons, L. et Coll, L. (2020). Future trade-offs and synergies among ecosystem services in Mediterranean forests under global change scenarios. *Ecosystem Services*, *45*, 101174. https://doi.org/10.1016/j.ecoser.2020.101174

Mori, A. S., Furukawa, T. et Sasaki, T. (2013). Response diversity determines the resilience of ecosystems to environmental change. *Biological Reviews*, *88*(2), 349-364. https://doi.org/10.1111/brv.12004

Mori, A. S. et Kitagawa, R. (2014). Retention forestry as a major paradigm for safeguarding forest biodiversity in productive landscapes: A global meta-analysis. *Biological Conservation*, *175*, 65-73. https://doi.org/10.1016/j.biocon.2014.04.016

Moriarty, K. M., Epps, C. W., Betts, M. G., Hance, D. J., Bailey, J. D. et Zielinski, W. J. (2015). Experimental evidence that simplified forest structure interacts with snow cover to influence functional connectivity for Pacific martens. *Landscape Ecology*, *30*(10), 1865-1877. https://doi.org/10.1007/s10980-015-0216-2

Moris, J. V., Reilly, M. J., Yang, Z., Cohen, W. B., Motta, R. et Ascoli, D. (2022). Using a trait-based approach to asses fire resistance in forest landscapes of the Inland Northwest, USA. *Landscape Ecology*, *37*(8), 2149-2164. https://doi.org/10.1007/s10980-022-01478-w

Mortensen, D. A., Rauschert, E. S. J., Nord, A. N. et Jones, B. P. (2009). Forest Roads Facilitate the Spread of Invasive Plants. *Invasive Plant Science and Management*, *2*(3), 191-199. https://doi.org/10.1614/IPSM-08-125.1

Moss, R. H., Babiker, M., Brinkman, S., Calvo, E., Carter, T., Edmonds, J. A., Elgizouli, I., Emori, S., Lin, E., Hibbard, K. et others. (2008). *Towards new scenarios for analysis of emissions, climate change, impacts, and response strategies*. Pacific Northwest National Lab.(PNNL), Richland, WA (United States).

Mosseler, A., Thompson, I. et Pendrel, B. A. (2003). Overview of old-growth forests in Canada from a science perspective. *Environmental Reviews*, *11*(S1), S1-S7. https://doi.org/10.1139/a03-018

MRNF. (2023). *Chiffres clés du Québec forestier - 2023*. https://cdn-contenu.quebec.ca/cdn-contenu/forets/documents/entreprises/RA_chiffres-cles_forets_MRNF.pdf

Muir, J. (1979). *John of the mountains: The unpublished journals of John Muir*. Univ of Wisconsin Press.

Naderializadeh, N. et Crowe, K. A. (2020). Formulating the integrated forest harvest-scheduling model to reduce the cost of the road-networks. *Operational Research*, *20*(4), 2283-2306. https://doi.org/10.1007/s12351-018-0410-5

Nambiar, E. K. S. (2019). Tamm Review: Re-imagining forestry and wood business: pathways to rural development, poverty alleviation and climate change mitigation in the tropics. *Forest Ecology and Management*, *448*, 160-173. https://doi.org/10.1016/j.foreco.2019.06.014

Narayanaraj, G. et Wimberly, M. C. (2011). Influences of forest roads on the spatial pattern of wildfire boundaries. *International Journal of Wildland Fire*, *20*(6), 792-803. https://doi.org/10.1071/WF10032

Narayanaraj, G. et Wimberly, M. C. (2012). Influences of forest roads on the spatial patterns of human- and lightning-caused wildfire ignitions. *Applied Geography*, *32*(2), 878-888. https://doi.org/10.1016/j.apgeog.2011.09.004

NASA. (2023, 25 octobre). *Tracking Canada's Extreme 2023 Fire Season* [Text.Article]. NASA Earth Observatory. https://earthobservatory.nasa.gov/images/151985/tracking-canadas-extreme-2023-fire-season

Nature. (2022). We must get a grip on forest science — before it's too late. *Nature*, *608*(7923), 449-449. https://doi.org/10.1038/d41586-022-02182-0

Nealis, V. et Peter, B. (2008). *Risk assessment of the threat of mountain pine beetle to Canadaś boreal and eastern pin forests*.

Neary, D. G., Ice, G. G. et Jackson, C. R. (2009). Linkages between forest soils and water quality and quantity. *Forest Ecology and Management*, *258*(10), 2269-2281. https://doi.org/10.1016/j.foreco.2009.05.027

Nepstad, D., Carvalho, G., Cristina Barros, A., Alencar, A., Paulo Capobianco, J., Bishop, J., Moutinho, P., Lefebvre, P., Lopes Silva, U. et Prins, E. (2001). Road paving, fire regime feedbacks, and the future of Amazon forests. *Forest Ecology and Management*, *154*(3), 395-407. https://doi.org/10.1016/S0378-1127(01)00511-4

Nevečerel, H., Pentek, T., Pičman, D. et Stankić, I. (2007). Traffic load of forest roads as a criterion for their categorization – GIS analysis. *Croatian Journal of Forest Engineering : Journal for Theory and Application of Forestry Engineering*, *28*(1), 27-38.

Newnham, R. (1995). ROADPLAN: A tool for designing forest road networks. *Journal of Forest Engineering*, *6*(2), 17-26.

Nolet, P., Kneeshaw, D., Messier, C. et Béland, M. (2018). Comparing the effects of even- and uneven-aged silviculture on ecological diversity and processes: A review. *Ecology and Evolution*, *8*(2), 1217-1226. https://doi.org/10.1002/ece3.3737

Nousiainen, D. et Mola-Yudego, B. (2022). Characteristics and emerging patterns of forest conflicts in Europe - What can they tell us? *Forest Policy and Economics*, *136*, 102671. https://doi.org/10.1016/j.forpol.2021.102671

Nowak, D. J., Hirabayashi, S., Bodine, A. et Greenfield, E. (2014). Tree and forest effects on air quality and human health in the United States. *Environmental Pollution*, *193*, 119-129. https://doi.org/10.1016/j.envpol.2014.05.028

Nowak, D. J., Hirabayashi, S., Doyle, M., McGovern, M. et Pasher, J. (2018). Air pollution removal by urban forests in Canada and its effect on air quality and human health. *Urban Forestry & Urban Greening*, *29*, 40-48. https://doi.org/10.1016/j.ufug.2017.10.019

Nunes, L. J. R., Meireles, C. I. R., Pinto Gomes, C. J. et Almeida Ribeiro, N. M. C. (2020). Forest Contribution to Climate Change Mitigation: Management Oriented to Carbon Capture and Storage. *Climate*, *8*(2), 21. https://doi.org/10.3390/cli8020021

Nyland, R. D. (2016). *Silviculture: Concepts and Applications, Third Edition*. Waveland Press.

