Université du Québec en Abitibi-Témiscamingue

DÉTERMINATION DES EFFETS DES PRATIQUES SYLVICOLES SUR LES STOCKS ET LES FLUX DE CARBONE EN FORÊT BORÉALE TOURBEUSE

Thèse
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AVANT-PROPOS

Ce document transcrit le travail effectué au cours de ma thèse de doctorat initié en janvier 2018 au sein de l'équipe de recherche de l'IRF. Il s'organise autour de trois parties : une introduction générale pour situer le cadre de l'ensemble du travail, les chapitres traitant des grands axes de mon projet et une conclusion générale. Les chapitres, au nombre de trois, sont écrits en anglais et présentés sous forme d'article.

Le chapitre 1 intitulé " Natural dynamics of carbon storage after forest fire in a boreal forested peatland" décrit la dynamique naturelle de séquestration de carbone en forêt paludifiée. Il met en évidence les impacts du feu qui est le principal perturbateur naturel de la forêt boréale au travers de la modélisation. Les effets sont à la fois étudiés à l'échelle du peuplement, mais aussi à l'échelle du paysage. Les grands compartiments du système de forêts paludifiées, arbres et mousses, sont pris en compte dans la modélisation. Ce chapitre permet de définir une base de référence pour la comparaison des pratiques sylvicoles. Cet article a été soumis en novembre 2024 à la revue scientifique " Canadian Journal of Forest Research " avec la collaboration de mes directeurs et des co-auteurs. Les co-auteurs, Nicole Fenton et Kelly Bona, ont contribué à différents niveaux du travail en tant que personnes-ressources dans la fourniture de données ou de logiciel, l'analyse formelle, la révision et l'édition de l'article.

Le chapitre 2 portant sur "Effects of harvesting and mechanical site preparation on soil organic matter decay and carbon stocks in a boreal forested peatland" est basé sur des études de terrain. Il a pour but de fournir des données sur la magnitude de variation du taux de décomposition et des stocks de carbone de sol à la suite des pratiques sylvicoles pour mieux modéliser leurs impacts dans le chapitre 3 suivant. Cet article a été publié en avril 2023 dans la revue scientifique "Forest Ecology and Management" avec la collaboration de mes directeurs et des co-auteurs. Les co-auteurs qui ont contribué à ce travail sont : Nicole Fenton par la fourniture de données, la révision et l'édition de l'article; Osvaldo Valeria par l'analyse formelle géomatique, la révision et l'édition de l'article; et Philippe Marchand dans l'analyse formelle statistique, la visualisation des résultats, la révision et l'édition de l'article.

Quant à ce dernier chapitre 3 intitulé "Modeling the impacts of silvicultural practices and management on C fluxes and stocks in a boreal forested peatland", il traite des impacts de plusieurs pratiques sylvicoles telles que la coupe, les préparations mécaniques suivies des plantations et des brûlages dirigés suivis des plantations à l’échelle du peuplement. Les stratégies d’aménagement qui en résultent à l’échelle du paysage sont également évaluées en termes de stockage de carbone. L’article n’a pas encore été soumis à une revue scientifique.

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AIC : Akaike Information Criterion
ANOVA : Analyse de variance
BSFM : Black spruce-feathermoss
BSSP : Black spruce- Sphagnum
C : Carbone
CH_4 : Méthane
CO_2 : Dioxyde de carbone
COD : Carbone Organique Dissout
CB : Coupe à blanc
CP : Coupe partielle
CPRS : Coupe avec protection de la régénération et du sol
CT : Coupe totale
DOM : Dead Organic Matter
FH : Horizon organique fragmenté et humifié
HSF : High Severity Fire
IQS : Indice de qualité de station
LSF : Low Severity Fire
MBC- SFC3 : Modèle de Bilan C du Secteur Forestier Canadien
MOM : Matière Organique Morte
MSP : Mechanical site preparation
MVOL : Maximum merchantable volume
NPP : Net Primary Productivity
OM : Organic matter
PB : Prescribed burning
PC : Partial cut
PCA : Principal Component Analysis
PMS : Préparation mécanique du sol
SOC : Soil organic carbon

LISTE DES SYMBOLES ET DES UNITÉS

% : pourcentage

°C : degré Celsius

°N : degré de latitude Nord

°W : degré de longitude Ouest

ha : hectare

m : mètre

cm : centimètre

g : gramme

an ou year : année

Mg C ha^{-1} ou C t ha^{-1} : mégagramme équivalent à une tonne de carbone par hectare

Mg C ha^{-1} an^{-1} ou C t ha^{-1} an^{-1} : tonne de carbone par hectare par an

Mt C : mégatonne de carbone équivalent à un million de tonne de C

g C m^{-2} $year^{-1}$: gramme de carbone par mètre carré et par an

Q10 : coefficient de température, facteur par lequel le taux d'un processus biologique change lorsque la température augmente de 10 degrés Celsius.

RÉSUMÉ

Les forêts boréales tourbeuses ou paludifiées avec leur épaisse couche de matière organique au sol (*i.e.*, tourbe) constituent l'un des réservoirs terrestres de carbone (C) le plus important. La tourbe représente environ 60-80% de cet important stock de C. Cependant, cette épaisse couche du sol organique provenant principalement de la décomposition des mousses sphaignes, entraine une baisse de croissance des arbres et menace la production de bois à long terme dans certaines régions du nord-ouest du Québec. Afin de maintenir la productivité forestière, des perturbations naturelles comme les feux sévères ou les opérations forestières intensives qui réduisent la tourbe sont nécessaires. Si certaines opérations favorisent la croissance et la séquestration du C dans les arbres, elles peuvent, en contrepartie, entrainer une réduction des stocks du C du sol organique en accélérant les pertes par décomposition. Une meilleure connaissance des effets de ces pratiques sur les flux de C constitue donc un enjeu dans la maîtrise de l'empreinte C de l'industrie forestière.

Notre objectif principal était donc d'évaluer les effets de ces pratiques sur le bilan C des forêts tourbeuses du nord-ouest québécois. Cette étude visait de façon spécifique à :

Cette thèse s'articule autour de trois objectifs spécifiques qui sont : mieux comprendre la dynamique naturelle de séquestration de C après feu, quantifier les impacts des pratiques sur les pertes de C de la couche organique du sol et déterminer les impacts globaux des pratiques. Elle peut être regroupée sous deux principaux axes d'étude : empirique et modélisation. L'étude empirique avait pour objectif de collecter des données de calibration pour la modélisation. Elle évaluait les effets à court terme (c-à-d, neuf ans après) de la coupe et de la préparation mécanique de terrain sur la décomposition des bryophytes et sur les stocks de C de la couche organique. La modélisation de la séquestration du C après feu, qui incluait des sévérités et des cycles de feu différents, avait pour objectif d'établir une base de comparaison des pratiques sylvicoles. Ainsi, à partir de cette base, les effets de diverses pratiques sylvicoles et des aménagements d'intensités différentes ont été évalué par modélisation. L'étude sur le terrain a montré que les préparations mécaniques de terrain (MSP) accéléraient les pertes de C par décomposition de la matière organique fraiche des bryophytes d'environ deux fois et demie, dans les vingt premiers centimètres du sol, par rapport aux sites non coupés, neuf ans après traitement. Elles réduisaient également les stocks de C dans la partie supérieure de la couche organique du sol mais n'avaient aucun impact sur les stocks totaux de C du sol comparé aux sites non coupés. Ces résultats ont confirmé l'impact de la sévérité des traitements au sol sur la séquestration du C en forêt tourbeuse. Ils nous ont donné une idée de la magnitude des effets des perturbations et nous ont renseignés sur les éléments à considérer pour le volet modélisation. La simulation à long terme a montré que les perturbations peuvent avoir des effets contrastés selon les types de forêt considérés (i.e., forêt à dominance de mousses hypnacées vs forêt à dominance de sphaignes). Les perturbations sévères du sol (i.e., Feu sévère, préparation mécanique du terrain et brûlage dirigé suivie de plantation) qui ont favorisé la croissance des arbres, ont entrainé un stockage de C plus rapide dans les forêts à dominance des mousses hypnacées. Au contraire, les perturbations moins sévères comme les coupes avec protection de la régénération et des sols (CPRS) ont entrainé un stockage de C plus rapide dans les forêts à dominance de sphaignes dû au sol organique provenant des mousses. Un autre point essentiel est que la composante arbre est la clé de recouvrement rapide du bilan C indépendamment des types de forêts tandis que les mousses jouent un rôle important dans le maintien du puits de C à long terme. Ainsi, les perturbations sévères ont compensé plus rapidement l'importante perte occasionnée à court terme comparées aux perturbations peu sévères. Au niveau du paysage, les longs cycles de feu tout comme les aménagements de faible intensité tels que les coupes partielles, avec le vieillissement du paysage, accumulent beaucoup de C dans le sol organique tandis que les aménagements qui rajeunissement le paysage avec une proportion plus importante de pratiques sylvicoles plus sévères, présentent les plus faibles stocks. Cependant, ces stratégies rajeunissant le couvert forestier maintiennent l'accumulation de C dans la biomasse vivante des arbres, laquelle décline dans les aménagements favorisant le vieillissement du paysage. Notre étude montre qu'une bonne caractérisation du type de peuplement est essentielle pour mieux prédire les effets des perturbations sur la dynamique du C des écosystèmes. Elle suggère que dans les zones paludifiées, les pratiques de sylviculture intensive et de plantation peuvent offrir un compromis pour les peuplements encore productifs, tandis que dans les sites à très faible productivité, les stratégies de conservation devraient être envisagées. Elle pourrait orienter les décisions de gestion forestière en classant les pratiques en fonction de l'état et du type de forêt initiale et des stratégies d'atténuation du changement climatique.

Mots-clés : bryophytes, carbone, couche organique, incendies, pratiques forestières, paludification.

INTRODUCTION

Le réchauffement climatique est une problématique largement reconnue au niveau mondial (IPCC, 2021). L'augmentation continuelle des émissions des gaz à effets de serre (GES) causée par les activités humaines a entrainé une hausse des températures mondiales de 1,1°C en moyenne depuis l'époque préindustrielle (IPCC, 2014, 2021). Si aucune action n'est menée, une augmentation globale des températures d'environ 4,8°C est prévue d'ici 2100 selon les modèles climatiques en projection du scénario extrême d'émission de GES (RCP8. 5 ou SSP5-8.5) (IPCC, 2014, 2021). Or, 60% des émissions anthropiques sont éliminées de l'atmosphère par les océans, la végétation et le sol (IPCC, 2014). Les forêts constituent les principaux réservoirs terrestres de carbone (C) (Gower, 2003; Pan *et al.*, 2011). Ainsi, l'une des contributions à l'atténuation du changement climatique promue par le protocole de Kyoto et la convention-cadre des Nations unies sur les changements climatiques (CCNUCC) est l'adoption de pratiques de récolte et d'aménagement de moindres impacts sur les forêts, notamment boréales où la déforestation est minime par rapport aux forêts tropicales (Ashton *et al.*, 2012).

Forêt boréale tourbeuse et carbone. ***Principales caractéristiques de la forêt boréale.*** La forêt boréale représente environ un tiers des forêts du monde (environ 1,4 milliards d'hectares) et séquestre dans la végétation et le sol 49% du C forestier mondial (Ashton *et al.*, 2012; Dixon, 1994 ; Kuusela, 1992). Formant une ceinture de l'Eurasie jusqu'à l'Amérique du Nord (Bonan and Shugart, 1989), la forêt boréale couvre une grande partie de l'Alaska, du Canada, de la Fennoscandie, de la Russie, du nord de la Mongolie et du nord-est de la Chine (Ashton *et al.*, 2012). Environ 28% de la forêt boréale se trouve au Canada (Ressources naturelles Canada, 2016). Le bois constitue une ressource naturelle importante pour l'économie de ces pays et une source d'approvisionnement mondial (Ashton *et al.*, 2012). La forêt boréale est dominée par les conifères, dont les épinettes (*Picea* spp.) qui sont les plus répandues, les pins (*Pinus* spp.), les mélèzes (*Larix* spp.), et les sapins (*Abies* spp.) (Bonan and Shugart, 1989; Hagner, 1999). Certains feuillus, principalement les peupliers (*Populus* spp.) et les bouleaux (*Betula* spp.), sont mélangés aux conifères et occupent les sites les plus riches. Le sous-bois est principalement dominé par les bryophytes et les lichens (Bonan and Shugart, 1989).

L'une des caractéristiques des stocks de C en forêt boréale est que la majorité du C (60%) se trouve dans le sol (Bradshaw and Warkentin, 2015 ; Pan *et al.*, 2011). Étant dans les hautes latitudes de l'hémisphère nord, le climat boréal est un climat continental avec des hivers longs extrêmement froids et secs (Bonan and Shugart, 1989). Les températures froides ralentissent la décomposition de la matière organique du sol qui finit par s'accumuler au fil du temps, faisant de la forêt boréale le plus large réservoir mondial du C du sol (Bradshaw and Warkentin, 2015). Cette couche organique peut s'épaissir davantage sous forme de tourbe dans les régions mal drainées et dominées par les bryophytes, et atteindre plusieurs mètres (Clymo *et al.*, 1998; Roulet *et al.*, 2007).

La formation de tourbe est un phénomène qui se déroule sur plusieurs siècles. Le développement en hauteur varie de 0,06 à 3,8 mm.an^{-1} (Lavoie *et al.*, 2005a). Les stocks actuels de C du sol de la plupart des tourbières proviennent de l'accumulation de la tourbe depuis la fin de la dernière période glaciaire (Zoltai and Martikainen, 1996). La limite minimale de l'épaisseur du sol organique pour être qualifiée de tourbe est variable selon les pays. Elle est de 40 cm au Canada tandis que dans d'autres pays comme l'Allemagne ou la Suède elle est de 20 et 30 cm respectivement (Lavoie *et al.*, 2005). On distingue deux types d'entourbement : le comblement qui est l'entourbement des lacs ou des plans d'eau, et la paludification qui est l'entourbement des sites terrestres. Cependant, le principal processus de formation de tourbe en forêt boréale est la paludification (Charman, 2002; Payette and Rochefort, 2001). On parle alors de tourbière ou de forêt tourbeuse ou paludifiée, en cas de présence d'arbres.

Forêt boréale tourbeuse et carbone. ***Paludification et dynamique de C dans les forêts tourbeuses.*** Les forêts paludifiées sont définies comme des tourbières constitués d'arbres d'au moins 5 m de hauteur (Thompson *et al.*, 2016; Zoltai and Martikainen, 1996) pouvant fournir un volume de bois d'au moins 100 m^{3} ha^{-1} à l'âge de maturé (100 à 120 ans) (Payette and Rochefort, 2001). Le C est séquestré à la fois dans les arbres et la tourbe (Bona, 2014; Bonan and Shugart, 1989).

La paludification peut se produire dans les bassins humides et de faible pente (édaphique ou primaire), mais aussi sur des sols bien drainés (paludification secondaire) (Charman, 2002; Payette and Rochefort, 2001). La paludification secondaire, souvent appelée paludification successionnelle, est dirigée par la succession forestière (Lavoie *et al.*, 2005; Payette and Rochefort, 2001). En effet, en l'absence prolongée de feu, l'accumulation progressive de matière organique dans les forêts réduit le drainage (élévation du niveau de la nappe phréatique), la température, le pH du sol et l'activité microbienne au point d'induire la paludification à long terme (Lavoie *et al.*, 2005). Les sphaignes sont deux à cinq fois plus productives et difficilement décomposables comparées aux mousses hypnacées dans les zones mal drainées (Payette and Rochefort, 2001). L'expansion des sphaignes au détriment des mousses hypnacées acidifie les sols forestiers et déplace l'eau de la nappe phréatique du sol minéral vers l'organique, modifiant ainsi le bilan hydrique (Lavoie *et al.*, 2005). L'absence d'oxygène liée à la présence continue d'eau et l'acidité du milieu ralentit considérablement la dégradation de la matière organique morte (MOM) des végétaux par les microorganismes (Payette and Rochefort, 2001). L'épaississement du sol organique fait migrer graduellement la zone d'enracinement des arbres de l'horizon minéral à l'horizon organique, ce dernier étant un moins bon substrat de croissance (Munson and Timmer, 1989; Roy *et al.*, 1999). L'épinette noire (*Picea mariana* [Mill.] BSP) du fait de sa capacité de reproduction par marcottage et le développement de racines superficielles, ainsi que les éricacées grâce à leur reproduction par drageonnement, sont parmi les espèces vasculaires les mieux adaptées à ces milieux (Bonan and Shugart, 1989; Payette and Rochefort, 2001). Par ailleurs, grâce à leur forte capacité d'absorption d'eau, les mousses immobilisent les nutriments des précipitations (Bonan and Shugart, 1989). La compétition de la végétation abondante du sous-bois (sphaignes et les éricacées) et la production de composés allélopathiques des éricacées (*Rhododendron groenlandicum* (Oeder) Kron & Judd, *Kalmia angustifolia* L.) agissent négativement sur le développement des semis et ralentissent la croissance des arbres (Mallik, 1987; Thiffault *et al.*, 2013; Zhu and Mallik, 1994). Les arbres croissent donc dans des environnements humides, froids et pauvres en éléments nutritifs, entraînant une diminution de leur croissance (Simard *et al.*, 2007; Thiffault *et al.*, 2013). Dans les tourbières ombrotrophes, les plus abondantes au nord du Québec et qui sont alimentées uniquement par les précipitations, le milieu s'acidifie davantage et s'appauvrit en espèces vasculaires contrairement aux tourbières minérotrophes dont l'apport en nutriments provient aussi des eaux souterraines et de ruissellement (Lavoie *et al.*, 2005; Payette and Rochefort, 2001). Les forêts tourbeuses sont alors caractérisées par des peuplements d'épinettes noires, une couverture dense de mousses (*Pleurozium schreberi*, *Ptilium crista-castrensis*, *Hylocomium splendens*) et de sphaignes (*Sphagnum* spp.) et d'arbustes éricacées avec une épaisse couche organique de sol (L, F et H ou organique provenant des mousses, système canadien de classification des sols) (Bonan and Shugart, 1989; Fenton and Bergeron, 2006; Lavoie *et al.*, 2005).