O'Hara, K. L. (2002). The historical development of uneven-aged silviculture in North America. *Forestry*, *75*(4), 339-346. https://doi.org/10.1093/forestry/75.4.339

Oliver, T. H., Heard, M. S., Isaac, N. J. B., Roy, D. B., Procter, D., Eigenbrod, F., Freckleton, R., Hector, A., Orme, C. D. L., Petchey, O. L., Proença, V., Raffaelli, D., Suttle, K. B., Mace, G. M., Martín-López, B., Woodcock, B. A. et Bullock, J. M. (2015). Biodiversity and Resilience of Ecosystem Functions. *Trends in Ecology & Evolution*, *30*(11), 673-684. https://doi.org/10.1016/j.tree.2015.08.009

Oreskes, N. et Conway, E. M. (2010). Defeating the merchants of doubt. *Nature*, *465*(7299), 686-687. https://doi.org/10.1038/465686a

Ortega, Y. K. et Capen, D. E. (1999). Effects of Forest Roads on Habitat Quality for Ovenbirds in a Forested Landscape. *The Auk*, *116*(4), 937‑946. https://doi.org/10.2307/4089673

Osborne, P., Aquilué, N., Mina, M., Moe, K., Jemtrud, M. et Messier, C. (2023). A trait-based approach to both forestry and timber building can synchronize forest harvest and resilience. *PNAS Nexus*, *2*(8), pgad254. https://doi.org/10.1093/pnasnexus/pgad254

Ostrom, E. (2009). A general framework for analyzing sustainability of social-ecological systems. *Science (New York, N.Y.)*, *325*(5939), 419‑422.

Oswalt, S. N. et Smith, W. B. (2014). *US forest resource facts and historical trends*. United States Deparment of Agriculture, Forest Service.

Paddison, L. (2023, 22 août). *Weather that drove eastern Canada's devastating wildfires made twice as likely by climate change*. CNN. https://www.cnn.com/2023/08/22/americas/canada-wildfires-climate-change-quebec/index.html

Paillet, Y., Bergès, L., Hjältén, J., Ódor, P., Avon, C., Bernhardt-Römermann, M., Bijlsma, R.-J., Bruyn, L. D., Fuhr, M., Grandin, U., Kanka, R., Lundin, L., Luque, S., Magura, T., Matesanz, S., Mészáros, I., Sebastià, M.-T., Schmidt, W., Standovár, T., … Virtanen, R. (2010). Biodiversity Differences between Managed and Unmanaged Forests: Meta-Analysis of Species Richness in Europe. *Conservation Biology*, *24*(1), 101‑112. https://doi.org/10.1111/j.1523-1739.2009.01399.x

Paluš, H., Parobek, J., Moravčík, M., Kovalčík, M., Dzian, M. et Murgaš, V. (2020). Projecting Climate Change Potential of Harvested Wood Products under Different Scenarios of Wood Production and Utilization: Study of Slovakia. *Sustainability*, *12*(6), 2510. https://doi.org/10.3390/su12062510

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. et Hayes, D. (2011). A Large and Persistent Carbon Sink in the World's Forests. *Science*, *333*(6045), 988‑993. https://doi.org/10.1126/science.1201609

Paquette, A., Bouchard, A. et Cogliastro, A. (2006). Survival and growth of under-planted trees: a meta-analysis across four biomes. *Ecological Applications*, *16*(4), 1575‑1589.

Paquette, A. et Messier, C. (2011). The effect of biodiversity on tree productivity: from temperate to boreal forests. *Global Ecology and Biogeography*, *20*(1), 170‑180. https://doi.org/10.1111/j.1466-8238.2010.00592.x

Park, A., Henschel, C., Kuttner, B. et McEachern, G. (2005). *A Cut Above - A Look at Alternatives to Clearcutting in the Boreal Forest*. CPAWS - Wildlands League.

Pellegrini, P. et Fernández, R. J. (2018). Crop intensification, land use, and on-farm energy-use efficiency during the worldwide spread of the green revolution. *Proceedings of the National Academy of Sciences of the United States of America*, *115*(10), 2335‑2340. https://doi.org/10.1073/pnas.1717072115

Pendrill, F., Persson, U. M., Godar, J. et Kastner, T. (2019). Deforestation displaced: trade in forest-risk commodities and the prospects for a global forest transition. *Environmental Research Letters*, *14*(5), 055003. https://doi.org/10.1088/1748-9326/ab0d41

Peng, L., Searchinger, T. D., Zionts, J. et Waite, R. (2023). The carbon costs of global wood harvests. *Nature*, *620*(7972), 110-115. https://doi.org/10.1038/s41586-023-06187-1

Perron, M. (2011). Le mélèze hybride du québec, performant et racé. *Avis de recherche forestière*, *37*.

Phalan, B. T. (2018). What Have We Learned from the Land Sparing-sharing Model? *Sustainability*, *10*(6), 1760. https://doi.org/10.3390/su10061760

Pilli, R., Vizzarri, M. et Chirici, G. (2021). Combined effects of natural disturbances and management on forest carbon sequestration: the case of Vaia storm in Italy. *Annals of Forest Science*, *78*(2), 46. https://doi.org/10.1007/s13595-021-01043-6

Plamondon Lalancette, P. et Movilla, M. (2021, 5 mars). Les chemins forestiers envahissent le Québec sauvage au détriment de l'environnement. *Radio Canada*. https://ici.radio-canada.ca/nouvelle/1775237/ministere-forets-routes-deboisement-deforestation

Pohjanmies, T., Eyvindson, K., Triviño, M., Bengtsson, J. et Mönkkönen, M. (2021). Forest multifunctionality is not resilient to intensive forestry. *European Journal of Forest Research*, *140*(3), 537-549. https://doi.org/10.1007/s10342-020-01348-7

Pommerening, A. et Murphy, S. T. (2004). A review of the history, definitions and methods of continuous cover forestry with special attention to afforestation and restocking. *Forestry: An International Journal of Forest Research*, *77*(1), 27-44. https://doi.org/10.1093/forestry/77.1.27

Poore, J. et Nemecek, T. (2018). Reducing food's environmental impacts through producers and consumers. *Science*, *360*(6392), 987-992. https://doi.org/10.1126/science.aaq0216

Popkin, G. (2015). The hunt for the world's missing carbon. *Nature*, *523*(7558), 20-22. https://doi.org/10.1038/523020a

Popkin, G. (2019). How much can forests fight climate change? *Nature*, *565*(7739), 280-282. https://doi.org/10.1038/d41586-019-00122-z

Poulain, C. Lignes directrices relatives à la gestion des refuges biologiques. Gouvernement du Québec, ministère des Forêts, de la Faune et des Parcs, Direction de l'aménagement et de l'environnement forestiers. https://mffp.gouv.qc.ca/documents/forets/amenagement/refuges-biologiques.pdf 2014.

Programme des Nations unies pour l'environnement et Organisation météorologique mondiale. (1988). Changing atmosphere: Implications for global security ; proceedings [of the] World Conference.

Proulx, S., Desjardins, N. et others. (2000). *Foresterie et papier: portrait de secteur: volet aménagement de la forêt*. Ministère de l'éducation,.

Puettmann, K., Coates, K. D. et Messier, C. (2009). *A critique of sylviculture: managing for complexity*. Island Press, Washington.

Puettmann, K. J., Wilson, S. M., Baker, S. C., Donoso, P. J., Drössler, L., Amente, G., Harvey, B. D., Knoke, T., Lu, Y., Nocentini, S., Putz, F. E., Yoshida, T. et Bauhus, J. (2015). Silvicultural alternatives to conventional even-aged forest management - what limits global adoption? *Forest Ecosystems*, *2*(1), 8. https://doi.org/10.1186/s40663-015-0031-x

Pukkala, T. (2016). Which type of forest management provides most ecosystem services? *Forest Ecosystems*, *3*(1), 9. https://doi.org/10.1186/s40663-016-0068-5

Pukkala, T., Lähde, E., Laiho, O., Salo, K. et Hotanen, J.-P. (2011). A multifunctional comparison of even-aged and uneven-aged forest management in a boreal region. *Canadian Journal of Forest Research*, *41*, 851-862. https://doi.org/10.1139/x11-009

Queiroz, E. A., Schoereder, J. H., Cornelissen, T. G., Brando, P. M., Maracahipes, L. et Paolucci, L. N. (2022). Reduced predation by arthropods and higher herbivory in burned Amazonian forests. *Biotropica*, *n/a*(n/a). https://doi.org/10.1111/btp.13129

Radio Canada. (2021, 4 mars). L'argent pousse dans les arbres. Dans Ici Télé Radio Canada, *Enquête*. https://ici.radio-canada.ca/tele/enquete/site/episodes/510542/clova-forets-arbre-industrie-ministere

Radio Canada. (2022, 17 mars). Le Far West du Québec. Dans Ici Télé Radio Canada, *Enquête*. https://ici.radio-canada.ca/tele/enquete/site/episodes/601353/route-chemin-foret--bois-tordeuse-epinette

Radkau, J. (2012). *Wood: A History*. Polity.

Reed, R. A., Johnson-Barnard, J. et Baker, W. L. (1996). Contribution of Roads to Forest Fragmentation in the Rocky Mountains. *Conservation Biology*, *10*(4), 1098-1106. https://doi.org/10.1046/j.1523-1739.1996.10041098.x

Referowska-Chodak, E. (2019). Pressures and Threats to Nature Related to Human Activities in European Urban and Suburban Forests. *Forests*, *10*(9), 765. https://doi.org/10.3390/f10090765

Regina, E. (2015). *Supercluster.KDTree C# Nuget Package*. https://github.com/eregina92/Supercluster.KDTree

Régnière, J., St-Amant, R. et Duval, P. (2012). Predicting insect distributions under climate change from physiological responses: spruce budworm as an example. *Biological Invasions*, *14*(8), 1571-1586. https://doi.org/10.1007/s10530-010-9918-1

Regolin, A. L., Oliveira-Santos, L. G., Ribeiro, M. C. et Bailey, L. L. (2021). Habitat quality, not habitat amount, drives mammalian habitat use in the Brazilian Pantanal. *Landscape Ecology*. https://doi.org/10.1007/s10980-021-01280-0

Reid, W. V., Mooney, H. A., Cropper, A., Capistrano, D., Carpenter, S. R., Chopra, K., Dasgupta, P., Dietz, T., Duraiappah, A. K., Hassan, R., Kasperson, R., Leemans, R., May, R. M., McMichael, A. J., Pingali, P., Samper, C., Scholes, R., Watson, R. T., Zakri, A. H., … Zurek, M. B. (2005). *Ecosystems and human well-being - Synthesis: A Report of the Millennium Ecosystem Assessment* (p.). Island Press. https://library.wur.nl/WebQuery/wurpubs/340442

Remsoft. (2019). *Remsoft Road Optimizer*. https://www.remsoft.com/road-optimizer/

Reuters. (2023). *German forest ranger aims to outsmart drought*. https://www.youtube.com/watch?v=PaveALkJiXk

Reyer, C., Guericke, M. et Ibisch, P. L. (2009). Climate change mitigation via afforestation, reforestation and deforestation avoidance: and what about adaptation to environmental change? *New Forests*, *38*(1), 15-34. https://doi.org/10.1007/s11056-008-9129-0

Reyer, C. P. O., Bathgate, S., Blennow, K., Borges, J. G., Bugmann, H., Delzon, S., Faias, S. P., Garcia-Gonzalo, J., Gardiner, B., Gonzalez-Olabarria, J. R., Gracia, C., Hernández, J. G., Kellomäki, S., Kramer, K., Lexer, M. J., Lindner, M., van der Maaten, E., Maroschek, M., Muys, B., … Hanewinkel, M. (2017). Are forest disturbances amplifying or canceling out climate change-induced productivity changes in European forests? *Environmental research letters: ERL [Web site]*, *12*(3), 034027. https://doi.org/10.1088/1748-9326/aa5ef1

Reyer, C. P. O., Brouwers, N., Rammig, A., Brook, B. W., Epila, J., Grant, R. F., Holmgren, M., Langerwisch, F., Leuzinger, S., Lucht, W., Medlyn, B., Pfeifer, M., Steinkamp, J., Vanderwel, M. C., Verbeeck, H. et Villela, D. M. (2015). Forest resilience and tipping points at different spatio-temporal scales: approaches and challenges. *Journal of Ecology*, *103*(1), 5-15. https://doi.org/10.1111/1365-2745.12337

Ribe, R. G. (2005). Aesthetic perceptions of green-tree retention harvests in vista views: The interaction of cut level, retention pattern and harvest shape. *Landscape and Urban Planning*, *73*(4), 277-293.

Ríos-Saldaña, C. A., Delibes-Mateos, M. et Ferreira, C. C. (2018). Are fieldwork studies being relegated to second place in conservation science? *Global Ecology and Conservation*, *14*, e00389. https://doi.org/10.1016/j.gecco.2018.e00389

Riva, F. et Fahrig, L. (2022). The disproportionately high value of small patches for biodiversity conservation. *Conservation Letters*. https://doi.org/10.1111/conl.12881

Roa Cea, I. B. (2011). *Effets des coupes partielles et à rétention variable sur la distance de débardage et les coûts de récolte, étude de cas*, 117.