Dans la ceinture d'argile à l'ouest du Québec, la paludification réduit la productivité des épinettes noires (Simard *et al.*, 2007). En effet, la ceinture d'argile découlant du retrait d'un ancien lac proglaciaire est caractérisée par des conditions favorisant la paludification dont une topographie plane, un faible drainage des sols argileux, et un allongement du cycle de feu depuis 1920 (*i.e.*, 140 à 400 ans) (Bergeron *et al.*, 2004; Bergeron *et al.*, 2001; Boulanger *et al.*, 2013; Gauthier *et al.*, 2015). Cependant, tandis que les deux types de paludification peuvent coexister, la paludification successionnelle peut être réversible par les feux consumant la quasi-totalité de la tourbe et par la préparation de terrain, contrairement à la paludification édaphique (Laamrani *et al.*, 2015; Lafleur *et al.*, 2018; Simard *et al.*, 2007). Notre étude s'est donc focalisée sur les forêts paludifiées ou en transition, issues de la succession forestière.

Forêt boréale tourbeuse et carbone. ***Cycle du carbone en forêt boréale paludifiée.*** Le cycle du C forestier est l'ensemble des processus d'échange de C entre la forêt et l'atmosphère. C'est un processus écologique global complexe impliquant différents stocks, des flux d'absorption et d'émission de C (Figure 1) (Ashton *et al.*, 2012). Le C atmosphérique est capté par la végétation à travers la production primaire nette (PPN), qui est la différence entre la photosynthèse et la respiration autotrophique des plantes et se traduit globalement par la croissance des végétaux (Chapin *et al.*, 2006; Kurz *et al.*, 2013). Il est réémis dans l'atmosphère, majoritairement sous forme de CO_2, par la respiration hétérotrophe microbienne et de méthane (CH_4). Cette dernière est aussi traduite par la décomposition de la matière organique morte transférée au sol (*i.e.*, bois mort, litière, racines, tourbe) (Kurz *et al.*, 2013). Le C organique du sol peut s'infiltrer par lixiviation en profondeur, dans le sol minéral, ou être perdu dans les eaux souterraines et les rivières (C organique dissout, COD) (Lavoie *et al.*, 2005; Payette and Rochefort, 2001). L'apport du C dans le sol organique provient également d'exsudats racinaires et des champignons symbiotiques (Clemmensen *et al.*, 2013). Les quantités de C accumulées dans les différents compartiments de la forêt désignent les stocks et peuvent être regroupées en réservoirs: biomasse aérienne (partie épigée des plantes vivantes), biomasse souterraine (racines vivantes), bois mort, litière, sol organique et minéral (IPCC, 2006; Kurz *et al.*, 2009).

En forêt paludifiée, les mousses exercent une influence majeure sur le cycle de C. Elles peuvent représenter de 11 à 61 % de leur PPN, et contribuer à près de la moitié de la photosynthèse et du C total de l'écosystème des forêts d'épinettes noires (Bisbee *et al.*, 2001; Bona, 2014). De plus, la matière organique morte des mousses constitue une entrée considérable de C dans le sol organique avec des taux de décomposition faibles (0,0005 – 0,15 an^{-1} pour les sphaignes, et 0,0006 -0.07 an^{-1} pour les mousses plumes) (Harden *et al.*, 1997; Moore and Basiliko, 2006) et un taux d'accumulation de 70-174 g C m^{-2} an^{-1} (Fissore *et al.*, 2019). La tourbe associée au bois mort et à la litière constitue le plus large stock de C en forêt tourbeuse (Moroni *et al.*, 2015).

Les stocks et les flux de C forestier sont soumis à des facteurs de contrôle, mais aussi à des perturbations naturelles (feux, épidémies d'insectes, etc.) ou anthropiques (aménagement, coupe, etc.). Les flux de C, la PPN et la décomposition sont influencés par les facteurs abiotiques et biotiques (climat, sol, hydrologie, etc.), la composition des espèces et l'âge du peuplement (Kurz *et al.*, 2013). Le taux de décomposition de la tourbe dépend précisément de la température du sol, de la profondeur de la nappe phréatique et de l'épaisseur et de la composition de la tourbe (Cornwell *et al.*, 2008; Preston *et al.*, 2012; Scanlon and Moore, 2000). Par exemple, un réchauffement du sol de 5°C ou un abaissement de la nappe phréatique de 10 à 30 cm, en favorisant l'activité microbienne et la décomposition de la tourbe, peut entrainer un doublement des émissions de C (Payette and Rochefort, 2001). Le niveau de la nappe phréatique modifie la vitesse de décomposition de la matière organique morte et entraine une démarcation de tourbe en deux types de couches : l'acrotelme au-dessus de la nappe, et le catotelme en dessous (Payette and Rochefort, 2001). La décomposition, à travers l'activité microbienne, est plus rapide dans l'acrotelme, en raison de l'alternance de conditions aérobies et anaérobies induites par les fluctuations de la nappe phréatique. À l'inverse, le catotelme est une zone anaérobique, et est par conséquent le lieu d'accumulation de la tourbe et d'émission du CH_4 (Payette and Rochefort, 2001). Bien que les quantités de gaz émises soient fortement liées à la position de la nappe phréatique, les tourbières émettent en moyenne 8,8 t CO_2 ha^{-1}contre 0,85 tonne par hectare de CH_4 (Trettin *et al.*, 1995). Par ailleurs, la matière organique morte au sol peut être plus ou moins facilement décomposable par les microorganismes selon les espèces végétales. Par exemple, les aiguilles d'épinettes noires contenant de forts taux de lignine et les sphaignes, riches en cellulose, sont des substrats difficilement décomposables comparés aux mousses hypnacées ou aux herbacées (Bonan and Shugart, 1989; Fenton *et al.*, 2010). De façon générale, les jeunes forêts, en raison de la pleine croissance des arbres, ont des flux d'entrée de C (PPN) plus importants que les forêts plus âgées (stade mature ou suranné) (Kurz *et al.*, 1993; Pregitzer and Euskirchen, 2004; Ryan *et al.*, 1997).

Il existe différents indicateurs pour juger de la capacité de la forêt à séquestrer du C. La production nette de l'écosystème (PNE) ou delta écosystème est la résultante des changements annuels de stocks de C et représente aussi la différence entre la PPN, i.e. la production totale de C organique par les plantes (biomasse et MOM), et la décomposition (Chapin *et al.*, 2006; Kurz *et al.*, 2013). Elle permet de quantifier l'accumulation ou la perte de C et de dire si l'écosystème est un puits (PNE positif), une source de C (PNE négatif) ou en équilibre pendant une période donnée (Kurz *et al.*, 2013). L'âge de la forêt expliquerait 92% de la variabilité totale de la PNE (Pregitzer and Euskirchen, 2004). En général, les forêts, avec une augmentation graduelle des stocks de C, ont un bilan PNE positif ou en équilibre à long terme, mais peuvent se transformer en source à la suite des perturbations naturelles ou anthropiques (Goulden *et al.*, 2011). Les perturbations sont alors traduites par la production nette du biome (PNB) qui déduit les pertes dues aux perturbations (Per) de la PNE (PNB= PNE − Per) (Chapin *et al.*, 2006).

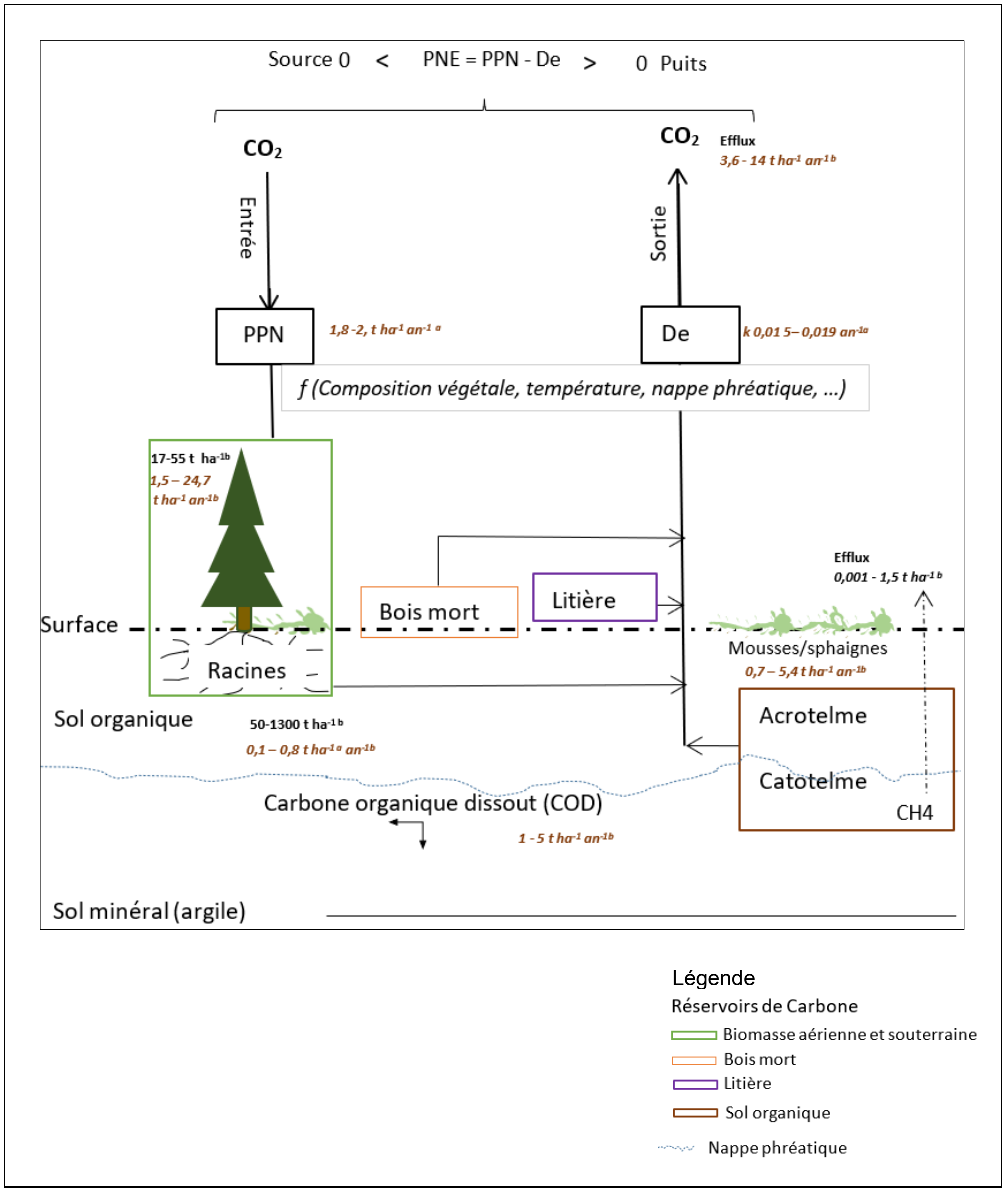


Figure 1
Cycle de C simplifié et réservoirs de C (biomasse-arbres, bois mort, litière, biomasse-mousse, tourbe) en forêt tourbeuse, inspirée de Payette and Rochefort (2001). PPN : productivité primaire nette, De : Décomposition. Les valeurs en noires représentent les stocks (en tonne par hectare, t C ha^{-1}= Mg C ha^{-1} = 10^{6} g C ha^{-1}). Les valeurs en marrons italiques représentent les flux (entrés et sorties) : taux d'accumulation et de croissance (t C ha^{-1} an^{-1}), k taux de décomposition constante de la tourbe (an^{-1}). Les efflux se réfèrent aux émissions totales issues de la décomposition et la respiration des plantes. Perturbations naturelles et anthropiques.

Une perturbation est un événement naturel (feux, épidémies, chablis, etc.) ou anthropique (coupes, etc.) qui cause un changement temporaire positif ou négatif à un milieu (Jentsch and White, 2019). L'une des différences entre les différents types de perturbations réside dans leur fréquence (cycle) et intensité (étendue et sévérité). En forêt boréale, les feux tout comme les coupes en agissant sur la dynamique forestière sont d'importants perturbateurs des stocks et des flux de C (Bradshaw and Warkentin, 2015; Deluca and Boisvenue, 2012). En général, les perturbations entraînent une redistribution du C dans les réservoirs de l'écosystème, ou bien un transfert du C vers l'atmosphère ou les produits bois (Kurz *et al.*, 2013). En fonction du type de perturbation, l'élimination de tout ou partie de la végétation vivante réduit l'apport en C vers le sol.

Feux et dynamique de C. Les feux entraînent une émission directe de C par la combustion de la biomasse (Terrier, 2017; Zoltai *et al.*, 1998) et rajeunissent les peuplements en structure et en composition (Harper *et al.*, 2002; Lecomte and Bergeron, 2005). La combustion de la tourbe rend disponible certains éléments minéraux, stimule le recyclage des nutriments et crée des microsites favorables à la germination et la croissance des arbres.. Après un feu, la décomposition de la tourbe est favorisée par une augmentation de la température, de l'aération, du pH du sol par les cendres et une réduction de l'humidité du sol (Jayen *et al.*, 2006 ;Terrier *et al.*, 2012). La forêt se comporte alors comme une source de C (PNE négative) jusqu'à ce qu'elle se régénère (e.g., 0-30 ans en forêt boréale), puis en puits avec un maximum de C séquestré à maturité dans les arbres (Pregitzer and Euskirchen, 2004; Ryan *et al.*, 1997). Avec le vieillissement, bien que les stocks (biomasses vivantes et mortes) soient plus importants, la production primaire nette des arbres diminue (Gao *et al.*, 2017; Ryan *et al.*, 1997), et la forêt est proche de l'équilibre en termes d'échange de C (PNE neutre) (Fleming *et al.*, 2002). Cependant, en forêts entourbées, l'équilibre serait rarement atteint dû à l'importante entrée du C dans la tourbe et à sa lente décomposition (Lecomte *et al.*, 2006).

Le comportement du feu (vitesse de propagation du feu, consommation de combustible, intensité, type de feu - feu de cime ou de surface - et profondeur du brûlage) entraine différentes trajectoires de la dynamique forestière (Lecomte *et al.*, 2006; Terrier *et al.*, 2015). La sévérité des feux se définit comme l'impact du feu sur un écosystème donné (Lentile *et al.*, 2006; Terrier *et al.*, 2015). Dans la ceinture d'argile, l'allongement du cycle de feu et la faible sévérité du feu au sol contrôlent la paludification, et donc la composition et la structure des forêts (Lecomte *et al.*, 2005; Simard *et al.*, 2007). La sévérité des feux au sol peut se traduire soit par la quantité de matière organique consumée (Miyanishi and Johnson, 2002) ou par la quantité de la matière organique résiduelle (MOR) après feu (Greene *et al.*, 2005; Nguyen-Xuan *et al.*, 2000). Les feux sévères (MOR < 5 cm d'épaisseur), consument presque totalement la couche organique tandis que les feux légers (MOR > 5 cm d'épaisseur), de faible intensité, ne la consument que partiellement (Lecomte *et al.*, 2005).

Après un feu sévère, les peuplements sont caractérisés par la fermeture de la canopée vers 100 ans, dès la première cohorte (*i.e.,* premiers arbres installés après feu) (Lecomte *et al.*, 2006; Simard *et al.*, 2007). Les peuplements sont de structure équienne, dense et productive (*e.g.* 14 m à 50 ans comme indice de qualité de station - IQS), favorisée par une couche organique mince et une abondance de mousses hypnacées (Simard *et al.*, 2007). Par la suite, à la sénescence des arbres de la première cohorte (vers 150-200 ans), la hauteur moyenne des peuplements des cohortes suivantes (2 et 3) diminue et la canopée s'ouvre de plus en plus (Figure 2), en raison de l'accumulation progressive de la matière organique au sol en absence de feu (Lecomte *et al.*, 2006; Simard *et al.*, 2007). En effet, la productivité des peuplements est modifiée par le retrait progressif de l'enracinement des arbres dans le sol minéral pour ne se retrouver que dans la couche organique, substrat aux moins bonnes conditions de croissance et de germination (Lecomte *et al.*, 2006; Simard *et al.*, 2007). Les peuplements présentent alors des structures irrégulières (biétagées à multiétagées), moins productives, avec une couche organique plus importante et un sous-bois abondant (Bergeron *et al.*, 1999; Simard *et al.*, 2007). Ainsi, la quantité de biomasses vivantes des arbres diminue en l'absence prolongée de feu tandis que celle du sol organique continue d'augmenter (Lecomte *et al.*, 2006).

Contrairement aux feux sévères, les peuplements issus des feux légers deviennent rapidement plus ouverts et moins productifs (*e.g.,* 12 IQS) avec plus de sphaignes, près de 50 ans après feu (Lecomte *et al.*, 2006; Simard *et al.*, 2007). Les feux peu sévères laissent des arbres sur pieds et créent des microsites de croissances des arbres très variables (Simard *et al.*, 2007). Ils favorisent ainsi des peuplements de début de succession de structure irrégulière relativement similaire à celle des vieux peuplements issus de feu sévère (Lecomte *et al.*, 2006; Simard *et al.*, 2007). De plus, après un feu peu sévère, la matière organique non consumée et son accumulation continue en l'absence prolongée de feu maintiennent les peuplements dans un état paludifié (Simard *et al.*, 2007). Cette paludification successionnelle peut être réversible en présence de feux sévères consumant la quasi-totalité de la tourbe. Les feux légers entrainent comparativement aux feux sévères, une baisse de 50% de la biomasse des arbres et une augmentation de 75% de l'épaisseur de la tourbe des peuplements dans les 100 premières années (Lecomte *et al.*, 2006). Cependant, la différence de structure entre les peuplements des feux sévères et légers disparait avec le temps (plus de 200 ans après feu) (Lecomte *et al.*, 2006). Les peuplements dominés par l'épinette noire convergent vers une structure relativement ouverte et peu productive due à la paludification. Les mousses hypnacées à décomposition relativement rapide sont remplacées par les sphaignes à décomposition plus lente (Fenton *et al.*, 2010; Simard *et al.*, 2007). Par exemple, *Pleurozium schreberi* est une mousse hypnacée de début de succession forestière, remplacée successivement par *Sphagnum capillifolium* (sphaignes de début de succession) et *Sphagnum fuscum* (sphaignes de fin de succession) (Fenton and Bergeron, 2006).