Roberge, J.-M., Laudon, H., Björkman, C., Ranius, T., Sandström, C., Felton, A., Sténs, A., Nordin, A., Granström, A., Widemo, F., Bergh, J., Sonesson, J., Stenlid, J. et Lundmark, T. (2016). Socio-ecological implications of modifying rotation lengths in forestry. *Ambio*, *45*(Suppl 2), 109-123. https://doi.org/10.1007/s13280-015-0747-4

Robitaille, A. et Saucier, J. (1998). *Paysages régionaux du québec méridional* (Gouvernement du québec, ministère des ressources naturelles). Les publications du Québec.

Roces-Díaz, J. V., Vayreda, J., De Cáceres, M., García-Valdés, R., Banqué-Casanovas, M., Morán-Ordóñez, A., Brotons, L., de-Miguel, S. et Martínez-Vilalta, J. (2021). Temporal changes in Mediterranean forest ecosystem services are driven by stand development, rather than by climate-related disturbances. *Forest Ecology and Management*, *480*, 118623. https://doi.org/10.1016/j.foreco.2020.118623

Ross, S. R. P.-J., Petchey, O. L., Sasaki, T. et Armitage, D. W. (2023). How to measure response diversity. *Methods in Ecology and Evolution*, *14*(5), 1150-1167. https://doi.org/10.1111/2041-210X.14087

Ross, S. R. P.-J. et Sasaki, T. (2023). Limited theoretical and empirical evidence that response diversity determines the resilience of ecosystems to environmental change. *Ecological Research*, *n/a*(n/a). https://doi.org/10.1111/1440-1703.12434

Ruel, J.-C., Fortin, D. et Pothier, D. (2013). Partial cutting in old-growth boreal stands: An integrated experiment. *The Forestry Chronicle*, *89*(03), 360-369. https://doi.org/10.5558/tfc2013-066

Runting, R. K., Ruslandi, Griscom, B. W., Struebig, M. J., Satar, M., Meijaard, E., Burivalova, Z., Cheyne, S. M., Deere, N. J., Game, E. T., Putz, F. E., Wells, J. A., Wilting, A., Ancrenaz, M., Ellis, P., Khan, F. A. A., Leavitt, S. M., Marshall, A. J., Possingham, H. P., … Venter, O. (2019). Larger gains from improved management over sparing–sharing for tropical forests. *Nature Sustainability*, *2*(1), 53-61. https://doi.org/10.1038/s41893-018-0203-0

Russell, M. B., Patton, S. R., Wilson, D. C., Domke, G. M. et Frerker, K. L. (2018). Impacts of Alternative Harvesting and Natural Disturbance Scenarios on Forest Biomass in the Superior National Forest, USA. *Forests*, *9*(8), 491. https://doi.org/10.3390/f9080491

Ryan, T., Phillips, H., Ramsay, J., Dempsey, J. et others. (2004). *Forest Road Manual: Guidelines for the design, construction and management of forest roads*. COFORD Dublin.

Safranyik, L., Carroll, A. L., Régnière, J., Langor, D. W., Riel, W. G., Shore, T. L., Peter, B., Cooke, B. J., Nealis, V. G. et Taylor, S. W. (2010). Potential for range expansion of mountain pine beetle into the boreal forest of North America. *The Canadian Entomologist*, *142*(5), 415-442. https://doi.org/10.4039/n08-CPA01

Saint-Laurent, M.-H., Beauchesne, D. et Lesmerises, F. (2014). *Evaluation des impacts des vieux chemins forestiers et des modalités de fermeture dans un contexte de restauration de l'habitat du Caribou Forestier au Québec*. Université du Québec à Rimouski.

Sakschewski, B., von Bloh, W., Boit, A., Poorter, L., Peña-Claros, M., Heinke, J., Joshi, J. et Thonicke, K. (2016). Resilience of Amazon forests emerges from plant trait diversity. *Nature Climate Change*, *6*(11), 1032-1036. https://doi.org/10.1038/nclimate3109

Sasaki, T., Furukawa, T., Iwasaki, Y., Seto, M. et Mori, A. S. (2015). Perspectives for ecosystem management based on ecosystem resilience and ecological thresholds against multiple and stochastic disturbances. *Ecological Indicators*, *57*, 395-408. https://doi.org/10.1016/j.ecolind.2015.05.019

Schall, P., Gossner, M. M., Heinrichs, S. et Ammer, C. (2018). *In beech forest landscapes composed of different management systems biodiversity increases with the share of even-aged forests.* https://doi.org/10.17011/conference/eccb2018/107350

Scheller, R. M. et Domingo, J. B. (2018). *LANDIS-II Base Fire User Guide* (version 3.1). https://drive.google.com/file/d/0B6eUM6Se6MFBem0yZFhxWllfY00/view

Scheller, R. M., Domingo, J. B., Sturtevant, B. R., Williams, J. S., Rudy, A., Gustafson, E. J. et 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. https://doi.org/10.1016/j.ecolmodel.2006.10.009

Scheller, R. M. et Miranda, B. R. (2021). *Biomass Succession v6.0 Extension User Guide*. https://github.com/LANDIS-II-Foundation/Extension-Biomass-Succession/blob/651927e219fdf19f226d65a8953e42ff339e8907/docs/LANDIS-II%20Biomass%20Succession%20v6%20User%20Guide.pdf

Scheller, R. M. et 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/10.1016/j.ecolmodel.2004.01.022

Scheller, R. M. et Mladenoff, D. J. (2007). An ecological classification of forest landscape simulation models: tools and strategies for understanding broad-scale forested ecosystems. *Landscape Ecology*, *22*(4), 491-505. https://doi.org/10.1007/s10980-006-9048-4

Schirpke, U., Tscholl, S. et Tasser, E. (2020). Spatio-temporal changes in ecosystem service values: Effects of land-use changes from past to future (1860–2100). *Journal of Environmental Management*, *272*, 111068. https://doi.org/10.1016/j.jenvman.2020.111068

Schmitt, S., Maréchaux, I., Chave, J., Fischer, F. J., Piponiot, C., Traissac, S. et Hérault, B. (2020). Functional diversity improves tropical forest resilience: Insights from a long-term virtual experiment. *Journal of Ecology*, *108*(3), 831-843. https://doi.org/10.1111/1365-2745.13320

Schröter, M., van der Zanden, E. H., van Oudenhoven, A. P. E., Remme, R. P., Serna-Chavez, H. M., de Groot, R. S. et Opdam, P. (2014). Ecosystem Services as a Contested Concept: a Synthesis of Critique and Counter-Arguments. *Conservation Letters*, *7*(6), 514-523. https://doi.org/10.1111/conl.12091

Schulze, K., Malek, Ž. et Verburg, P. H. (2020). The Impact of Accounting for Future Wood Production in Global Vertebrate Biodiversity Assessments. *Environmental Management*, *66*(3), 460-475. https://doi.org/10.1007/s00267-020-01322-4

Schütz, J.-P., Pukkala, T., Donoso, P. J. et von Gadow, K. (2012). Historical Emergence and Current Application of CCF. Dans T. Pukkala et K. von Gadow (dir.), *Continuous Cover Forestry* (p. 1-28). Springer Netherlands. https://doi.org/10.1007/978-94-007-2202-6_1

Seidl, R., Albrich, K., Erb, K., Formayer, H., Leidinger, D., Leitinger, G., Tappeiner, U., Tasser, E. et Rammer, W. (2019). What drives the future supply of regulating ecosystem services in a mountain forest landscape? *Forest Ecology and Management*, *445*, 37-47. https://doi.org/10.1016/j.foreco.2019.03.047

Seidl, R., Honkaniemi, J., Aakala, T., Aleinikov, A., Angelstam, P., Bouchard, M., Boulanger, Y., Burton, P. J., De Grandpré, L., Gauthier, S., Hansen, W. D., Jepsen, J. U., Jõgiste, K., Kneeshaw, D. D., Kuuluvainen, T., Lisitsyna, O., Makoto, K., Mori, A. S., Pureswaran, D. S., … Senf, C. (2020). Globally consistent climate sensitivity of natural disturbances across boreal and temperate forest ecosystems. *Ecography*, *43*(7), 967-978. https://doi.org/10.1111/ecog.04995

Seidl, R., Rammer, W., Scheller, R. M. et Spies, T. A. (2012). An individual-based process model to simulate landscape-scale forest ecosystem dynamics. *Ecological Modelling*, *231*, 87-100. https://doi.org/10.1016/j.ecolmodel.2012.02.015

Seidl, R., Schelhaas, M.-J., Rammer, W. et Verkerk, P. J. (2014). Increasing forest disturbances in Europe and their impact on carbon storage. *Nature Climate Change*, *4*(9), 806-810. https://doi.org/10.1038/nclimate2318

Seidl, R., Spies, T. A., Peterson, D. L., Stephens, S. L. et Hicke, J. A. (2016). REVIEW: Searching for resilience: addressing the impacts of changing disturbance regimes on forest ecosystem services. *Journal of Applied Ecology*, *53*(1), 120-129. https://doi.org/10.1111/1365-2664.12511

Seidl, R., Thom, D., Kautz, M., Martin-Benito, D., Peltoniemi, M., Vacchiano, G., Wild, J., Ascoli, D., Petr, M., Honkaniemi, J., Lexer, M. J., Trotsiuk, V., Mairota, P., Svoboda, M., Fabrika, M., Nagel, T. A. et Reyer, C. P. O. (2017). Forest disturbances under climate change. *Nature Climate Change*, *7*(6), 395-402. https://doi.org/10.1038/nclimate3303

Sessions, J., Heinrich, R. et Castaneda-Langlois, H. (2016). Forest Road Engineering. Dans L. Pancel et M. Köhl (dir.), *Tropical Forestry Handbook* (p. 2301-2362). Springer. https://doi.org/10.1007/978-3-642-54601-3_162

Seymour, R. S. et Hunter, M. L., Jr. New Forestry in Eastern Spruce-Fir Forests: Principles and Applications to Maine. Maine Agricultural and Forest Expriment Station. https://digitalcommons.library.umaine.edu/cgi/viewcontent.cgi?referer=https://www.google.com/&httpsredir=1&article=1027&context=aes_miscpubs 1992.

Sharma, A., Bohn, K. K., Jose, S. et Dwivedi, P. (2016). Even-Aged vs. Uneven-Aged Silviculture: Implications for Multifunctional Management of Southern Pine Ecosystems. *Forests*, *7*(4), 86. https://doi.org/10.3390/f7040086

Sheppard, J. P., Chamberlain, J., Agúndez, D., Bhattacharya, P., Chirwa, P. W., Gontcharov, A., Sagona, W. C. J., Shen, H., Tadesse, W. et Mutke, S. (2020). Sustainable Forest Management Beyond the Timber-Oriented Status Quo: Transitioning to Co-production of Timber and Non-wood Forest Products—a Global Perspective. *Current Forestry Reports*, *6*(1), 26-40. https://doi.org/10.1007/s40725-019-00107-1

Shinneman, D. J., Cornett, M. W. et Palik, B. J. (2010). Simulating restoration strategies for a southern boreal forest landscape with complex land ownership patterns. *Forest Ecology and Management*, *259*(3), 446-458. https://doi.org/10.1016/j.foreco.2009.10.042

Shirasawa, H. et Hasegawa, H. (2014). A comparative study of heuristic algorithms for the multiple target access problem. *Journal of Forest Research*, *19*(5), 437-449. https://doi.org/10.1007/s10310-013-0432-z

Shorohova, E., Kneeshaw, D., Kuuluvainen, T. et Gauthier, S. (2011). Variability and dynamics of old-growth forests in the circumboreal zone: implications for conservation, restoration and management. *Silva Fennica*, *45*(5). https://doi.org/10.14214/sf.72

Silvério, D. V., Brando, P. M., Bustamante, M. M. C., Putz, F. E., Marra, D. M., Levick, S. R. et Trumbore, S. E. (2019). Fire, fragmentation, and windstorms: A recipe for tropical forest degradation. *Journal of Ecology*, *107*(2), 656-667. https://doi.org/10.1111/1365-2745.13076

Simard, D. G., Fyles, J. W., Paré, D. et Nguyen, T. (2001). Impacts of clearcut harvesting and wildfire on soil nutrient status in the Quebec boreal forest. *Canadian Journal of Soil Science*, *81*(2), 229-237. https://doi.org/10.4141/S00-028

Song, Z., Seitz, S., Li, J., Goebes, P., Schmidt, K., Kühn, P., Shi, X. et Scholten, T. (2019). Tree diversity reduced soil erosion by affecting tree canopy and biological soil crust development in a subtropical forest experiment. *Forest Ecology and Management*, *444*, 69-77. https://doi.org/10.1016/j.foreco.2019.04.015

Spasojevic, M. J., Bahlai, C. A., Bradley, B. A., Butterfield, B. J., Tuanmu, M.-N., Sistla, S., Wiederholt, R. et Suding, K. N. (2016). Scaling up the diversity–resilience relationship with trait databases and remote sensing data: the recovery of productivity after wildfire. *Global Change Biology*, *22*(4), 1421-1432. https://doi.org/10.1111/gcb.13174

Statistics Canada. (2018). *Human Activity and the Environment 2017: Forests in Canada*. https://www150.statcan.gc.ca/n1/pub/16-201-x/16-201-x2018001-eng.htm

Steffen, W., Broadgate, W., Deutsch, L., Gaffney, O. et Ludwig, C. (2015). The trajectory of the Anthropocene: The Great Acceleration. *The Anthropocene Review*, *2*(1), 81-98. https://doi.org/10.1177/2053019614564785

Steffen, W., Sanderson, A., Tyson, P., Jäger, J., Matson, P., Moore, B., Oldfield, F., Richardson, K., Schellnhuber, H. J., Turner, B. L. et Wasson, R. J. (2005). *Global Change and the Earth System: A Planet Under Pressure*. Springer. https://doi.org/10.1007/b137870

Steffen, W., Sanderson, R. A., Tyson, P. D., Jäger, J., Matson, P. A., Moore III, B., Oldfield, F., Richardson, K., Schellnhuber, H.-J., Turner, B. L. et others. (2006). *Global change and the earth system: a planet under pressure*. Springer Science & Business Media.