Ces différentes trajectoires forestières, liées à la sévérité et au cycle du feu, peuvent donc influencer la séquestration et les émissions de C dans les milieux (Lecomte *et al.*, 2006). Cependant, peu d'études existent sur la dynamique à long terme du C après un feu en forêt boréale tourbeuse (Gao *et al.*, 2017). De plus, les différents régimes des feux (cycle, sévérité, taille) rendent difficile la prédiction des flux de C (Lavoie *et al.*, 2005).

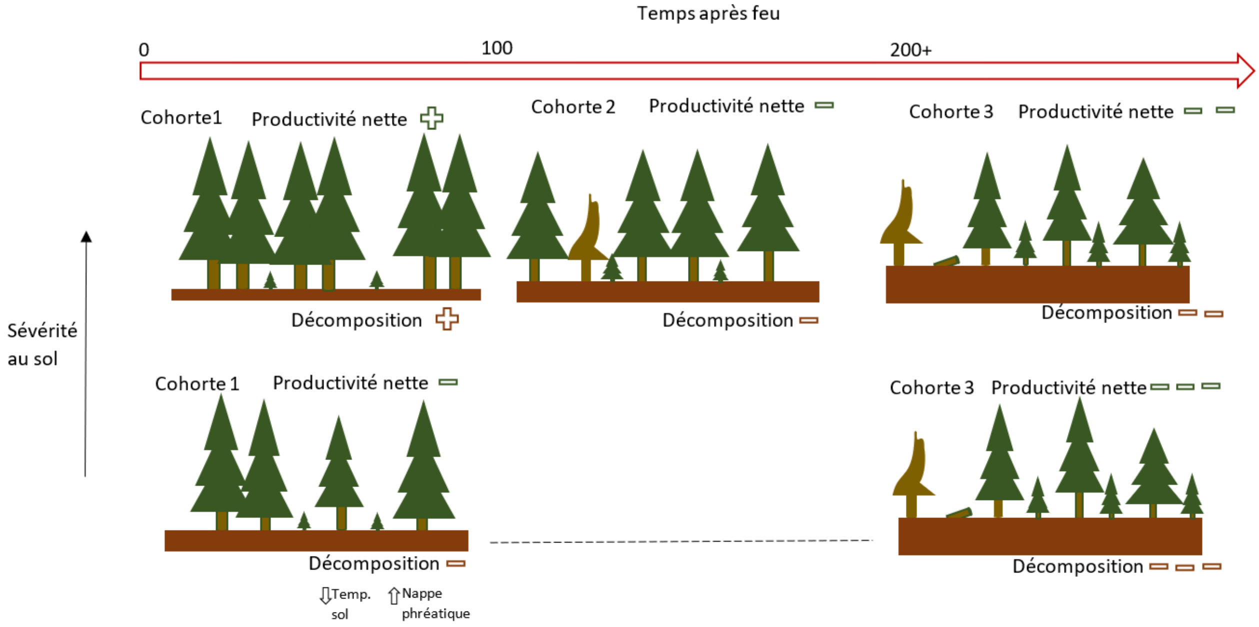


Figure 2
Effets du feu sur la dynamique des forêts tourbeuses ; symboles :+ − plus ou moins de productivité faisant référence à la biomasse vivante ; + − plus ou moins de décomposition faisant référence à la MOM ; arbres verts et marrons : épinettes vigoureuses et épinettes en voie de sénescence ou mortes ; température du sol (temp.sol) et niveau de la nappe phréatique

Perturbation anthropique : pratiques sylvicoles et aménagement forestier. ***Pratiques sylvicoles.*** Les coupes forestières entrainent une exportation du C de la biomasse aérienne des arbres vers le secteur des produits ligneux et dans une moindre mesure vers la MOM du sol à travers les débris (Kurz *et al.*, 2013; Trettin *et al.*, 1992). Contrairement aux feux, en dehors des émissions liées à la machinerie, les coupes n’entrainent pas d’émission directe de C dans l’atmosphère.

La coupe totale et la coupe partielle sont des pratiques sylvicoles d’intensité et de fréquence différentes pouvant affecter les flux et les stocks de C (Lavoie *et al.*, 2005). La coupe partielle (CP) consiste à ne récolter qu’une partie des arbres matures d’une forêt, tandis qu'avec la coupe totale (CT), la quasi-totalité des arbres matures d'un peuplement est récoltée en une seule rotation (e.g., 80 à 100 ans). La coupe avec protection de la régénération et du sol (CPRS) est la coupe totale actuellement pratiquée en forêt boréale depuis 1990. Elle favoriserait la paludification dans la ceinture d'argile du fait de son faible impact sur le sol comparé à la coupe à blanc (CB) anciennement pratiquée (Gauthier and Vaillancourt, 2008; Lavoie *et al.*, 2005). La CB réduit le couvert de sphaignes et d'éricacées, favorise la décomposition et crée des microsites favorables à la croissance des arbres comparée à la CPRS (Lafleur *et al.*, 2010; Lafleur *et al.*, 2016). Cette dernière, malgré la densité de régénération élevée induite, réduit la séquestration de C dans les arbres, mais favorise l'accumulation des stocks de C du sol (Lafleur *et al.*, 2010; Lafleur *et al.*, 2016). De plus, les CPRS sont souvent réalisées en hiver dans les forêts paludifiées pour éviter les ornières, ce qui limite encore plus son impact au sol et maintient intacte la couche de sphaignes et d'éricacées (Payette and Rochefort, 2001). Cependant, les coupes totales peuvent aussi réduire les apports de C du sol issu des bois morts (Gauthier and Vaillancourt, 2008; Moroni *et al.*, 2015).

La CP est encore en expérimentation en forêts paludifiées et peut présenter des taux de rétention des arbres pouvant aller jusqu'à 80% (Gauthier and Vaillancourt, 2008). Le maintien du couvert forestier des CP garde le niveau de la nappe phréatique bas par le maintien de l'évapotranspiration et l'interception des précipitations. De plus, le maintien de l'ombrage réduit la prolifération des sphaignes (Fenton *et al.*, 2010; Fenton *et al.*, 2009). Ainsi, la CP favoriserait dans une moindre mesure la paludification malgré le faible impact au sol comparé à la coupe totale (Lavoie *et al.*, 2005). En outre, par l'allongement des rotations de coupes, la CP maintient la succession forestière (passage de la cohorte 1 aux cohortes 2 et 3) et permet le vieillissement des forêts (Fenton *et al.*, 2009). Or, dans les peuplements plus âgés, la présence des gros arbres augmente les stocks de C de la végétation (Noormets and Nouvellon, 2015) tandis que le bois mort et l'approvisionnement continuel de la litière contribuent aux stocks de C du sol (Kurz *et al.*, 2013). La CP permettrait donc de minimiser les pertes de C par rapport aux coupes totales (Noormets and Nouvellon, 2015). En revanche, le succès des CP peut être compromis dans des forêts susceptibles à la paludification (*e.g.,* épaisseur de la couche organique >17 cm) (Moussaoui *et al.*, 2019).

Des pratiques sylvicoles plus sévères telles que les préparations de terrain (préparation mécanique du sol -PMS- ou brûlage dirigé) sont préconisées à la suite des CPRS pour lutter contre la paludification (Lafleur *et al.*, 2016; Renard *et al.*, 2016). Ces pratiques sylvicoles, souvent suivies de régénération artificielle (plantation ou ensemencement), favorisent la croissance des semis et des arbres par la remise en disponibilité des éléments minéraux, la réduction de la compétition du sous-bois et l'augmentation des lits de germination (Henneb *et al.*, 2015; Lafleur *et al.*, 2010; Lafleur *et al.*, 2011; Thiffault *et al.*, 2004). Les PMS permettent notamment de remuer en profondeur la couche organique pour exposer le sol minéral tout en réduisant le tapis de mousses et les arbustes éricacées. Plusieurs techniques de PMS, comme le déblaiement des débris de coupe (Paquette, 2008) ou le scarifiage, peuvent agir différemment sur la tourbe selon leur sévérité et les conditions de sites (pente, épaisseur de la tourbe, etc.) (Henneb *et al.*, 2015; Mjöfors *et al.*, 2017). Le scarifiage est la préparation de terrain la plus utilisée en forêt boréale (Gauthier and Vaillancourt, 2008). De façon générale, les pratiques sylvicoles plus sévères stimulent l'activité microbienne par l'augmentation de la température et l'oxydation du sol à court terme (Figure 3) (Lavoie *et al*., 2005; Payette and Rochefort, 2001). Elles entraîneraient une augmentation de la décomposition de la tourbe et une émission de C (Figure 3) (Lavoie *et al.*, 2005; Trettin *et al.*, 1992; Trettin and Jurgensen, 1992) qui pourrait être compensée par la croissance des arbres (Jandl *et al.*, 2007). Les effets des MSP sont souvent comparés aux effets des feux sévères tandis que les coupes sont généralement comparées aux effets des feux légers (Bergeron *et al.*, 2007). Les coupes forestières de faible impact sur le sol pourraient minimiser les pertes du C du sol tout en réduisant le stockage de C dans la biomasse des arbres (Jandl *et al.*, 2007) par rapport aux préparations de terrain.

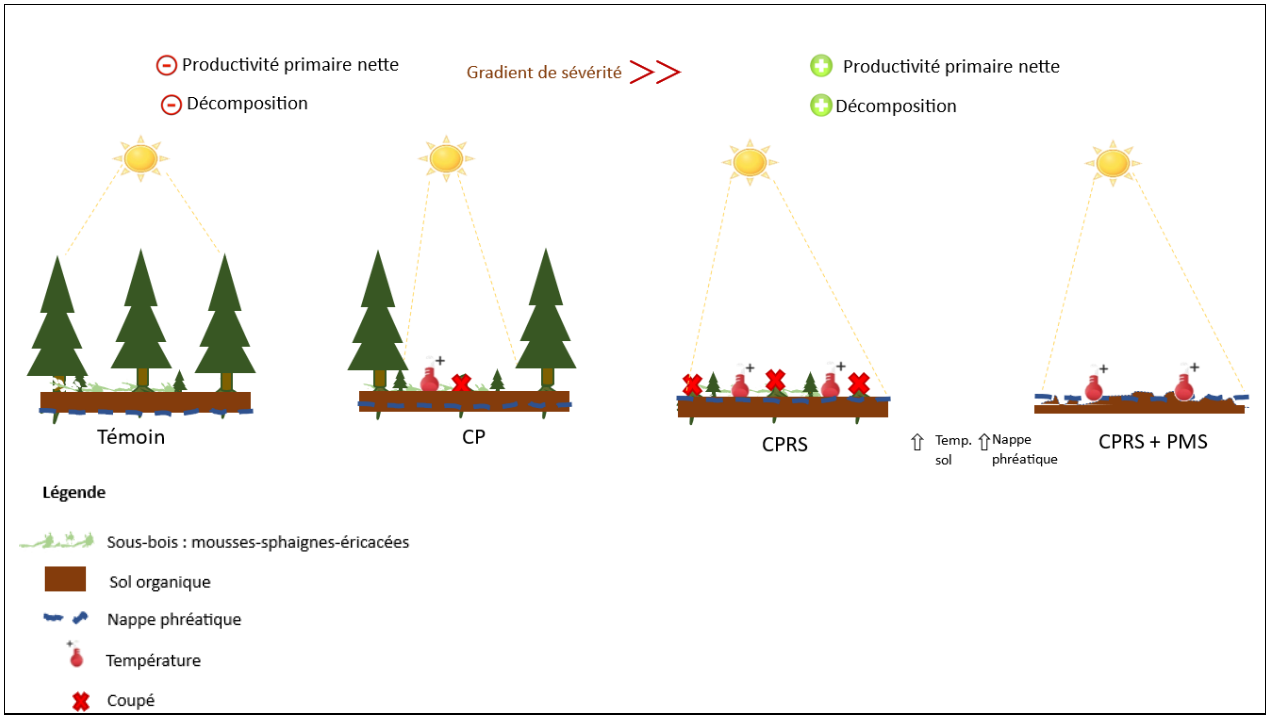


Figure 3
Effets à court terme des pratiques sylvicoles de degré de sévérité au sol différentes (CP : coupe partielle, CPRS : coupe avec protection de la régénération et du sol, CPRS+PMS : CPRS suivie d'une préparation mécanique du sol) en forêts tourbeuses. température du sol : temp.sol et niveau de la nappe phréatique.

Perturbation anthropique : pratiques sylvicoles et aménagement forestier. ***Aménagement forestier***. L'application des pratiques sylvicoles dans le cadre de l'aménagement forestier pourrait donc façonner la distribution des réservoirs et les flux de C au niveau du paysage (*i.e.*, plusieurs centaines de milliers d'hectares) (Boisvenue *et al.*, 2014; Noormets and Nouvellon, 2015). L'aménagement forestier actuel en forêt boréale, dominé par les CPRS, rajeunit la forêt comparée aux perturbations naturelles (feu, épidémies d'insectes ravageurs et chablis) ou un aménagement dominé par les CP (Gauthier and Vaillancourt, 2008). Les peuplements convergent vers des structures simples de cohorte unique (une seule classe d'âge) au détriment des structures complexes de vieilles forêts marquées par plusieurs cohortes (différentes classes d'âge) (Gauthier and Vaillancourt, 2008). Ainsi, l'aménagement écosystémique est de plus en plus promu au Canada et inscrit dans la loi au Québec (Bergeron *et al.*, 2001; Bergeron *et al.*, 1999; Gauthier and Vaillancourt, 2008). Il vise à réduire les écarts entre forêts naturelles et aménagées, en simulant les effets des perturbations naturelles régionales sur la structure et la composition des forêts (Christensen *et al.*, 1996). Dans la ceinture d'argile, l'aménagement écosystémique se résume en l'utilisation des CPRS et de la préparation de terrain pour imiter les effets des feux sévères et assurer la productivité forestière des cohortes 1, et des CP pour préserver les proportions des cohortes 2 et 3 (Gauthier and Vaillancourt, 2008). L'aménagement écosystémique pourrait aussi viser à minimiser les émissions et stocker plus de C par le maintien dans le paysage à la fois de jeunes peuplements productifs et de vieux peuplements (Boisvenue *et al.*, 2014; Taylor *et al.*, 2008). Cependant, les données portant sur les effets de l'aménagement, la productivité ligneuse et la décomposition sont insuffisantes (Fenton *et al.*, 2009; Strömgren *et al.*, 2017; Turetsky *et al.*, 2002). De plus, la prise en compte des changements climatiques s'avère nécessaire sur le devenir du C des forêts tourbeuses aménagées (Boisvenue *et al.*, 2014; Terrier, 2014).

Réponses aux changements climatiques. Dans l'est de la forêt boréale canadienne, plusieurs modèles climatiques prédisent d'ici 2100, une augmentation de 4°C pour la température et une augmentation d'environ 11% (10-20 mm) pour les précipitations en hiver, mais une diminution de 18% durant l'été (Bergeron *et al.*, 2011; Boer *et al.*, 1992; Peng *et al.*, 2002). Les sols s'assècheraient et les risques d'incendie augmenteraient due à l'insuffisance des précipitations comparées à l'augmentation des températures (Hély *et al.*, 2010; Tarnocai, 2006). L'augmentation de la fréquence et de la sévérité des feux (Augustin *et al.*, 2022 ;Bergeron *et al.*, 2011; Boulanger *et al.*, 2014; Hély *et al.*, 2010) pourrait favoriser les émissions directes et indirectes de CO_2 par la décomposition de la MOM, et affecter la dynamique du C par les changements dans la composition et la structure des forêts (Boisvenue *et al.*, 2014; Tarnocai, 2006). En contrepartie, la hausse des températures associée à un allongement de la saison de croissance et l'augmentation des émissions de CO_2 favoriserait la croissance des arbres et donc la séquestration de C dans les arbres (Girardin *et al.*, 2008; Peng *et al.*, 2002). En revanche, dans la ceinture d'argile, les forêts tourbeuses resteraient des puits de C et pourraient résister à la sévérité des feux par le maintien de l'humidité du sol par les sphaignes et l'augmentation des précipitations (Lavoie *et al.*, 2005; Terrier *et al.*, 2014). De nombreuses incertitudes des modèles actuels de bilan de C notamment sur le régime de feu, le changement de composition et de productivité de la forêt rendent difficile la prise en compte de l'effet des changements climatiques sur les flux de C en aménagement (Ameray *et al.*, 2023 ; Augustin *et al.*, 2022 ; Terrier *et al.*, 2014).

Evaluation des impacts des perturbations sur les flux et stocks de C. Selon l'échelle spatiale et temporelle considérée, des méthodes combinant des mesures de terrain et de modélisation peuvent être utilisées pour évaluer les impacts des perturbations sur le C forestier. Les études empiriques sont menées à l'échelle du peuplement tandis que la modélisation se fait à une plus grande échelle spatiale (*e.g.*, paysage) et temporelle (plusieurs centaines d'années) (Anderson *et al.*, 2020; Kurz *et al.*, 2013). La dynamique du C forestier étant complexe, la modélisation est souvent utilisée pour mieux comprendre et quantifier les flux de C dans les différentes composantes (sol et végétation) (Boisvenue *et al.*, 2014; Peltoniemi *et al.*, 2007), et prédire les interactions entre les différents facteurs environnementaux (température, précipitation et hydrologie) et la dynamique du C (Pan *et al.*, 2011; Yu *et al.*, 2012). Combinée aux études empiriques (mesures répétées de biomasse, de C du sol, de flux par Eddy covariance, les données d'inventaire, etc.), elle est nécessaire pour prédire les réponses des forêts tourbeuses à différents scénarios sylvicoles, de perturbations naturelles et de changements climatiques (Kurz *et al.*, 2013; Lavoie *et al.*, 2005; Shaw *et al.*, 2016).