Stevens-Rumann, C. S., Kemp, K. B., Higuera, P. E., Harvey, B. J., Rother, M. T., Donato, D. C., Morgan, P. et Veblen, T. T. (2018). Evidence for declining forest resilience to wildfires under climate change. *Ecology Letters*, *21*(2). https://doi.org/10.1111/ele.12889

Stier-Jarmer, M., Throner, V., Kirschneck, M., Immich, G., Frisch, D. et Schuh, A. (2021). The Psychological and Physical Effects of Forests on Human Health: A Systematic Review of Systematic Reviews and Meta-Analyses. *International Journal of Environmental Research and Public Health*, *18*(4), 1770. https://doi.org/10.3390/ijerph18041770

Stone, I., Ouellet, J.-P., Sirois, L., Arseneau, M.-J. et St-Laurent, M.-H. (2008). Impacts of silvicultural treatments on arboreal lichen biomass in balsam fir stands on Québec's Gaspé Peninsula:

Implications for a relict caribou herd. *Forest Ecology and Management*, *255*(7), 2733-2742. https://doi.org/10.1016/j.foreco.2008.01.040

Storch, F., Boch, S., Gossner, M. M., Feldhaar, H., Ammer, C., Schall, P., Polle, A., Kroiher, F., Müller, J. et Bauhus, J. (2023). Linking structure and species richness to support forest biodiversity monitoring at large scales. *Annals of Forest Science*, *80*(1), 3. https://doi.org/10.1186/s13595-022-01169-1

St-Pierre, F., Drapeau, P. et St-Laurent, M.-H. (2021). Drivers of vegetation regrowth on logging roads in the boreal forest: Implications for restoration of woodland caribou habitat. *Forest Ecology and Management*, *482*, 118846. https://doi.org/10.1016/j.foreco.2020.118846

Strano, E., Nicosia, V., Latora, V., Porta, S. et Barthélemy, M. (2012). Elementary processes governing the evolution of road networks. *Scientific reports*, *2*(1), 296.

Stritih, A., Senf, C., Seidl, R., Grêt-Regamey, A. et Bebi, P. (2021). The impact of land-use legacies and recent management on natural disturbance susceptibility in mountain forests. *Forest Ecology and Management*, *484*, 118950. https://doi.org/10.1016/j.foreco.2021.118950

Strittholt, J. R., Dellasala, D. A. et Jiang, H. (2006). Status of mature and old-growth forests in the Pacific Northwest. *Conservation Biology*, *20*(2), 363-374.

Strukelj, M., Brais, S. et Paré, D. (2015). Nine-year changes in carbon dynamics following different intensities of harvesting in boreal aspen stands. *European Journal of Forest Research*, *134*(5), 737-754. https://doi.org/10.1007/s10342-015-0880-4

Stückelberger, J. A., Heinimann, H. R. et Burlet, E. C. (2006). Modeling spatial variability in the life-cycle costs of low-volume forest roads. *European Journal of Forest Research*, *125*(4), 377-390. https://doi.org/10.1007/s10342-006-0123-9

Stückelberger, J., Heinimann, H. R. et Chung, W. (2007). Improved road network design models with the consideration of various link patterns and road design elements. *Canadian journal of forest research*, *37*(11), 2281-2298.

Sturrock, R. N., Frankel, S. J., Brown, A. V., Hennon, P. E., Kliejunas, J. T., Lewis, K. J., Worrall, J. J. et Woods, A. J. (2011). Climate change and forest diseases. *Plant Pathology*, *60*(1), 133-149. https://doi.org/10.1111/j.1365-3059.2010.02406.x

Sturtevant, B. R., Gustafson, E. J., Li, W. et 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/10.1016/j.ecolmodel.2004.01.021

Sturtevant, B. R., Miranda, B. R., Shinneman, D. J., Gustafson, E. J. et Wolter, P. T. (2012). Comparing modern and presettlement forest dynamics of a subboreal wilderness: Does spruce budworm enhance fire risk? *Ecological Applications*, *22*(4), 1278-1296. https://doi.org/10.1890/11-0590.1

Swanson, M. E., Franklin, J. F., Beschta, R. L., Crisafulli, C. M., DellaSala, D. A., Hutto, R. L., Lindenmayer, D. B. et Swanson, F. J. (2011). The forgotten stage of forest succession: early-successional ecosystems on forest sites. *Frontiers in Ecology and the Environment*, *9*(2), 117-125. https://doi.org/10.1890/090157

Sylvain, J.-D., Roy, M.-E. et Tousignant, D. (2022). *Projet SIIGSOL : une cartographie numérique des sols pour soutenir l'aménagement forestier*. Direction de la recherche forestière du Ministère des Ressources Naturelles et des Forêts du Québec. https://mffp.gouv.qc.ca/nos-publications/projet-siigsol-cartographie-numerique-amenagement-forestier/?retour=https%3A%2F%2Fmffp.gouv.qc.ca%2Fauteurs-ministeriels%2Fjean-daniel-sylvain

Tahvonen, O. (2009). Optimal Choice Between Even- and Uneven-Aged Forestry. *Natural Resource Modeling*, *22*(2), 289-321. https://doi.org/10.1111/j.1939-7445.2008.00037.x

Taylor, P. et Fahrig, L. (2006). Landscape connectivity: back to the basics. *Connectivity conservation. Cambridge University Press, Cambridge*.