Plusieurs modèles forestiers de dynamique de C forestier existent et utilisent différentes approches de simulation (empiriques, semi-empiriques et processus biochimiques) (Deluca and Boisvenue, 2012; Schmid *et al.*, 2006). Certains de ces modèles sont spécifiques au sol (*e.g.*, YASSO, CENTURY, ROMUL, RothC) et d'autres sont plus généraux et s'intéressent à la fois au C du sol et de la végétation (Biome-BGC, Forest-DNDC, 3-PG, MASSIMO, SILVA, TRIPLEX-FLUX, EFISCEN, CO2FIX, MBC-SFC3) (Deluca and Boisvenue, 2012; Peltoniemi *et al.*, 2007; Taylor *et al.*, 2008). Cependant, il existe encore beaucoup d'incertitudes dans l'estimation des modèles surtout pour le C du sol (Bona, 2014; Kurz *et al.*, 2013). Un modèle de prédiction de la dynamique du C des forêts tourbeuses devrait intégrer à la fois la productivité des végétaux (arbres et mousses) et la décomposition (Lavoie *et al.*, 2005). Cependant, la contribution des mousses dans le cycle du C (Boisvenue *et al.*, 2014; Bona, 2014) et l'intégration de différentes stratégies d'aménagement (Taylor *et al.*, 2008) sont rarement prises en compte dans les modèles. Biome-BGC, basé sur des processus biochimiques complexes et de nombreux paramètres difficilement mesurables (Running and Gower, 1991), fait partie des rares modèles qui incluent toutes les composantes de l'écosystème (arbres, mousses et sol) (Bona, 2014; Bond-Lamberty *et al.*, 2004). De plus, la plupart de ces modèles ne sont pas adaptés aux projets de crédits C et aux méthodes de comptabilisation du C du groupe d'experts intergouvernemental sur l'évolution du climat (GIEC) (Masera *et al.*, 2003).

Le Modèle de Bilan C du Secteur Forestier Canadien (MBC- SFC3) est un modèle semi-empirique de comptabilisation nationale du C forestier conforme aux normes du GIEC. Basé sur des données courantes de planification forestière (*i.e.*, inventaires forestiers et courbes de croissance et de rendement), il inclut les perturbations naturelles et l'aménagement dans la simulation de la dynamique du C de l'écosystème forestier (*i.e.*, croissance des arbres, dépôt de litière et décomposition de la matière organique) (Kurz *et al.*, 2009). Cependant, les stocks et les émissions de C du sol dans CBM-SFC3 présentent de fortes incertitudes et n'incluent pas la tourbe (Bona, 2014; Kurz *et al.*, 2009). Des recherches sont en cours à travers l'élaboration d'un module spécial appelé « CaMP » pour intégrer les particularités des forêts tourbeuses (décomposition de la tourbe, mousse et fluctuation de la nappe phréatique) (Canadian Model for Peatland ; (Bona, 2014; Shaw *et al.*, 2016).

Problématique. Au nord-ouest du Québec, dans la ceinture d'argile, la paludification réduit la productivité ligneuse des épinettes noires de 50 à 80 %, menaçant ainsi la production de bois (Simard *et al.*, 2007). Des pratiques sylvicoles intensives de récolte et de préparation de terrain sont nécessaires pour créer des conditions favorables à la croissance des arbres et au maintien de la production de bois. Cependant, si ces pratiques favorisent la séquestration de C dans les arbres, elles pourraient en contrepartie réduire les stocks de C du sol en favorisant la décomposition et en réduisant le couvert de bryophytes. Une meilleure connaissance des effets de ces pratiques sur les flux de C constitue donc une nécessité dans la maîtrise de l'empreinte C de l'ensemble de l'industrie forestière en milieu paludifié. Dans un souci de lutte contre les changements climatiques, il est nécessaire d'évaluer le bilan C des pratiques sylvicoles qui y sont menées. Tandis que les pratiques sylvicoles intensives sont susceptibles de relâcher une partie des stocks de C du sol dans l'atmosphère et favoriser l'accumulation de C dans la biomasse aérienne et souterraine des arbres, les CPRS et CP pourraient accroître les stocks de C dans leur sol par leur faible impact sur ce dernier (Lavoie *et al.*, 2005). Une connaissance approfondie de l'ensemble des effets de ces pratiques servirait de base au secteur forestier pour élaborer les meilleures stratégies afin de maximiser la production forestière tout en minimisant les émissions de C. Ce doctorat s'intéresse donc à évaluer les effets des pratiques sylvicoles à court et long-terme utilisées dans les forêts boréales paludifiées de la ceinture d'argile en vue de l'identification des pratiques de moindre impact sur le C forestier.

Objectifs et hypothèses

a) Objectifs

L'objectif général de cette thèse est d'évaluer les effets des pratiques de coupes et de préparation de terrain sur les stocks et les flux de C dans les forêts tourbeuses. Elle combine des études de terrain et l'utilisation du modèle de MBC-SFC3.

De façon spécifique, elle vise à :

- Mieux comprendre la dynamique naturelle de séquestration du C en forêt tourbeuse afin d'établir une base de comparaison des pratiques.

- Décrire et évaluer les effets de diverses pratiques de coupe et de préparation de terrain sur la décomposition de la tourbe et les facteurs environnementaux qui l'influencent.

- Quantifier et comparer les impacts de ces pratiques sylvicoles sur les flux et les stocks naturels de C de ces forêts à l'échelle du peuplement et du paysage.

- Contribuer à l'intégration des forêts tourbeuses dans le modèle MBC-SFC3 et le module CaMP.

Cette thèse s'articule autour de trois chapitres :

- Le premier chapitre s'est intéressé à la dynamique naturelle de séquestration de C.

Il s'agit de décrire et de modéliser la capacité des forêts tourbeuses à séquestrer le C en fonction de la sévérité des feux et du temps après feu, dans des conditions naturelles. Cette partie a permis de définir l'état de référence pour comparer les effets des pratiques sylvicoles sur la dynamique des flux de C.

- Le deuxième chapitre a porté sur les effets des pratiques sylvicoles sur la décomposition de la tourbe.

L'amélioration de la connaissance sur la dynamique du C du sol nécessite la compréhension de la décomposition de la tourbe. Dans ce chapitre, basé sur les observations de terrain, nous avons évalué les taux de décomposition et les stocks de C du sol. Ils nous ont permis d'avoir des résultats plus représentatifs de notre zone d'étude et de mieux paramétrer le modèle MBC-SFC3 (taux de décomposition des mousses).

- Le troisième chapitre a permis d'évaluer par modélisation les impacts des pratiques sylvicoles sur les flux et les stocks de C à l'échelle du peuplement et à l'échelle du paysage.

À l’échelle du peuplement, les effets de diverses techniques de récolte et de préparation de terrain ont été prédits à court et long terme. Il s’agit 1) de la CPRS, 2) de la CPRS suivie des PMS (c.-à-d. scarificateurs à disque, herse), 3) de la CP (différentes modalités de coupe) et 4) du brûlage dirigé. Les flux et les stocks de C résultants de ces différents scénarios alternatifs sylvicoles ont été comparés au scénario de référence (peuplement après feu).

À l’échelle du paysage, nous avons combiné différents scénarios de feu et d’aménagement et modélisé les flux et les stocks de C. Nous avons fait varier les proportions du paysage soumises au feu (fréquence variable) et à différents scénarios sylvicoles.

b) Questions et Hypothèses de recherche

- Questions

Il s’agit donc de savoir :

− Q1 : Comment évoluent naturellement (après feu) les stocks et les flux de C dans les forêts tourbeuses ?

− Q2 Comment les travaux sylvicoles influent sur la décomposition et les stocks de C du sol organique à court terme ?

− Q3-a : Comment les travaux sylvicoles influent sur la séquestration et l’émission de C en forêt tourbeuse au fil du temps en agissant sur la productivité et la décomposition ?− Q3-b : À plus grande échelle, quelle tendance suivra la dynamique de C à court et long terme avec l’aménagement actuel ? Peut-on l’inverser par d’autres stratégies d’aménagement ?

- Hypothèses générales

Nous supposons que :

H1-a. Les stocks de C de la biomasse vivante des arbres sont en moyenne plus élevés après feu sévère qu'après feu léger, mais les stocks du sol sont plus élevés après feu léger du fait de la matière organique morte des sphaignes. À long terme (*i.e.,* 400 ans), les stocks totaux (biomasse vivante et sol) sont plus importants après feu léger que feu sévère.

H1-b. Au niveau du paysage, les longs cycles de feu, surtout en présence de feux légers, accumulent plus de stocks de C principalement dans le sol, que les cycles courts en présence de feux sévères (Boisvenue *et al.*, 2014; Lecomte *et al.*, 2006).

H2-a. La décomposition de la MOM est plus élevée dans les PMS (fort impact, température du sol élevée, stimulation de l'activité microbienne) que dans les CPRS (faible impact, température du sol élevée) et les témoins non coupés (Fenton *et al.*, 2010; Prescott *et al.*, 2000).

H2-b. Les stocks de C du sol sont plus faibles dans les PMS que dans les CPRS et les témoins non coupés (Fenton *et al.*, 2010; Prescott *et al.*, 2000; Taylor *et al.*, 2008).

H3-a. À l'échelle du peuplement, les CP, par le maintien d'un peuplement résiduel, une faible décomposition de la MOM et l'abondance de bois morts, stockent à long terme plus de C et émettent moins de CO_2 que les CPRS, les CPRS suivies de PMS et le brûlage dirigé (Fenton *et al.*, 2010; Taylor *et al.*, 2008).

H3-b. À l'échelle du paysage, les longs cycles de feu favoriseront des stocks de C plus élevés provenant du sol, suivi respectivement, de l'aménagement actuel, de l'aménagement incluant plus de pratiques sylvicoles intensives, et des cycles de feu courts (Boisvenue *et al.*, 2014).

Globalement, cette thèse combine une approche par modélisation et une étude de terrain pour évaluer les effets des pratiques de coupes et des feux sur les stocks et les flux de C. Dans le premier chapitre, nous avons utilisé le modèle MBC-SFC3 incluant le module mousse (MBC-SFC3 ; Kurz *et al.*,2009, MOSS-C; Bona *et al.*, 2016) pour simuler les stocks et les flux de C après feu sur une période de 400 ans. MBC-SFC3 est un modèle semi-empirique basé la modélisation statistique et des processus. Cependant, nous avons réimplémenté le module mousse et l'avons adapté pour mieux traduire la dynamique des mousses et l'accumulation des stocks de C dans la tourbe. Le deuxième chapitre repose sur un dispositif expérimental de mesure sur le terrain des taux de décomposition de la matière organique fraiche de 3 types de bryophytes (*Pleurozium schreberi*, *Sphagnum capillifolium* et *Sphagnum Fuscum*) et des stocks du C du sol, neuf après CPRS et des MSP. Dans ce chapitre nous avons aussi évalué les facteurs abiotiques (profondeur, nappe phréatique, horizon du sol, etc.) susceptibles d'être modifiés par les pratiques sylvicoles. Le dernier chapitre basé aussi sur une approche par modélisation comme le premier chapitre, simule la dynamique de séquestration C des pratiques sylvicoles et les stratégies d'aménagements une période de 250 ans.

1. FOREST TYPE DRIVES THE RESPONSE OF BOREAL FORESTED PEATLANDS TO WILDFIRE: A SIMULATION STUDY

Article soumis en novembre 2024 à Canadian Journal of Forest Research: Ange-Marie Botroh, David Paré , Xavier Cavard , Nicole J. Fenton , Kelly Ann Bona , Yves Bergeron.

1.1 Abstract

Boreal black spruce forests contribute to climate change mitigation by accumulating large amounts of carbon (C) in the peat. When left undisturbed, a thick peat layer, associated with the poorly drained soil, can inhibit tree growth, and this trade-off between peat and tree biomass can have implications on the forest C dynamics. Similarly, wildfire severity and frequency can modify C accumulation patterns, but this impact remains poorly documented. We used the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) to explore over a 400-year simulation period, the effects of high and low severity fire on C dynamics of two forest types (black spruce- Sphagnum and black spruce-feathermoss). We found that total carbon stocks increased to higher levels after low than after high severity fire in BSSP due to peat accumulation. Conversely, in black spruce-feathermoss, high severity fire resulted in greater carbon storage than low severity fire due to the higher tree biomass. Tree growth is key to the rapid recovery of carbon pools regardless of forest types, and specifically after high severity fire, while mosses maintain C sinks over the long term. This study suggests that a good characterization of forest type (*Sphagnum* dominance versus feathermoss) is key to better predictions of the effects of a change in fire regime on ecosystem carbon dynamics.

Key words: bryophytes, carbon, fire, organic layer, paludification.

1.2 Résumé

Les forêts boréales d'épinettes noires contribuent à l'atténuation du changement climatique en accumulant de grandes quantités de carbone (C) dans la tourbe. Lorsqu'elle n'est pas perturbée, une épaisse couche de tourbe, associée à un sol mal drainé, peut inhiber la croissance des arbres, et ce compromis entre la tourbe et la biomasse des arbres peut avoir des répercussions sur la dynamique du C de la forêt. De même, la gravité et la fréquence des incendies de forêt peuvent modifier les patrons d'accumulation du C, mais cet impact reste peu documenté. Nous avons utilisé le modèle du bilan du carbone du secteur forestier canadien (CBM-CFS3) pour explorer, sur une période de simulation de 400 ans, les effets des incendies de forte intensité et de faible intensité sur la dynamique du C de deux types de forêts (épinette noire - sphaigne et épinette noire - mousse hypnacée). Nous avons constaté que les stocks totaux de C ont augmenté à des niveaux plus élevés après l'incendie de faible intensité qu'après celui de forte intensité dans la forêt d'épinette noire - sphaigne en raison de l'accumulation de tourbe. Inversement, dans la forêt d' épinette noire - mousse hypnacée , le l'incendie de forte intensité a entraîné un stockage de C plus important que celui de faible intensité en raison de l'importante biomasse des arbres. La croissance des arbres est essentielle à la reconstitution rapide des réservoirs de C indepnedamment des types de forêts, en particulier après le HSF, tandis que les mousses maintiennent les puits de C sur le long terme. Cette étude suggère qu'une bonne caractérisation du type de forêt est essentielle pour mieux prédire les effets d'un changement de régime d'incendie sur la dynamique du C de l'écosystème.

Mots clés : bryophytes, carbone, incendie, couche organique, paludification.

1.3 Introduction

The boreal forest represents an important terrestrial carbon (C) reservoir by storing almost half of the world's C contained in living vegetation and soil (Ashton *et al.*, 2012; Dixon, 1994; Pan *et al.*, 2011). Moreover, more than two thirds of this stock (163 Pg) is found in the soil, due notably to peat formation (Dixon, 1994). Peat accumulates when the rate of productivity of plants, mainly bryophytes (*i.e.,* peat-forming moss such as *Sphagnum* spp.), is greater than the rate of decomposition of organic matter in cold and poorly drained soils (Charman, 2002; Payette and Rochefort, 2001). Paludification is the most common process in the boreal forest, by which forests slowly accumulate enough peat to transition to a peatland system (Charman, 2002; Payette and Rochefort, 2001). In addition to occurring in sites that are wet because of edaphic conditions (*i.e.,* lowlands), paludification can also be initiated in uplands (*i.e.,* well-drained sites) and accentuated by the prolonged absence of fire (Lecomte *et al.*, 2006; Roulet *et al.*, 2007; Sousa *et al.*, 2021).

Fire is the main natural disturbance in the boreal forest with large impacts on C dynamics (Amiro *et al.*, 2009; Bradshaw and Warkentin, 2015; Harden *et al.*, 2000). It can lead to large C losses by consuming biomass (vegetation and soil organic matter) and stimulating soil organic matter decomposition. It also contributes to redistributing C into its different ecosystem pools, and by influencing plant regeneration and stand composition and structure (Smithwick *et al.*, 2005; Terrier, 2017; Van Bellen, 2012; Walker *et al.*, 2018). At the landscape level, C stocks and fluxes are highly dependent on the fire regime (*e.g.*, frequency and severity) (Andrieux *et al.*, 2018; Breshears and Allen, 2002; Miquelajauregui *et al.*, 2018). For example, high severity fires can almost completely consume the organic matter layer while low severity fires consume a much lower proportion (Lecomte *et al.*, 2006; Simon and Schwab, 2005). Fire cycle is defined as the time needed to burn an area equivalent to the study area (Bergeron *et al.*, 2011), while fire severity describes the effects of a fire on biophysical or ecological features of forest ecosystems (Ryan, 2002). Fire severity can be quantified by changes in the fraction of organic matter (whether living or dead) consumed (Kasischke and Bruhwiler, 2002). In the western and central North American boreal forest, relatively short fire cycles (< 100 years) currently predominate while longer fire cycles are found in the eastern boreal forest (> 300 years) (Bergeron *et al.*, 2004; Johnson, 1996).

In the Clay Belt of north-western Québec and north-eastern Ontario (Canada), the fire cycle has extended during the twentieth century from 140 years to more than 400 years (Bergeron *et al.*, 2004; Bergeron *et al.*, 2001; Boulanger *et al.*, 2013; Gauthier *et al.*, 2015). A prolonged absence of fire combined with low fire severity can drive paludification, even in well drained-sites (Lecomte *et al.*, 2007; Payette and Rochefort, 2001). Paludification is characterized by open canopy stands of low productivity dominated by black spruce (*Picea mariana* [Miller]), a high abundance of *Sphagnum* species, a high water table and a thick peat layer (Fenton and Bergeron, 2006; Simard *et al.*, 2007). However, when paludification is reverted due to high fire frequency, or high fire severity, forests can become more productive with dense canopies, low *Sphagnum* ground cover and a thin organic soil layer (Lecomte *et al.*, 2006), particularly in the first century after disturbance. But the difference in stand structure between high severity and low severity fires disappears after a long period of time (*e.g.,* more than 200 years after fire), and forests converge to open and low-productivity stands due to the successional shifts from feathermoss-dominated to *Sphagnum*-dominated bryophyte communities (Lecomte *et al.*, 2006). Paludification results in a 50-80% reduction in black spruce productivity at the landscape scale (Simard *et al.*, 2007). Landscapes liable to paludification are thus characterized by a closed and partially open canopy cover (Belleau *et al*, 2012). The different forest trajectories linked to fire severity can therefore influence C sequestration and emissions in the Clay Belt region. Moreover, in this region, under the climate change projection scenario RCP8.5, the fire cycle is projected to shorten from 400 years to 100 years, accompanied by an increase in fire severity and size (Bergeron *et al.*, 2006; Boulanger *et al.*, 2013; Flannigan *et al.*, 2005). This can have important consequences on forest structure, composition, and thereby C dynamics (Boisvenue *et al*, 2014). Studying the effects of fire regimes (fire severity and cycle length) on carbon storage including moss and peat is therefore important to understanding the potential of forest carbon balance for climate change mitigation in this boreal region, and others prone to paludification (Jandl *et al.*, 2007; Terrier *et al.*, 2015). Modelling can be a useful tool to investigate the potential long-term effects of disturbance on C stock dynamics (Kurz *et al.*, 2013; Lavoie *et al.*, 2005). The Carbon Budget Model of the Canadian Forest Sector version 3 (CBM-CFS3) can account for the dynamics of organic matter originating from mosses and forested peatlands using the MOSS-C plug-in module (Bona, 2014; Bona *et al.*, 2020; Bona *et al.*, 2016; Shaw *et al.*, 2016).

Our objective was to model the natural C dynamics, including mosses, after fire in a boreal region prone to paludification. In this study, we used CBM-CFS3 (Bona *et al.*, 2016; Kurz *et al.*, 2013) to investigate the effects of changes in fire severity and frequency on the carbon storage and balance and their aboveground / belowground ecosystem partition. At the stand level, we compare the C dynamics of the two main forest types found in this landscape: poorly drained black spruce-*Sphagnum* dominated stands (and productive black spruce-feathermoss dominated stands. At the landscape level, including fire severity, we explore the effects of the current fire cycle length (400 years) and a shorter cycle of 100 years, which is projected under the RCP 8.5 climate scenario.

We hypothesized that:

H1. At the stand level, for spruce-Sphagnum dominated stands where C is mainly stored in the peat component, a low severit fire favors *Sphagnum* moss growth, and generates greater ecosystem C accumulation than high severity. Conversely, in black spruce-feathermoss dominated stands, where C is primarily stored in the tree component, high severity fire, which stimulates tree growth, generates a greater ecosystem C accumulation than low severity fire (Lecomte *et al.*, 2006; Simard *et al.*, 2007).

H2. At the landscape level, low severity fire and long fire cycles, with large moss-derived C contributions, result in higher ecosystem C storage compared to short fire cycles and high severity fire.

1.4 Methods

1.4.1 Study area

The study area corresponds to Forest Management Unit (FMU) 085-51 (between 50°09’N and 48°50’N latitudes, and 79°31’W and 78°05’W). FMU085-51, with an area of 991 616.3 ha, lies in the physiographic region of the Clay Belt, in the black spruce-feathermoss bioclimatic domain (Figure 4). Forests are dominated by black spruce (*Picea mariana* [Mill.] BSP), ericaceous shrubs (mainly *Kalmia angustifolia* L. and *Rhododendron groenlandicum* (Oeder) Kron & Judd), feathermosses (mainly *Pleurozium schreberi* [Brid.] Mitten), and *Sphagnum* spp. (e.g., *Sphagnum fuscum* [Schimp.] Klinggr., *Sphagnum capillifolium* [Ehrh.] Hedw.) (Fenton *et al.*, 2010; Harper *et al.*, 2003). In the northern part of our study area, clay is incorporated with fine till that forms a compact deposit called Cochrane till (Daubois *et al.*, 2015). This northern part of the Clay Belt is characterized by a high risk of paludification whereas the southern part has medium and low risk of paludification (Mansuy *et al.*, 2018). The topography is flat, and soils are mainly classified as Gleysols, Luvisols, or Organic (Canadian Agricultural Services Coordinating Committee , Soil Classification Working Group, , 1998). The cold climate is subpolar and subhumid continental with a short growing season length of 150-160 days between May and August, a mean annual temperature of 0.6°C and precipitations of 907.6 mm (Joutel, Lebel Sur Quevillon, Lac Berry stations, 1981-2010 period, Environment and Climate Change Canada, accessed 2021). This cold and moderately humid climate associated with the poorly drained clay soil, the low topographic relief, the prolonged absence of fire and the abundance of *Sphagnum* promote paludification in this area (Bergeron *et al.*, 2001; Fenton *et al.*, 2010; Payette and Rochefort, 2001).

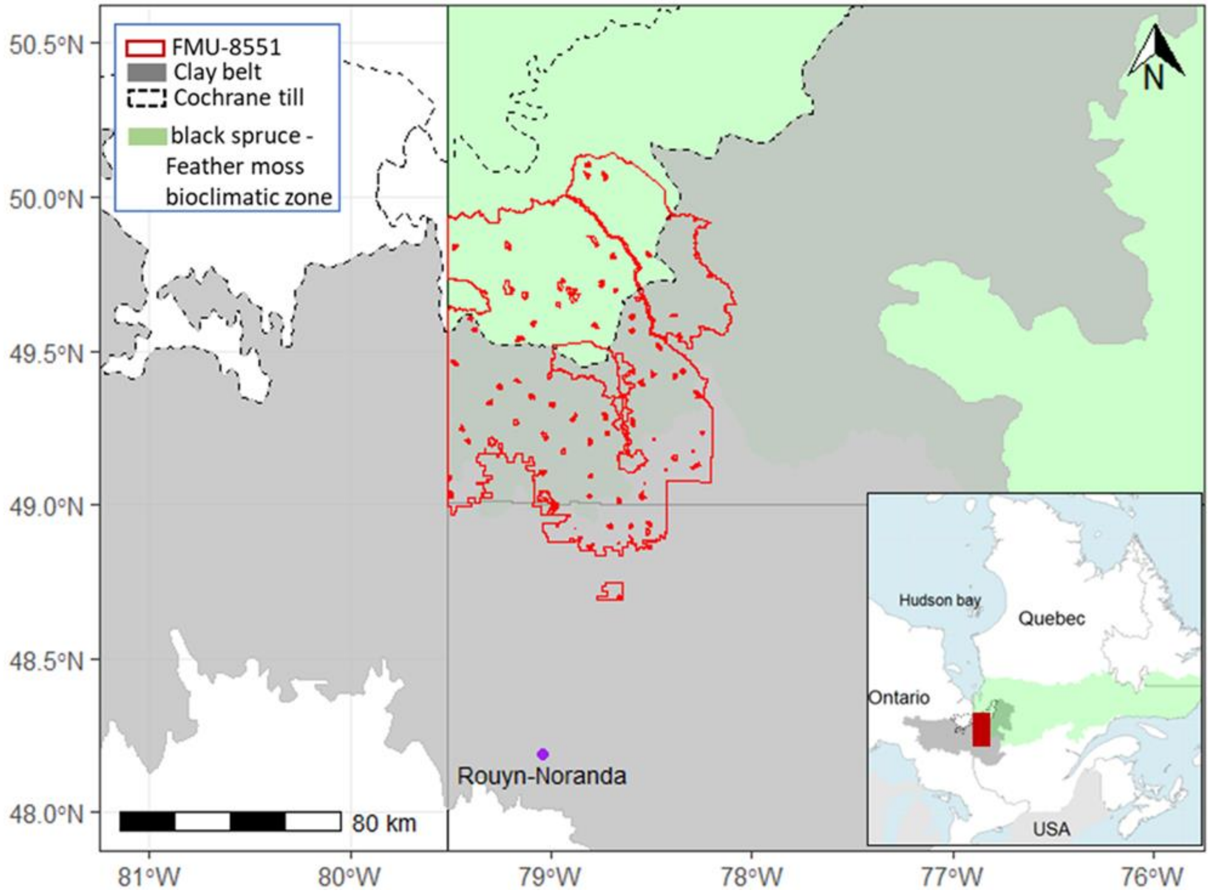


Figure 4
Location of the study area: Forest Management Unit 085-51, in the black spruce-feathermoss bioclimatic domain.

1.4.2 Model description

CBM-CFS3 structure. We used the operational carbon budget model of the Canadian Forest Sector version 3 (CBM-CFS3; Kurz *et al.*, 1992; 2009) to simulate C stocks and fluxes for 400 years after fire. CBM-CFS3 combines statistical and process modeling to track C stocks, stock changes and fluxes in annual time-steps. C pools in CBM-CFS3 can be grouped into two groups: biomass and soil (Figure 5). Soil pools contain dead organic matter (DOM, Litter + debris + dead wood), organic (FH horizon) and mineral soil layers. We used a moss module (MOSS-C; Bona *et al.,* 2016) included in the research version of CBM. This module, based on the CBM pool structure and functioning, tracks the growth of live moss biomass and their turnover into two soil organic layers with contrasting decay rates (fast and slow decay) for two types of bryophytes (feather mosses and *Sphagnum* spp.) (Figure 5). To increase the usability and flexibility of the moss module the MOSS-C module was re-built in R programming language (R version 4; R Development Core Team 2021).

Vegetation growth. The dynamic of living C biomass was based on stand growth simulation. CBM uses allometric equations developed for specific Canadian forest types to transform merchantable wood volume from yield curves to aboveground biomass and to link aboveground biomass to belowground biomass (Boudewyn *et al.*, 2007; Kurz *et al.*, 1996; Li *et al.*, 2003). The wood biomass of non-merchantable and sapling size trees is estimated by curve-smoothing algorithms (Kurz *et al.*, 2009). Biomass was converted to C mass by a dry matter factor of 0.5 g C/g (Kurz *et al.*, 2009; Lamlom and Savidge, 2003). Net Primary Productivity (NPP) is used as C annual input. Moss productivity depends on moss ground cover and NPP (Bona *et al.*, 2016). The merchantable tree volume was used to predict tree canopy openness, which was then used to model bryophyte ground cover and NPP rates for each type of bryophyte. The predicted NPP rates are multiplied by the proportion of bryophyte ground cover of each type to estimate the annual NPP inputs. A delay in regeneration of 10 years was applied after fire (Bona *et al.*, 2016; Zoltai *et al.*, 1998). However, the default relationships of the moss module do not include different fire severities and were adapted to black spruce upland forests, which can feature a denser and closer canopy than forested peatland (Bona et *al.*, 2016). In this study, we adapted and developed different relationships and distinguished the effect of fire severity on moss ground cover based on field data of our study area prone to paludification. Data were collected by Fenton et *al.*, 2005 in the Clay Belt from a chronosequence of 18 black spruce stands, 50 to 350 years after fire. It consisted in a network of 360 quadrats (5 m× 5 m), established in 2003 and used to measure forest floor thickness, fire severity, basal area, canopy closure, and cover of *Sphagnum* spp. and ericaceous spp. (Fenton and Bergeron, 2011; Fenton *et al.*, 2005) (appendix D, chapiter 1). Multiple regression models were applied to these different variables (*e.g.*, canopy closure, basal area) to predict *Sphagnum* species ground cover according to fire severity (appendix E, chapter 1). The ground cover of feathermoss was deducted from the *Sphagnum* species ground cover because we assume that feathermoss and *Sphagnum* species represent 100% of the ground bryophyte cover. The regression models indicate a higher feathermoss cover after HSF than after LSF in young stands, but feathermosses were gradually replaced by *Sphagnum* species at long term (appendix E, chapter 1). Specific constant NPP to feathermoss and *Sphagnum* cited in Bisbee *et al.*, 2001 (appendix C, chapter 1) were multiplied by the proportion of each type of bryophyte ground cover to estimate the annual inputs NPP for biomass.

Turnover and decomposition. Living biomass C is transferred to soil pools through turnover (*i.e.*, biomass mortality) and litterfall rates (Figure 5). Soil pools vary as C is transferred between different soil pools and or lost to the atmosphere through decomposition (Kurz *et al.*, 2009). Decomposition is simulated by applying decay rate that depends on the mean air temperature, the specific base decay rate at a reference temperature of 10 ◦C (Q10) (Kurz *et al.*, 2009). The mean air temperature of 0 ◦C and precipitations of 909 mm were used in our simulation (Joutel station, 1981-2010 period, Environment and Climate Change Canada, access 2021). Based on their decay rates, soil pools are classified as very fast to slow according to their position relative to the mineral soil (aboveground and belowground) and the type of biomass input (Figure 5). In order to populate the soil pools at the beginning of a model run, initialization is needed. The initialization procedure links biomass and soil dynamics by the repeated application of historical disturbance events until soil C pools stabilize (i.e., difference tolerance of 1.00% of slow soil C pools at the end of two successive rotations). It ends with the simulation of the last disturbance specified in the forest inventory (Kurz *et al.*, 2009). Parameters related to turnover, decomposition (base decay rate, Q10, etc.), C transfer, and fire disturbance matrices were modified based on published literature (appendix C, chapter 1). The base decay rates varied according to bryophyte type and remained constant throughout the simulation period for the fast *Sphagnum* pool, as well as the fast and slow Feathermoss pool. The base decay rate for the slow *Sphagnum* pool depends on the maximum merchantable volume (MVOL) of the stand, to include indirectly the low decomposition rate under the effect of poor drainage in low-productivity sites (Bona *et al.*, 2016).

Wildfire disturbance. CBM-CFS3 employs disturbance matrices, specific to each ecozone, to describe the proportions of C transferred between C pools, atmosphere, and forest products sector following disturbance events. Fire matrices result in the modeling of fuel consumption for each pool using national forest inventory, fire weather information, and the Boreal Fire Effects model (BORFIRE) (de Groot, 2006; Kurz *et al.*, 2009). Default fire disturbance matrices and parameters of the spatial unit of the eastern boreal shield were used and modified to include variation in fire severity (appendix B, chapter 1). We define fire severity as the effects of fire on trees and bryophyte living biomass as well as on soil pools. The proportion of burnt C for living biomass and organic soil layer pools were calibrated using values cited in Shetler *et al*. (2008), and other studies (Andrieux *et al.*, 2018; Lecomte *et al.*,2007) for black spruce forests dominated by *Sphagnum* or feather moss (appendix C, chapter 1). Low severity fires tended to have lower impact on biomass and partially consume the organic layer while high severity fires can almost completely consume the organic matter layer (Simard *et al*, 2007; Kasischke and Bruhwiler, 2002). For instance, high severity fires consumed more living biomass (100% feather moss and 90% *Sphagnum*) and organic soil layer (45% in average) than low fire severity with low biomass consumption (100% feather moss and 77% *Sphagnum*, 20% organic soil layer in average) (Benscoter and Vitt, 2008; Kasischke and Bruhwiler, 2002; Shetler *et al.*, 2008). Slow soil C pool is conserved during a wildfire, assumed to be wet or saturated (Bona *et al.*, 2016; Kurz *et al.*, 2009). For more details about the model, see Kurz et *al*. (2009) for CBM-CFS3 and Bona et *al*., 2016 for the moss module.

The following annual outputs can be used to evaluate simulation scenarios: Total ecosystem C Stocks, C Stock Changes (delta ecosystem), Emissions. Total ecosystem C storage was estimated as the sum of C in all the live biomass and soil pools (litter, debris, dead wood and organic (FH horizon) soil layer), mineral soil was excluded. We also focused on live biomass and organic soil layer pools. We presented all outputs in Mg C ha^{-1} (1 Mg C ha^{-1} = 1 t C ha^{-1} = 10^6 g C ha^{-1}) for stand level and Mt C (1 Mt C =10^6 t C = 10^6 Mg C = 10^{12} g C) for landscape level. A positive value for the annual change in C stock indicates a net gain in C stocks (‘‘sink’’), a negative value indicates a net loss (‘‘source’’) (IPCC, 2006; Kull *et al.*, 2014).

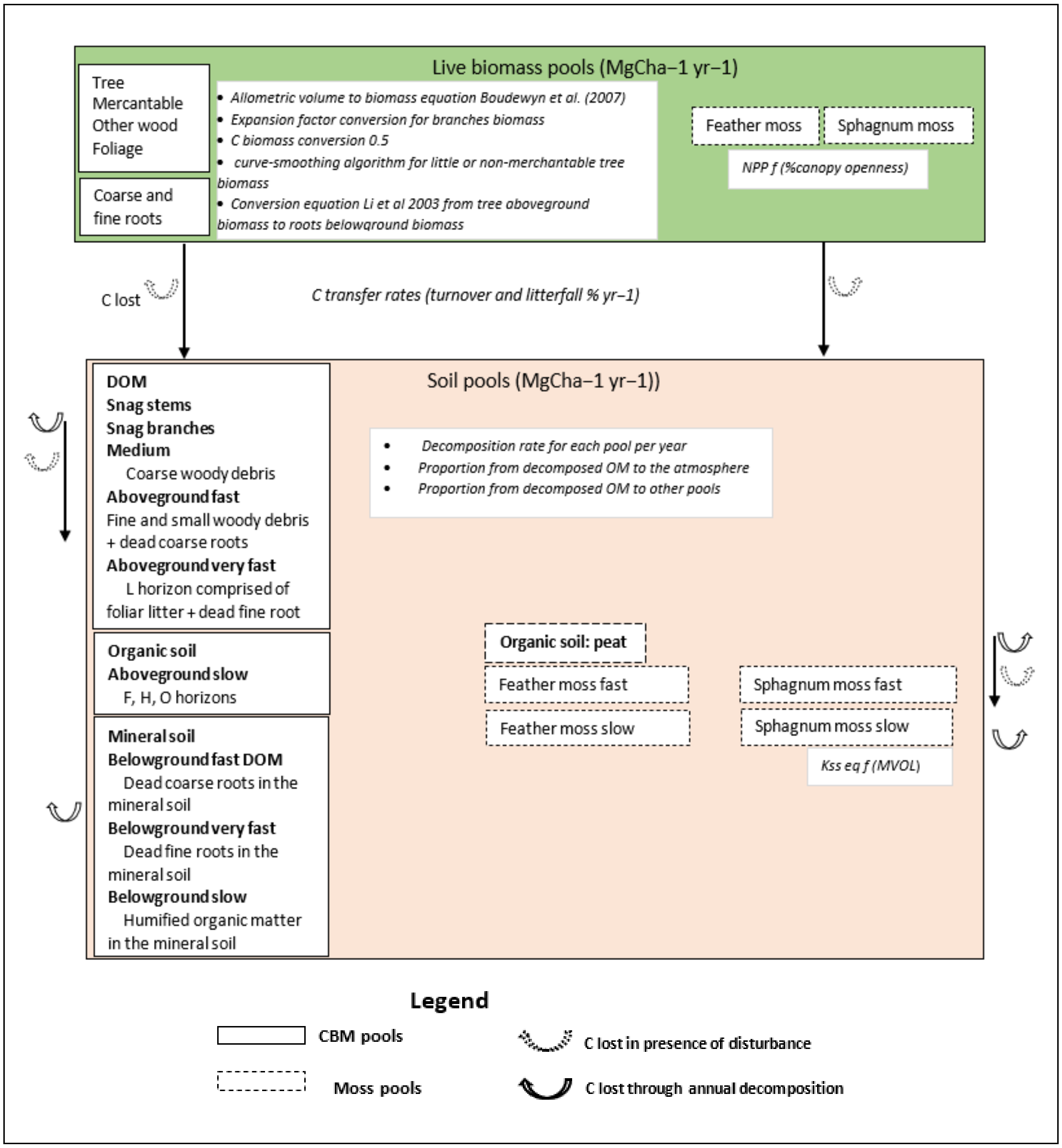


Figure 5
Diagram illustrating CBM-CFS3 and the moss module. CBM-CFS3 pools are represented with solid boxes, while moss module pools are represented with dashed boxes. Carbon transfers are represented with straight arrow and carbon lost to the atmosphere is represented with curved arrows, which can be solid to represent annual decomposition or dashed in the presence of disturbance. For default parameter values and more details on model equations, refer to Appendix C-chapter1, NPP Net primary productivity, Kss constant base decay for slow Sphagnum organic soil layer pool.