Terry, E. L., McLellan, B. N. et Watts, G. S. (2000). Winter habitat ecology of mountain caribou in relation to forest management. *Journal of Applied Ecology*, *37*(4), 589-602. https://doi.org/10.1046/j.1365-2664.2000.00523.x

Thom, D. et Seidl, R. (2016). Natural disturbance impacts on ecosystem services and biodiversity in temperate and boreal forests. *Biological Reviews*, *91*(3), 760-781. https://doi.org/10.1111/brv.12193

Thompson, I. D. (1994). Marten Populations in Uncut and Logged Boreal Forests in Ontario. *The Journal of Wildlife Management*, *58*(2), 272-280. https://doi.org/10.2307/3809391

Thompson, M. P., Gannon, B. M. et Caggiano, M. D. (2021). Forest Roads and Operational Wildfire Response Planning. *Forests*, *12*(2), 110. https://doi.org/10.3390/f12020110

Timoney, K. P. et Peterson, G. (1996). Failure of natural regeneration after clearcut logging in Wood Buffalo National Park, Canada. *Forest Ecology and Management*, *87*(1), 89-105. https://doi.org/10.1016/S0378-1127(96)03831-5

Timpane-Padgham, B. L., Beechie, T. et Klinger, T. (2017). A systematic review of ecological attributes that confer resilience to climate change in environmental restoration. *PLOS ONE*, *12*(3), e0173812. https://doi.org/10.1371/journal.pone.0173812

Tittler, R., Filotas, É., Kroese, J. et Messier, C. (2015). Maximizing Conservation and Production with Intensive Forest Management: It's All About Location. *Environmental Management*, *56*(5), 1104-1117. https://doi.org/10.1007/s00267-015-0556-3

Tittler, R., Hannon, S. J. et Norton, M. R. (2001). Residual tree retention ameliorates short-term effects of clear cutting on some boreal songbirds. *Ecological Applications*, *11*(6), 1656-1666.

Tittler, R., Messier, C. et Fall, A. (2012). Concentrating anthropogenic disturbance to balance ecological and economic values: applications to forest management. *Ecological Applications*, *22*(4), 1268-1277.

Tittler, R., Messier, C. et Goodman, R. C. (2016). Triad forest management: local fix or global solution. Dans *Ecological forest management handbook* (vol. 33, p. 33-45). CRC Press Boca Raton.

Toledo, M., Poorter, L., Peña-Claros, M., Alarcón, A., Balcázar, J., Leaño, C., Licona, J. C., Llanque, O., Vroomans, V., Zuidema, P. et Bongers, F. (2011). Climate is a stronger driver of tree and forest growth rates than soil and disturbance. *Journal of Ecology*, *99*(1), 254-264. https://doi.org/10.1111/j.1365-2745.2010.01741.x

Torras, O. et Saura, S. (2008). Effects of silvicultural treatments on forest biodiversity indicators in the Mediterranean. *Forest Ecology and Management*, *255*(8), 3322-3330. https://doi.org/10.1016/j.foreco.2008.02.013

Toscani, P., Sekot, W. et Holzleitner, F. (2020). Forest roads from the perspective of managerial Accounting—Empirical evidence from austria. *Forests*, *11*(4), 378.

Tremblay, J. A., Boulanger, Y., Cyr, D., Taylor, A. R., Price, D. T. et 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. https://doi.org/10.1371/journal.pone.0191645

Trigger, D. et Mulcock, J. (2005). Forests as Spiritually Significant Places: Nature, Culture and `Belonging' in Australia. *The Australian Journal of Anthropology*, *16*(3), 306-320. https://doi.org/10.1111/j.1835-9310.2005.tb00313.x

Trombulak, S. C. et Frissell, C. A. (2000). Review of Ecological Effects of Roads on Terrestrial and Aquatic Communities. *Conservation Biology*, *14*(1), 18-30. https://doi.org/10.1046/j.1523-1739.2000.99084.x

Trumbore, S., Brando, P. et Hartmann, H. (2015). Forest health and global change. *Science*, *349*(6250), 814-818.

Tscharntke, T., Clough, Y., Wanger, T. C., Jackson, L., Motzke, I., Perfecto, I., Vandermeer, J. et Whitbread, A. (2012). Global food security, biodiversity conservation and the future of agricultural intensification. *Biological Conservation*, *151*(1), 53-59. https://doi.org/10.1016/j.biocon.2012.01.068

Uhl, C., Veríssimo, A., Mattos, M. M., Brandino, Z. et Vieira, I. C. G. (1991). Social, economic, and ecological consequences of selective logging in an Amazon frontier: the case of Tailândia. *Forest ecology and management*, *46*(3-4), 243-273.

United States Department of Agriculture. (1997). *Draft Environmental Impact Statement - Kisatchi National Forest*. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5391435.pdf

van der Ree, R., Jaeger, J. A. G., van der Grift, E. A. et Clevenger, A. P. (2011). Effects of Roads and Traffic on Wildlife Populations and Landscape Function: Road Ecology is Moving toward Larger Scales. *Ecology and Society*, *16*(1). https://doi.org/10.5751/ES-03982-160148

Van Wagner, C. E. (1978). Age-class distribution and the forest fire cycle. *Canadian Journal of Forest Research*, *8*(2), 220-227. https://doi.org/10.1139/x78-034

Vanlandeghem, V., Drapeau, P., Prima, M., St-Laurent, M. et Fortin, D. (2021). Management-mediated predation rate in the caribou–moose–wolf system: spatial configuration of logging activities matters. *Ecosphere*, *12*(6). https://doi.org/10.1002/ecs2.3550

Venier, L. A., Thompson, I. D., Fleming, R., Malcolm, J., Aubin, I., Trofymow, J. A., Langor, D., Sturrock, R., Patry, C., Outerbridge, R. O., Holmes, S. B., Haeussler, S., De Grandpré, L., Chen, H. Y. H., Bayne, E., Arsenault, A. et Brandt, J. P. (2014). Effects of natural resource development on the terrestrial biodiversity of Canadian boreal forests. *Environmental Reviews*, *22*(4), 457-490. https://doi.org/10.1139/er-2013-0075

Verkerk, P. J., Costanza, R., Hetemäki, L., Kubiszewski, I., Leskinen, P., Nabuurs, G. J., Potočnik, J. et Palahí, M. (2020). Climate-Smart Forestry: the missing link. *Forest Policy and Economics*, *115*, 102164. https://doi.org/10.1016/j.forpol.2020.102164

Verkerk, P. J., Mavsar, R., Giergiczny, M., Lindner, M., Edwards, D. et Schelhaas, M. J. (2014). Assessing impacts of intensified biomass production and biodiversity protection on ecosystem services provided by European forests. *Ecosystem Services*, *9*, 155-165. https://doi.org/10.1016/j.ecoser.2014.06.004

Vicente-Serrano, S. M., Quiring, S. M., Peña-Gallardo, M., Yuan, S. et Domínguez-Castro, F. (2020). A review of environmental droughts: Increased risk under global warming? *Earth-Science Reviews*, *201*, 102953. https://doi.org/10.1016/j.earscirev.2019.102953