1.4.3 Data

Forest inventory data of FMU 085-51 were completed in 2010 (Québec government agency Ministère des Ressources naturelles et des Forêts, 2010), yield curves and regional ecological characteristics from this inventory were used to parameterize the CBM model. In this study, natural black spruce stands that were paludified or liable to paludification were selected in the forest inventory based on: fire origin, ecological forest type (black spruce-feathermoss and black spruce-*Sphagnum* dominated stand), forest density (high to low), drainage class (moderate to poor) and surface deposits (*e.g.*, organic, till, glaciolacustrine). A total of 32, 987 black spruce stands originated from fire representing an area of 181,500.6 haha were selected for the simulations (18% of FMU area). These selected stands presented various age classes but were dominated by the older age class of 120+ years (Figure 6). The uneven-aged stands, accounting for only 11% of the selected area, are included in the age class of 120+ years because the 120+ class often contains multicohort forests (Martin et al., 2020 and CBM does not explicitly consider multiple cohorts (Kull *et al.*, 2014; Kurz *et al.*, 2009).

Our simulation hypothesis was primarily based on the dominance of ground cover by bryophytes, which can reflect the site's quality (productivity and soil condition) (Tableau 1, Appendix F, chapter 1). We grouped the selected black spruce stands in 2 groups based on the initial site conditions in 2010 and to reflect the possible trajectories of stand during our simulation period: poorly drained black spruce-*Sphagnum* dominated stands (BSSP) and productive black spruce-feathermoss dominated stands (BSFM). The BSSP, prone to paludification, includes stands that transitioned from closed or partially closed to open and less productive with abundant *Sphagnum* species cover, as well as initially less productive stands with abundant *Sphagnum* species cover. In contrast, the BSFM, includes more productive stands that remained productive, *i.e.*, where feathermoss was not replaced by *Sphagnum* spp. over the simulation period (Belleau *et al.*, 2011; Kohlenberg, 2015; Shetler *et al.*, 2008; Simard *et al.*, 2007). In the forest inventory, BSSP stands tended to be less drainedcompared to BSFM stands (Tableau 1, Appendix F, chapter 1). BSSP forests represented 82% and BSFM forests 18% of selected stands (Tableau 1).

Tableau 1
Selected stands of FMU 8551 in the 2010 forest inventory data.

Forest group	Characteristics	Area (ha)
Black spruce-*Sphagnum* (BSSP)	dominant moss species: *Sphagnum* low forest density: dominant cover density between 40% and 60% poor drainage class	148738 (82% selected stands)
Black spruce-feathermoss (BSFM)	dominant moss species: feathermoss intermediate forest density: dominant cover density between 60% and 80% relatively well drained to moderate drainage class	32762,6 (18% of selected stands)

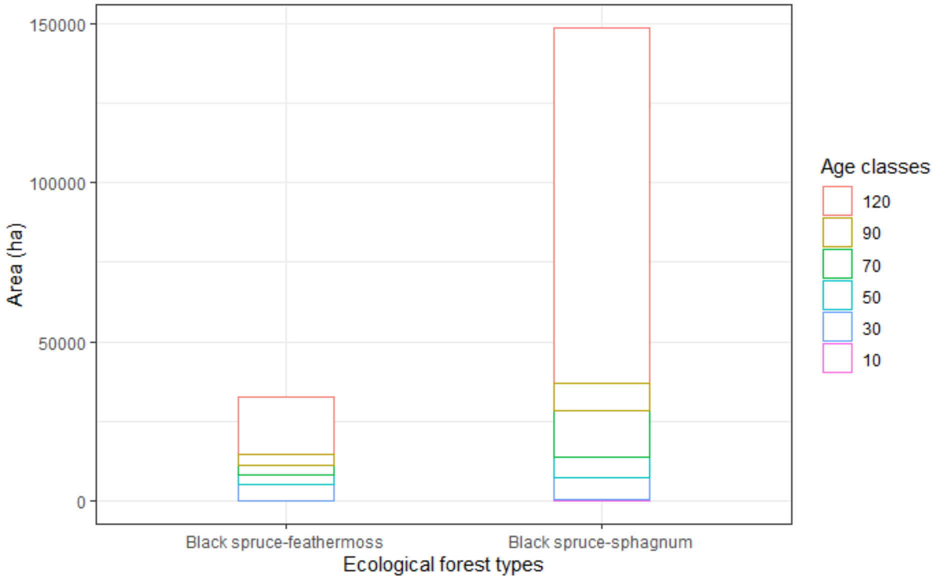


Figure 6
Landscape study characteristics: forest groups (Black spruce-feathermoss and Black spruce-Sphagnum), area and age classes of selected stands in forest inventory information 2010.

Different black spruce yield curves reflecting site quality were assigned to the different stand groups (BSSP vs. BSFM) (appendix G, chapter 1). These yield curves (volume-age) were provided by Québec government agency Bureau du Forestier en Chef (BFEC) and were made with forest growth simulation model NATURA (stand-level) and ARTEMIS (tree scale) (Auger, 2017). The yield curve is constructed from a regression done on the same permanent and ecologically similar plots of the black spruce-feather-moss bioclimatic subdomain. We adjusted and extrapolated the curves to align with our simulation duration of 400 years, considering that the curve's original duration was limited to 250 years (appendix G, chapter 1). The selection and analysis of yield curves were based on previous studies (Garet *et al.*, 2009; Simard *et al.*, 2007). We maintain the beginning of the curve until the maximum volume, and after the inflection point, apply a decreasing exponential function as described in Lecomte *et al.* (2006) (appendix G, chapter 1). Different reduction rates were applied to extrapolated yield curves to maintain the closed canopy in BSFM stands and to open canopy in BSSP stand by the end of the simulation period (400 years) (Andrieux *et al.*, 2018, Belleau *et al.*, 2012, Simard *et al.*, 2007).

1.4.4 Simulation: scenarios and assumptions

Wildfire scenarios based on fire severity and fire cycle, were used to compare C stocks and balance at the stand and landscape levels (Tableau 2). The simulations were conducted over a period of 400 years. Model initialization was performed with the historical fire cycle of 140 years (Bergeron *et al*, 2004).

At the stand level, we compared the effects of low severity fires (LSF) and high severity fires (HSF) for two main forest site types found in the landscape: black spruce-*Sphagnum* dominated stand (BSSP) and black spruce-feathermoss dominated stand (BSFM). Then four scenarios were simulated over 400 years (current fire cycle): 1-LSF occurring in BSSP stand, (2) HSF occurring in BSSP stand, (3) HSF in BSFM stand, and (4) LSF occurring in BSFM stand (Tableau 2). We began the simulation with initial stand ages of 120 years because it is the dominant age class in the landscape.

We used proportion C burned, tree growth, successional moss cover, and decomposition rate of organic soil layer to reflect difference between forest type and fire severity. Because *Sphagnum* species are more resistant to fire than feather mosses (Benscoter *et al.*, 2011; Benscoter and Wieder, 2003; Terrier *et al.*, 2014), we applied the higher organic C soil layer burn proportion observed in sites dominated by feathermoss BSFM as the HSF effect reference, and the lower C burn proportion as the LSF effect reference in sites dominated by *Sphagnum* BSSP. In each forest type, we applied a factor of approximately 1.71 (Lecomte *et al*, 2006) to the fire reference severity on the organic layer to determine the LSF or HSF, respectively, in BSFM and BSSP sites (appendix C, chapter 1). HSF was set to lead to a more productive forest with relatively low *Sphagnum* cover compared to LSF (appendix E, appendix G , chapter 1). According to fire severity, predictive relationships and different yield curves were used to reflect moss dynamic and tree growth reduction in BSSP stand and higher site productivity in BSFM over time (Andrieux *et al.*, 2018, Belleau *et al.*, 2012).

In BSSP, two successional trajectories were presented: a productive one after HSF, starting with young, closed stands that transitioned to open and less productive stands as feathermoss was gradually replaced by *Sphagnum* over the long term, and a second, less productive trajectory after LSF, beginning with young, open stands already abundant in *Sphagnum*. In BSFM stands less prone to paludification, we adjusted the moss ground cover relationship from BSSP stands, avoiding the gradual switch from feathermosses to *Sphagnum* species over time, to reflect a closed forest canopy with lower *Sphagnum* species cover (see Appendix E, chapter 1). Instead, we modified the cover after HSF and LSF to reach a maximum of 15% and 30% *Sphagnum* cover, respectively, according to published literature on productive black spruce-feathermoss dominated stands (Bona *et al.*,2016; Bond-Lamberty *et al.*, 2004, O'Neill *et al.*, 2006, Ward *et al.*, 2004). For the purpose of these model simulations, BSSP stands were initiated by LSF, while BSFM stands were initiated by HSF in relatively well-drained areas. Furthermore, because *Sphagnum* species decompose more slowly than feathermoss and modify site drainage (Benscoter and Wieder, 2003; Fenton *and al.*, 2010; Lang *and al.*, 2009), we used soil parameters (Q10, decay rate constant) based on published literature to reflect the thickness of organic soil layer in BSSP compared to BSFM stands (Appendix C, chapter 1).

At the landscape level, we investigated the effect of shortening to 100-year fire cycle based on prediction using climate scenario RCP 8.5 (Bergeron *et al.*, 2010; Boisvenue *et al.*, 2014; Boulanger *et al.*, 2013) compared to the long current 400-year fire cycle, including fire severity. To explore the effect of fire frequency and severity on forest C dynamic, the following scenarios were tested: (1) the baseline scenario was long fire cycle of 400 years (current cycle) with the occurrence of LSF, (2) long fire cycle of 400 years with occurrence of HSF , (3) the short 100-year fire cycle with occurrence of LSF, (4) the short 100-year fire cycle with occurrence of HSF (Tableau 2). The 400-year fire cycle was simulated by randomly burning on average 0.0025 of total landscape area per year, and the 100-year fire cycle scenario by burning 0.01 of the total landscape area per year (Boulanger *et al.*, 2014). A truncated log-normal distribution was used for the randomly selected annual burnt area.

At the initial simulation conditions (time step = 0), the landscape with different age classes (Figure 6), was characterized by 82% of BSSP more prone to paludification and 18% of BSFM (Tableau 1). In the 82% of the landscape, initiation occurred with low fire severity, while in the remaining 18% of the landscape, initiation occurred with high fire severity. Each scenario was tested on the same initial landscape condition. Different classes of productivity yield curves (ranging from low to high) have been applied in each forest group based on the site quality (appendix G, chapter 1). Furthermore, at the landscape level, we included the potential regeneration failure during the simulation period in the fire scenario. Regeneration failure can occur when a stand burned before reaching the maturity threshold age, resulting in a decrease in site productivity (Splawinski *et al*, 2019). Regeneration failure of black spruce occurs when stand age is less than 50 years old (Schab *et al.*, 2021; Splawinski *et al.*, 2019; Viglas *et al.*, 2013). We used the lower yield curve than the preharvest stand state to reflect the regeneration failure (Schab *et al.*, 2021).

Tableau 2
Scenarios at the stand and landscape levels.

Scenarios	Description
Stand-level	
Scenario 1	LSF occurring in BSSP stand, BSSP-LSF
Scenario 2	HSF occurring in BSSP stand, BSSP-HSF
Scenario 3	HSF in BSFM stand, BSFM-HSF
Scenario 4	LSF occurring in BSFM stand, BSFM-LSF
Landscape-level	
Scenario 1	400-year fire cycle (current cycle, 0.25% annual burn rate) with the occurrence of low fire severity LSF,
Scenario 2	400-year fire cycle (current cycle, 0.25% annual burn rate) with occurrence of high fire severity HSF
Scenario 3	100-year fire cycle (1% annual burn rate) with occurrence of with occurrence of low fire severity LSF
Scenario 4	100-year fire cycle (1% annual burn rate) with occurrence of high fire severity HSF

LSF: low severity fire

HSF: high severity fire

BSSP: black spruce-*Sphagnum* dominated stands

BSFM: black spruce-feathermoss dominated stands

1.5 Results

1.5.1 Stand level

At initial conditions (120-year-old stands), greater total C stocks were found in BSSP than BSFM stands, thanks to thick organic layer in BSSP and because BSSP were initiated under LSF while BSFM were initiated with HSF. BSSP had a greater proportion of C in the organic layer than in the tree biomass (77% versus 15% of total excluding the mineral soil) while BSFM had a greater proportion of C in tree biomass than organic layer (48% versus 32% of total excluding the mineral soil).

Immediately after disturbance, total C stocks decreased more after HSF than LSF (**Figure 7**,**Figure 8**a). Thereafter, following this decline they increased to higher levels after HSF than LSF due to the faster recovery of tree biomass with HSF (**Figure 8**b). As illustrated in Figure 8 (a, b), the difference between HSF and LSF and the amplitude of the changes (both the loss and the subsequent increase) are much greater for BSFM than for BSSP in the initial phase following fire (about 80 years). For BSSP, the advantage of HSF over LSF in total C socks started to diminish at age 100 as the organic layer develops faster under LSF (**Figure 8**c). From age 250 to the end of the simulation period, LSF contains more C than HSF for BSSP stands. In contrast, the difference in the C stock for HSF and LSF remains steady and positive for BSFM stands throughout the simulation period with the exception of the first 50 years (Figure 8a).

HSF generated a greater tree growth for both stand types and this difference, while it slowly declined, remained positive for the full simulation period (Figure 8b). The C stocks of the organic layer were always smaller with HSF. However, the temporal pattern of the difference between HSF and LSF varied with stand type (Figure 8c). It remained steady from year 80 to the end of the simulation period for BSFM while it steadily declined for BSSP as the organic layer continued to accumulate in the long term with LSF.

During the 400-year simulation period, based on final and initial C stocks, the organic layer accumulated on average nearly twice as much C with LSF than with HSF (11 g C m^{-2} $year^{-1}$ vs. 6.4 g C m^{-2} $year^{-1}$, respectively) in BSSP stands. In BSFM stands, C accumulation rates were less contrasted between fire intensity types with 4.7 g C m^{-2} $year^{-1}$ with LSF and 3 g C m^{-2} $year^{-1}$with HSF.

Living mosses made up a small portion of total biomass (3-5%) (appendix I, chapter 1). However, mosses accounted for 20% (mostly *Sphagnum*) on average in LSF and 17% (with 11% of sphagnum) in HSF of total forest net primary productivity (NPP) in a drier BSFM stand, whereas in a BSSP stand, they accounted for 9% (2.86% of *Sphagnum*) in HSF and 12 % (6.78% of *Sphagnum*) of NPP in LSF.

Immediately following fire, C losses and emissions were greater after HSF (-40 Mg C ha^{-1} year^{-1} in BSSP stand, -56 Mg C ha^{-1} year^{-1} in BSFM stand) than after LSF -25 Mg C ha^{-1} year^{-1} in BSSP stand, -34 Mg C ha^{-1} year^{-1} in BSFM stand). However, the stand rapidly became a greater sink with HSF due to the rapid post-fire tree growth. Thereafter, the C balance remained a weak sink in BSSP stands, particularly in LSF compared to HSF, due to the larger peat pool (organic layer), while it shifted to a net source in BSFM stand at age 250 (Tableau 3, appendix I, chapter 1).

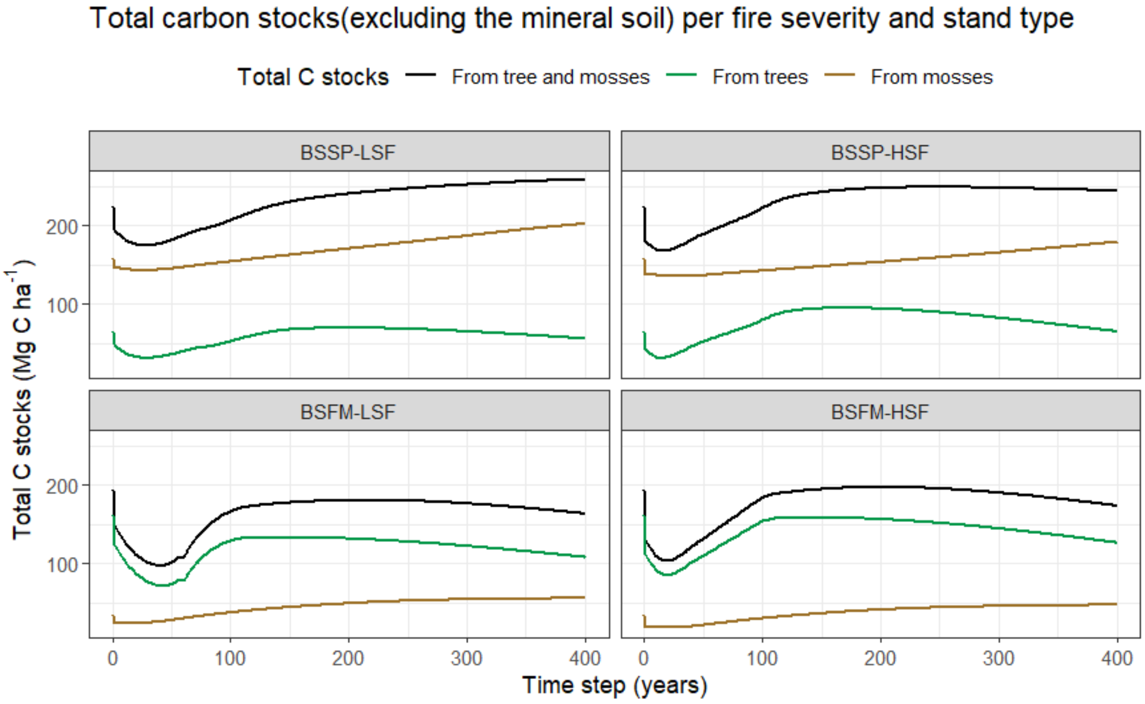
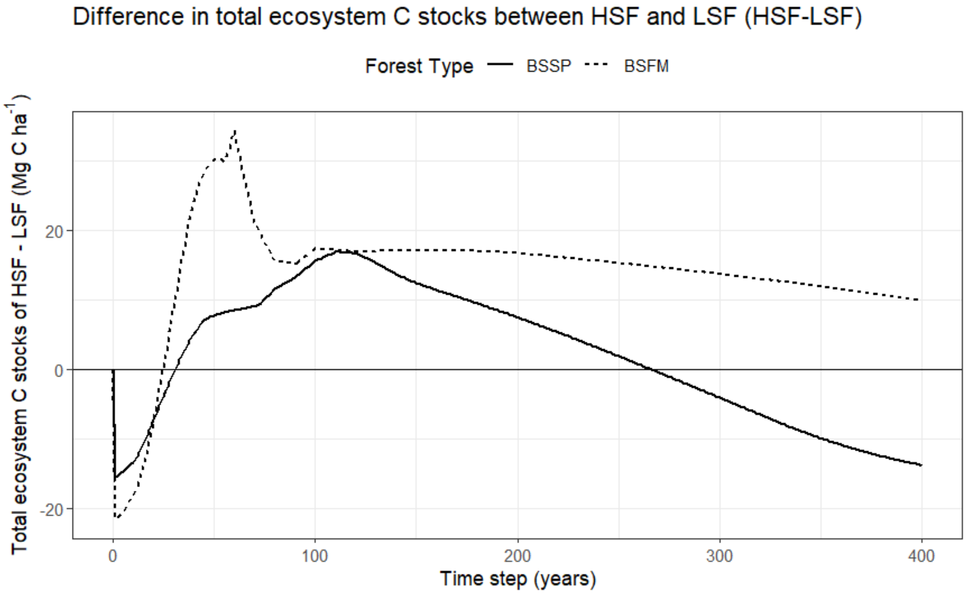
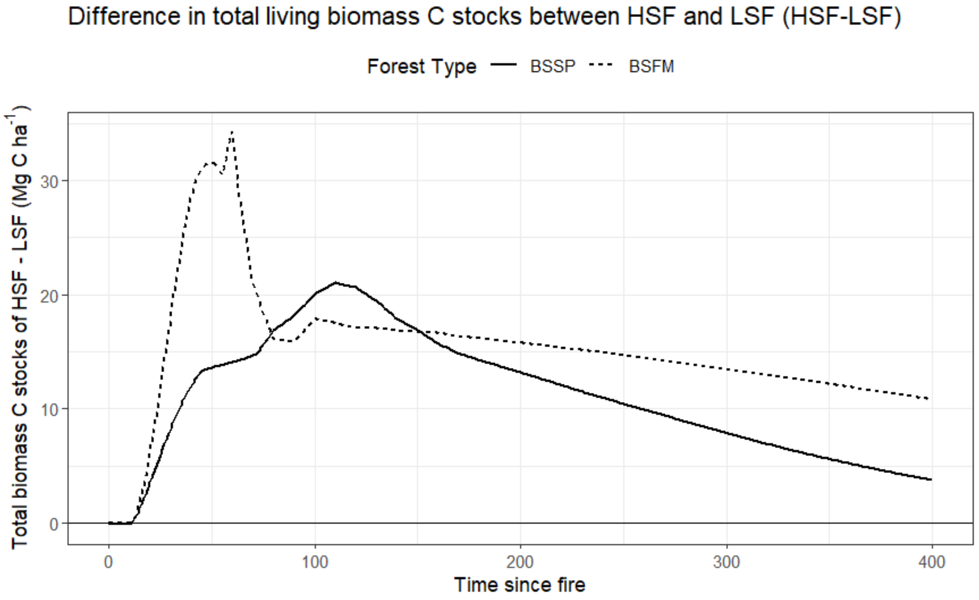


Figure 7
Stand level total ecosystem carbon stocks (Mg C ha^{-1} , live biomass + litter + debris+ dead wood + organic soil layer) evolution with time since disturbance for the two stand types (BSSP and BSFM) and the two fire severity classes (LSF and HSF). Total carbon stocks from trees in green, total carbon stocks from mosses (live biomass and moss-derived carbon in organic layer) in brown, and total carbon stocks from mosses and trees in black. LSF: low severity fire; HSF: high severity fire; BSSP: black spruce-Sphagnum dominated stands; BSFM: black spruce-feathermoss dominated stands.

a)


Difference in total ecosystem C stocks between HSF and LSF (HSF-LSF)
Forest Type
BSSP
BSFM
Total ecosystem C stocks of HSF - LSF (Mg C ha-1)
20
0
-20
0
100
200
300
400
Time step (years)

b)


Difference in total living biomass C stocks between HSF and LSF (HSF-LSF)
Forest Type
BSSP
BSFM
Total biomass C stocks of HSF - LSF (Mg C ha-1)
30
20
10
0
0
100
200
300
400
Time since fire

c)

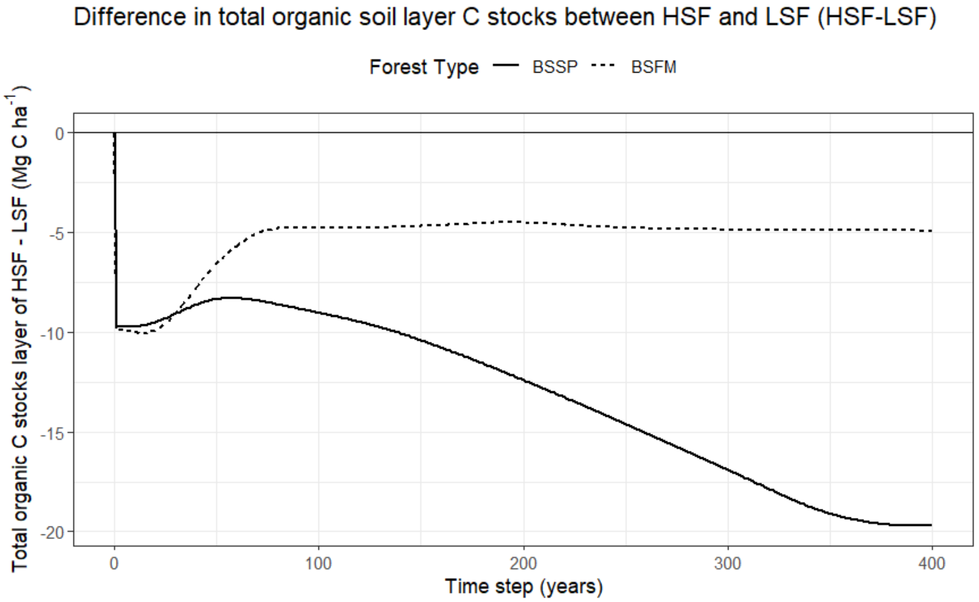


Figure 8
Difference (HSF - LSF, Mg C ha^{-1}) at the stand level, in carbon stocks in total ecosystem excluding the mineral soil (a, live biomass moss and tree + litter + debris+ dead wood + organic soil layer), in living biomass(b) and in organic soil layer (c) according to fire severity and forest type. Positive values indicate that forest regenerated following HSF contain more carbon than those issued from LSF, and negative values indicate the opposite. LSF: low severity fire; HSF: high severity fire; BSSP: black spruce-Sphagnum dominated stands; BSFM: black spruce-feathermoss dominated stands.

Tableau 3
Summary of total carbon balance (i.e. total C emissions/removals, Mg C ha^{-1} year-1 = 100g C m^{-2} y^{-1}) per disturbance scenario: carbon balance at different time steps (initial, at disturbance year, and at the end of simulation), delay to switch to a sink, maximum total carbon balance and stand age reached, and average of total balance as well as separate averages for all tree compartment and moss compartment (live biomass and moss-derived C) contributions for all simulation period.

	Scenario			
	BSSP		BSFM	
Values	HSF	LSF	HSF	LSF
C balance for the initial year of simulation (Mg C ha^{-1} year$^{-1)}$	0	0	0	0
C balance during the disturbance year (Mg C ha^{-1} year$^{-1)}$	-40.333	-24.906	-55.711	-34.115
First positive C balance value (sink, Mg C ha^{-1} year$^{-1)}$	0.0418	0.011	0.0412	0.030
Age of switch from source to sink (years)	17	30	21	42
Maximum C balance (Mg C ha^{-1} year$^{-1)}$	0.899	0.624	1.241	2.561
Age for the maximum C balance (years)	100	100	40	61
C balance for the final year of simulation (Mg C ha^{-1} year$^{-1)}$	-0.002	0.043	-0.181	-0.156
C balance average (Mg C ha^{-1} year$^{-1)}$	0.098	0.106	-0.038	-0.085
C balance average from tree (Mg C ha^{-1} year$^{-1)}$	0.0468	-0.003	-0.071	-0.139
C balance average from moss (Mg C ha^{-1} year$^{-1)}$	0.052	0.109	0.033	0.055

LSF: low severity fire

HSF: high severity fire

BSSP: black spruce-*Sphagnum* dominated stands

BSFM: black spruce-feathermoss dominated stands

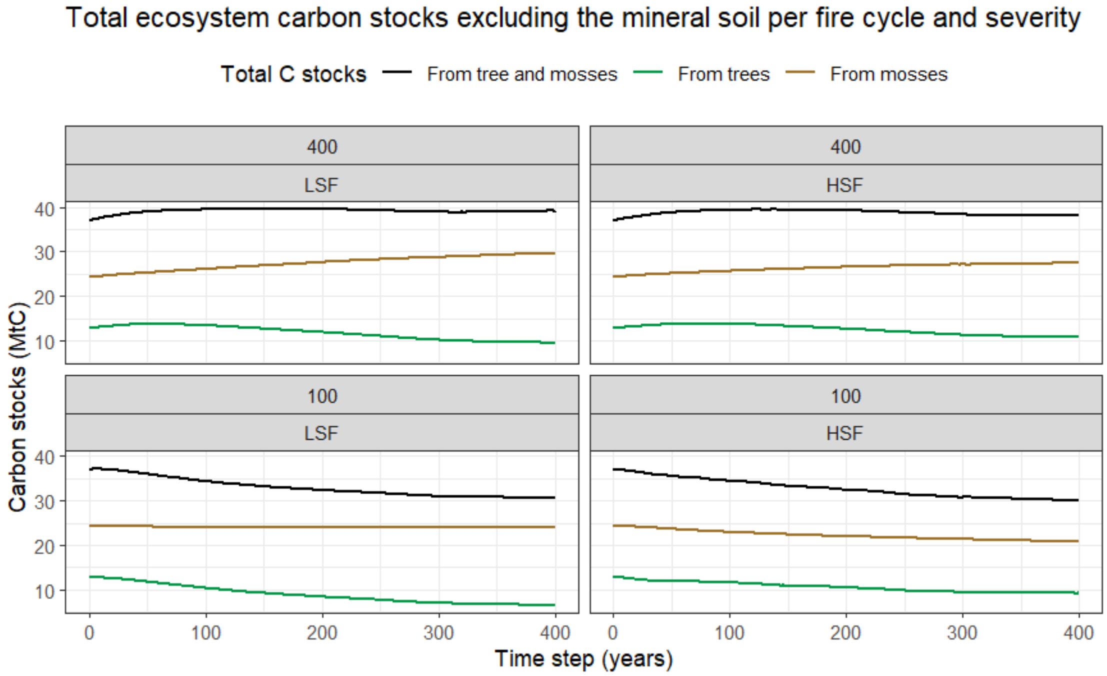
1.5.2 Landscape level

As expected, total C stocks were higher with a long fire cycle (400 years) than with a short fire cycle (100 years) (Figure 9a). They varied from 37.03 MtC (initial landscape C stock) to 39 M tC ha−1 with LSF and to 38 MtC with HSF, while under a short fire cycle their total C stock decreased from 37.03 MtC to 31 MtC in LSF and 30 MtC in HSF (Figure 9a). LSF and HSF seemed to exhibit similar total C stocks trends with both fire cycle lengths. Total C stocks remained stable in time for the 400-year cycle while they declined in the shorter cycle (Figure 9 a,Tableau 4).

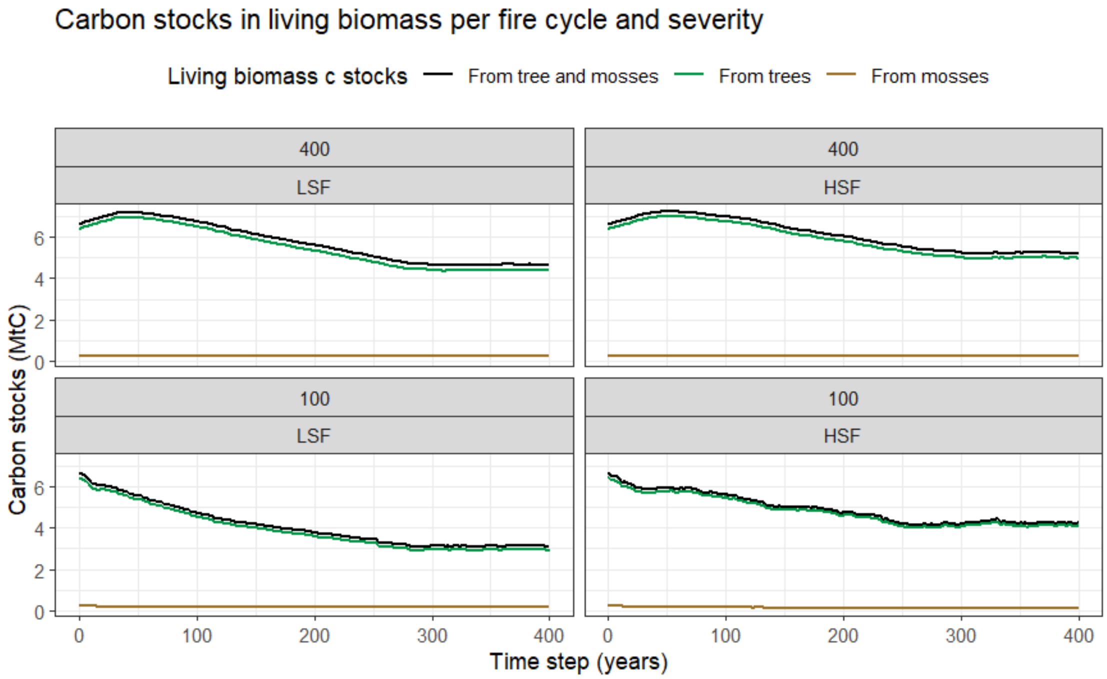
C balance and emissions. Forests remained a C sink along the 400-year fire cycle scenario while they were a net source in the 100-year fire cycle (Figure 10). In the 400-year fire cycle scenario, the landscape was a strong sink for the first 200 years, then it transitioned to an almost neutral C balance for the remaining period (Figure 10). LSF in the long fire cycle, led to a net C sink for the reminder of the simulation period due to the role of mosses. At the beginning of the simulation, in the short 100-year fire cycle scenario, the landscape switched from a sink to a source with higher C emissions, especially in the HSF scenario (Figure 10, Tableau 5). The 400-year cycle was the greatest gross emitter of C to the atmosphere in both the HSF (0.397± 0.027 MtC year^{-1}) and LSF (0.376 ± 0.028 MtC year^{-1}) scenarios, compared to the short fire cycle (0.359 ± 0.043 MtC year^{-1} for HSF0.299 ±0.052 MtC year^{-1} for LSF).

Age class distribution. As expected, at the initial conditions, class distribution was dominated by the old age class (120+ years). Over the simulation period, the shorter fire cycle resulted in a higher proportion of younger age-class (<100 years) distribution compared to the longer fire cycle after 400 years of simulation (appendix I, chapter 1).

a)


Total ecosystem carbon stocks excluding the mineral soil per fire cycle and severity
Total C stocks
From tree and mosses
From trees
From mosses
400
LSF
400
HSF
100
LSF
100
HSF
Carbon stocks (MtC)
10
20
30
40
0
100
200
300
400
Time step (years)

b)


Carbon stocks in living biomass per fire cycle and severity
Living biomass c stocks
From tree and mosses
From trees
From mosses
400
LSF
400
HSF
100
LSF
100
HSF
Carbon stocks (MtC)
0
2
4
6
0
100
200
300
400
Time step (years)

c)

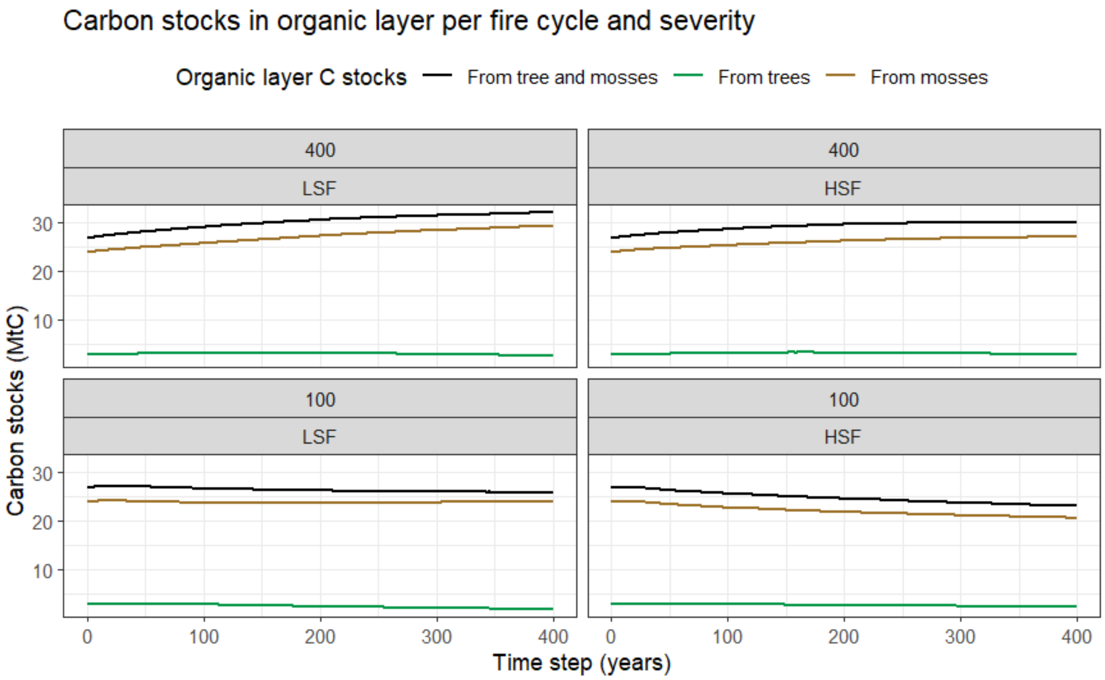


Figure 9
Landscape (181,500.6 ha) simulated carbon stocks dynamic (MtC; 10^{12} gC or 10^6 tC, without mineral soil) for 400-year and 100-year fire cycles and the two fire severity classes (LSF and HSF): a) Total ecosystem C stocks excluding the mineral soil (live biomass + litter + debris+ dead wood + organic soil layer), b) living biomass C stocks (from trees, from mosses) and c) organic layer (moss-derived and from trees). C stocks from in green, C stocks from moss in brown, and total (moss + tree) in black. LSF: low severity fire; HSF: high severity fire.