Vilà-Cabrera, A., Coll, L., Martínez-Vilalta, J. et Retana, J. (2018). Forest management for adaptation to climate change in the Mediterranean basin: A synthesis of evidence. *Forest Ecology and Management*, *407*, 16-22. https://doi.org/10.1016/j.foreco.2017.10.021

Violle, C., Navas, M.-L., Vile, D., Kazakou, E., Fortunel, C., Hummel, I. et Garnier, E. (2007). Let the concept of trait be functional! *Oikos*, *116*(5), 882-892. https://doi.org/10.1111/j.0030-1299.2007.15559.x

Vitousek, P. M. (1994). Beyond Global Warming: Ecology and Global Change. *Ecology*, *75*(7), 1861-1876. https://doi.org/10.2307/1941591

Walker, B., Crépin, A.-S., Nyström, M., Anderies, J. M., Andersson, E., Elmqvist, T., Queiroz, C., Barrett, S., Bennett, E., Cardenas, J. C., Carpenter, S. R., Chapin, F. S., de Zeeuw, A., Fischer, J., Folke, C., Levin, S., Nyborg, K., Polasky, S., Segerson, K., … Vincent, J. R. (2023). Response diversity as a sustainability strategy. *Nature Sustainability*, *6*(6), 621-629. https://doi.org/10.1038/s41893-022-01048-7

Walker, R. B., Coop, J. D., Downing, W. M., Krawchuk, M. A., Malone, S. L. et Meigs, G. W. (2019). How Much Forest Persists Through Fire? High-Resolution Mapping of Tree Cover to Characterize the Abundance and Spatial Pattern of Fire Refugia Across Mosaics of Burn Severity. *Forests*, *10*(9), 782. https://doi.org/10.3390/f10090782

Wallace, K. J. (2007). Classification of ecosystem services: Problems and solutions. *Biological Conservation*, *139*(3), 235-246. https://doi.org/10.1016/j.biocon.2007.07.015

Wang, J. A., Baccini, A., Farina, M., Randerson, J. T. et Friedl, M. A. (2021). Disturbance suppresses the aboveground carbon sink in North American boreal forests. *Nature Climate Change*, *11*(5), 435-441. https://doi.org/10.1038/s41558-021-01027-4

Wang, X., Blanchet, F. G. et Koper, N. (2014). Measuring habitat fragmentation: An evaluation of landscape pattern metrics. *Methods in Ecology and Evolution*, *5*(7), 634-646. https://doi.org/10.1111/2041-210X.12198

White, J. W., Rassweiler, A., Samhouri, J. F., Stier, A. C. et White, C. (2014). Ecologists should not use statistical significance tests to interpret simulation model results. *Oikos*, *123*(4), 385-388. https://doi.org/10.1111/j.1600-0706.2013.01073.x

Whittington, J., Hebblewhite, M., DeCesare, N. J., Neufeld, L., Bradley, M., Wilmshurst, J. et Musiani, M. (2011). Caribou encounters with wolves increase near roads and trails: a time-to-event approach: Wolf-caribou encounter rates. *Journal of Applied Ecology*, *48*(6), 1535-1542. https://doi.org/10.1111/j.1365-2664.2011.02043.x

Wirth, C., Gleixner, G. et Heimann, M. (2009). Old-Growth Forests: Function, Fate and Value – an Overview. Dans C. Wirth, G. Gleixner et M. Heimann (dir.), *Old-Growth Forests: Function, Fate and Value* (p. 3-10). Springer. https://doi.org/10.1007/978-3-540-92706-8_1

Witmer, G. W. et deCalesta, D. S. (1985). Effect of forest roads on habitat use by Roosevelt elk. *Northwest Science*, *59*.

Xu, W., He, H. S., Huang, C., Duan, S., Hawbaker, T. J., Henne, P. D., Liang, Y. et Zhu, Z. (2022). Large fires or small fires, will they differ in affecting shifts in species composition and distributions under climate change? *Forest Ecology and Management*, *510*, 120131. https://doi.org/10.1016/j.foreco.2022.120131

Yang, J., Cao, M. et Swenson, N. G. (2018). Why Functional Traits Do Not Predict Tree Demographic Rates. *Trends in Ecology & Evolution*, *33*(5), 326-336. https://doi.org/10.1016/j.tree.2018.03.003

Yelle, V. (2013). *Social perception of ecosystem management in Québec's black spruce forest : can large harvests emulating fire be acceptable to forest users, stakeholders and the uninformed public ?* http://hdl.handle.net/20.500.11794/24231

Yocom, L. L., Jenness, J., Fulé, P. Z. et Thode, A. E. (2019). Previous fires and roads limit wildfire growth in Arizona and New Mexico, U.S.A. *Forest Ecology and Management*, *449*, 117440. https://doi.org/10.1016/j.foreco.2019.06.037

Zemke, J. J. (2016). Runoff and Soil Erosion Assessment on Forest Roads Using a Small Scale Rainfall Simulator. *Hydrology*, *3*(3), 25. https://doi.org/10.3390/hydrology3030025

Zhang, F., Dong, Y., Xu, S., Yang, X. et Lin, H. (2020). An approach for improving firefighting ability of forest road network. *Scandinavian Journal of Forest Research*, *35*(8), 547-561. https://doi.org/10.1080/02827581.2020.1829029

Zhao, C., Brissette, F., Chen, J. et Martel, J.-L. (2020). Frequency change of future extreme summer meteorological and hydrological droughts over North America. *Journal of Hydrology*, *584*, 124316. https://doi.org/10.1016/j.jhydrol.2019.124316

Zhao, J., Daigneault, A. et Weiskittel, A. (2022). Estimating regional timber supply and forest carbon sequestration under shared socioeconomic pathways: A case study of Maine, USA. *PLOS Climate*, *1*(5), e0000018. https://doi.org/10.1371/journal.pclm.0000018

Zheng, B., Ciais, P., Chevallier, F., Chuvieco, E., Chen, Y. et Yang, H. (2021). Increasing forest fire emissions despite the decline in global burned area. *Science Advances*, *7*(39), eabh2646. https://doi.org/10.1126/sciadv.abh2646

Zhou, T., Luo, X., Hou, Y., Xiang, Y. et Peng, S. (2020). Quantifying the effects of road width on roadside vegetation and soil conditions in forests. *Landscape Ecology*, *35*(1), 69-81. https://doi.org/10.1007/s10980-019-00930-8

Zuazo, V. H. D. et Pleguezuelo, C. R. R. (2009). Soil-Erosion and Runoff Prevention by Plant Covers: A Review. Dans E. Lichtfouse, M. Navarrete, P. Debaeke, S. Véronique et C. Alberola (dir.), *Sustainable Agriculture* (p. 785-811). Springer Netherlands. https://doi.org/10.1007/978-90-481-2666-8_48