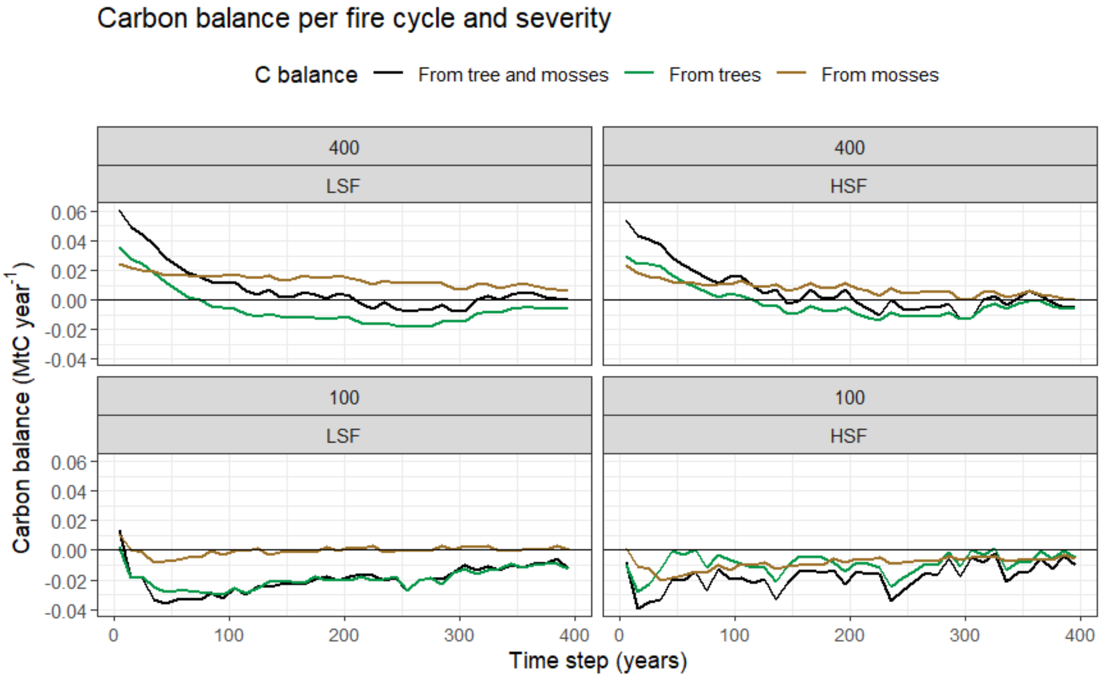


Figure 10
The landscape carbon balance (MtC year-1; 1012 gC or 106 tC by year): tree contribution in green, moss in brown, and combined moss and tree in black. The annual carbon balance values were smoothed averages over 10 years. LSF: low severity fire; HSF: high severity fire.

Tableau 4
Differences in landscape carbon stocks (total, tree and moss) between the last year of simulation and the first year of simulation per disturbance scenario. Ratio in percent of the difference between final and initial values relative to the initial value.

	Scenario			
	400 fire cycle		100 fire cycle	
	HSF	LSF	HSF	LSF
Total	2.97 %	5.78 %	-18.9 %	-17.10 %
Total Tree	-16.02 %	-25.53%	-27.65%	-48.83%
Total Moss	12.96 %	22.26 %	-14.30 %	-0.40 %

LSF: low severity fire

HSF: high severity fire

Tableau 5
Average of carbon balance (MtC by year; 1012 gC or 106 tC by year) per fire cycle and severity at the landscape level, according to tree and moss contribution, SD standard deviation.

Scenarios		**Tree and moss**	**SD**	**Tree**	**SD**	**Moss**	**SD**
400	LSF	0.008	0.017	-0.006	0.013	0.013	0.005
400	HSF	0.007	0.018	-0.001	0.011	0.008	0.007
100	LSF	-0.019	0.016	-0.019	0.010	-0.0002	0.009
100	HSF	-0.018	0.024	-0.009	0.015	-0.009	0.011

1.6 Discussion

1.6.1 Impact of fire severity at the stand level

In BSSP stands, we found that following wildfire, total C stocks eventually increased to higher levels with LSF compared to HSF due to a continuous accumulation of peat. In contrast, it declined slightly in BSFM stands in the long term, and less in HSF than LSF (**Figure 7**) due to a better tree growth in the former.

Low severity fires conserved *Sphagnum* moss and enhanced its productivity in BSSP while HSF stimulated tree growth in BSFM. These results demonstrated that the accumulation of peat is the main C reservoir in BSSP stands, while the living tree biomass is the dominant one in BSFM stands. Organic soil layer derived from mosses represents 80% of the total stocks in average in BSSP. These results indicate that the abundance of *Sphagnum* is key to postfire C recovery (Shetler *et al*, 2008) and points out the role of fire severity in BSSP stands (Lecomte *et al*, 2015, Shetler *et al*, 2008). According to Shetler *et al.* (2008), the abundance of unburned *Sphagnum* explained 50-60% of the variability in postfire organic soil layer depths in lowland stands but they found no relationships in upland stands.

C dynamics in aerial biomass matched the general pattern of tree growth for young forests, displaying rapid growth until a certain age, and then gradually decreasing in productivity. In BSSP stands, a curve with lower tree biomass after LSF compared to HSF was chosen to reflect the well-known negative effect of the thickness of the organic layer on tree growth (Lecomte *et al.*, 2006; Simard *et al.*, 2007). The partial consumption of organic layer during LSF and its slow decomposition rate leads to nutrient immobilization and reduction in tree growth (Fenton and Bergeron, 2006; Lecomte *et al.*, 2006; Simard *et al.*, 2007).

The faster sink recovery in HSF (17– 21 years) than LSF (30-42 years) is due to higher stand productivity both in BSSP and BSFM (Tableau 3). Our simulations showcased an important role of tree growth in the recovery of C stocks in forested peatlands. Despite higher moss abundance, the time taken to switch to a sink after LSF remained longer (30-42 years) here than in empirical studies. Wieder *et al* (2009) showed that bogs of central Alberta quickly become net C sink around 13 years after fire, however fire severity was not measured in this study. A general average of 20 years has been mentioned by Zoltai *et al.* (1998) for boreal northern peatlands. This difference is mainly due to the rapid recovery of the moss and shrub layers in some study areas (Wieder *et al.*, 2009). The understory vegetation (composition and abundance) can vary a lot, and it was not taken into account in our study. However, the moss dynamic equation used in our study, in relation to fire severity, was based on field data from the study area (Fenton and Bergeron, 2011; Fenton *et al.*, 2005).

Our results suggest that the impacts of wildfires between high severity (HSF) and low severity (LSF) depends on initial stand characteristics (BSSP and BSFM), C distribution, C accumulation, and the contribution of dead moss over time. Overall, our results are within the ranges reported by other studies of boreal black spruce peatland forests (Tableau 6). However, wide range of results on soil C stocks and accumulation rates were found in the literature due to differences in degrees of paludification, forest composition (conifer, broadleaf or mixedwood stands), soil sampling methods, and the inclusion of different soil pools. Regarding the organic layer, the accumulation rates of 11 g C m^{-2} $year^{-1}$ and 6.4 g C m^{-2} $year^{-1}$ simulated over 400-year under LSF and HSF scenarios respectively for BSSP stands, was within the range of 6 to 18 g C m^{-2} $year^{-1}$ found in Beaulne *et al.* (2021) and the range of 9 to 22 g C m^{-2} $year^{-1}$ found in Magnan *et al.,* (2020), both studies being in the Clay Belt region. C accumulation in the organic layer can indeed be higher in poorly drained black spruce-*Sphagnum* dominated stands than well-drained black spruce-feathermoss dominated stands (Beaulne *and al.*, 2021; Magnan *et al.*, 2020).

Tableau 6
Comparison of values of carbon fluxes and stocks from this study and the literature on Boreal Black Spruce forests.

C fluxes and stocks	This study	Literature	References
Living biomass total stocks (Mg C ha^{-1})	At 150 years after fire	At 151 years after fire	Wang *et al.*, 2003, included bryophytes and shrub
	BSSP stand: 37.2 for LSF 52.1 for HSF	37.4 for black spruce wet stands	
	BSFM stand: 75 LSF 84 HSF	For black spruce dry stands 83.3	
Living biomass black spruce stocks (Mg C ha^{-1})	BSSP stand: Average of 26.4 in LSF Average of 37.9 in HSF	15-53 with an average of 34; 28 – 57 with an average of 42.5	Magnan *et al.*, 2021 ; Beaulne *et al.*, 2021
	BSFM stand: Average of 52.4 in LSF Average of 72.6 in HSF	Average 42 black spruce mesic stand	Andrieux *et al* (2018)
Living biomass moss stocks (Mg C ha^{-1})	BSSP stand: 2 for LSF 1.76 for HSF at the end of simulation	2 for bog forested	Zoltai *et al.*, 1998 in bog forested in Southern Manitoba
	BSFM stand: 1.5 LSF 1 HSF at the end of simulation		

Tableau 6 (suite)
Comparison of values of carbon fluxes and stocks from this study and the literature on Boreal Black Spruce forests.

Organic layer C stocks /peat (Mg C ha^{-1})	BSSP stand: 172 – 220 from 0 to 400 years with average of 186 LSF, 173 HSF	90 for max peat thickness of 40 cm 400 year after fire; 620 -1 524 for peat thickness varied from 78 to 394 cm, and about 130 for 30 cm max of peat thickness; 226-660 thickness from 40 to 100 cm; 240 in average for peat thickness from 20 to 90 cm.	Lecomte *et al.*, 2006 intermediate sample soil; Magnan et al, 2020 total soil organic sample method; Beaulne et al, 2021; Sousa et al, 2021 in Alaska Botroh *et al.*, 2023, in clay belt (appendix C, chapter 1)
	BSFM stand: 62 -81 from 0 to 400 years with average 70.52 LSF and 65.34271 HSF,	70 - 80-85 Mg C /ha from 2 to 312 years, mesic site	Andrieux *et al* (2018)
Organic layer or peat C accumulation rate (g C m^{-2} year^{-1})	BSSP stand: Long term average rate 11 in LSF and 6 in HSF Post disturbance rate 14 in LSF and 12 in HSF	9.3 to 22.8 (15.9 in average) long term accumulation carbon (LORCA); 6 to 18; 20 to 50; 57	Magnan *et al*, 2020; Beaulne *et al*, 2021; Payette *et al*, 2001 global average northern boreal peatland, Lecomte *et al*, 2006 in clay belt over 700 years, chronosequence.

Tableau 6 (suite)
Comparison of values of carbon fluxes and stocks from this study and the literature on Boreal Black Spruce forests.

	BSFM stand: Long term average rate 4.7 in LSF and 3 in HSF Post disturbance rate 0.07 in LSF and 0.82 in HSF	9.2 post disturbance rate; 2	Andrieux *et al*., 2018, Well drained site; Gao *et al*., 2018 in 210 years chronosequence boreal forest in Ontario, Canada.
Total stocks with mineral soil (Mg C ha^{-1})	BSSP stand: 331.8 - 374.27 for LSF 331.8 - 371.6 for HSF, from 0 to 400 years	430 to 500	Wang *et al*., 2003
	BSFM stand: 390.47 - 356.51 for LSF 390.47 - 375.37 for HSF,	83 to 356 from 2 years and 283 years after fire	Andrieux *et al* (2018)
Sink reached delay (years)	BSSP stand: 17 HSF; 30 LSF	20; 13	Zoltai *et al*., 1998, Wieder *et al*., 2009 for the boreal peatlands
	BSFM stand: 21 HSF; 42 LSF	20	Kurz *et al*., 2009 for the boreal forest generally.

LSF: low severity fire

HSF: high severity fire

BSSP: black spruce-*Sphagnum* dominated stands

BSFM: black spruce-feathermoss dominated stands

1.6.2 Impact of fire frequency and severity at the landscape level

At the landscape scale, because our region is dominated by sites more prone to paludification (82% of BSSP stands and 18% BSFM), simulations results showed a net C accumulation over the long 400-fire cycle in both fire severity classes. Over the 400 years of simulations, C storage increased more in LSF (5.78%) than in HSF (2.97%) in the long fire cycle because moss is less altered in LSF and contributes greatly to C sequestration (Figure 9a, Tableau 4). In contrast, the short fire cycle (100-year), projected under the RCP 8.5 climate change scenario, showed a reduction of C stocks from 17–19% respectively in LSF and HSF. This reduction has been observed in other modeling studies (Boisvenue *and al.*, 2014; Kasischke *and al.*, 1995; Miquelajauregui *and al.*, 2018). The simulation results of the predicted shortening of fire cycle by Miquelajauregui *et al.* (2018), showed a reduction of 2 to 27% in C storage in the boreal black spruce forests bioclimatic domain of Quebec, Canada. Boisvenue *et al.* (2014), also showed a reduction of 13% in total C storage under 100-years fire cycle compared to 400-year fire cycle in the same study area as our study, FMU - 8551. The low C storage in the short 100-year fire cycle may be attributed to the increasing proportion of young stands compared to 400-year fire cycle which retains more old stands in the landscape. The more frequent fires resulted in young stands with low C input (tree and moss) being unable to accumulate C quickly enough to compensate for the C losses (direct fire emissions, vegetation mortality and decomposition) (Boisvenue *et al*, 2014). Over time, with 100 years cycle this reduction in the annual capacity of C production (net primary productivity) and the increase in emissions switched the landscape to a net source.

Furthermore, a shorter fire cycle of 100 years, is expected to significantly impact ecosystem C dynamics due to the occurrence of high severity fires (Boisvenue *et al.*, 2015, Gauthier *et al*, 2015). In our simulations, despite the small difference found between the HSF and LSF scenarios for total C stocks in 100-year fire cycle, fire severity changed the C partition between trees, mosses, and organic layer (Figure 9 b, c). HSF under the 100-year fire cycle had the lowest organic layer C stocks of all scenarios, suggesting a reduction in *Sphagnum* abundance. The high frequency of severe fires is known to alter forest soil C pools (Groot *et al.*, 2003). The high frequency of severe fires could increase soil organic matter decomposition, shift bryophytes composition (*i.e.*, *Sphagnum* versus feathermoss) (Boiffin, 2014; Miquelajauregui *et* *al.*, 2018). However, Terrier *et al.* (2014) suggested that, in the Clay Belt, spruce–*Sphagnum* stands will persist under climate change and will be more resistant to an increase in fire severity than spruce–feathermoss stands due to the moist and cool conditions of the organic layer. Simulation could be extended to others climate change conditions in future studies to improve climate change effect on landscape level, as the impact of fire severity on C partitioning could actually result in more notable differences in total landscape C stocks.

1.6.3 Model limitations and perspectives

This study explores the post-fire C dynamics in paludified or boreal forests liable-to-paludification. Modeling studies rarely consider forested peatlands which have active forestry activities. Bona *et al.* (2016) developed a moss module for upland forests, which improved the accuracy of CBM predictions of soil C stocks by five-fold for these forests which hold smaller peat-C stocks compared to forested peatlands. Here, we adapted the relationship between moss and tree dynamics to the forested peatlands of the Clay Belt region of eastern Canada. The results obtained were close to literature values, demonstrating the adequacy of the model to predict C stocks in this context. However, peatland forests are not homogeneous and the model simulations in this study had their limitations. For instance, the full range of variation in the degree of paludification (peat thickness) and vegetation composition of forested peatlands were not completely represented, mainly at the early stages of forest succession where other tree species than black spruce could be present. Also, bryophyte species diversity and productivity could be higher than what was represented here, where an average of NPP for *Sphagnum* species and feathermoss was used. Moreover, we did not account for the possibility that, over the long term, BSFM stands could transition to paludified stands or that some *Sphagnum*-dominated stands could remain productive. Finally, water table fluctuation, that can improve the prediction of peatland C emission by decomposition, is not included in our study. Taking into consideration the variability of these conditions in future investigations could help improving simulation results. Nevertheless, our study stressed the importance of considering forest types when predicting the effect of wildfire properties (frequency and severity) on C dynamics. Including initial stand characteristics (BSSP and BSFM), C distribution, and the contribution of dead moss over time can significantly enhance our understanding of post disturbance forest C sequestration potential for climate change mitigation and forest management strategies in paludified areas.

1.7 Acknowledgements

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2. NINE-YEARS EFFECT OF HARVESTING AND MECHANICAL SITE PREPARATION ON BRYOPHYTE DECOMPOSITION AND CARBON STOCKS IN A BOREAL FORESTED PEATLAND

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2.1 Abstract

The boreal forest holds the world's largest soil carbon (C) reservoir. A large portion of it is contained in a thick organic layer originating from the slow decay of bryophytes. Because a thick organic layer slows down tree growth, reduces forest productivity, and thereby reduces the potential wood supply, silvicultural treatments that aim to maintain or restore forest productivity after harvesting often involve mechanical site preparation. However, while these treatments can increase growth and C storage in trees, they can also lead to accelerated decomposition of the soil organic matter, reducing C storage. In this study, we assessed the nine-years effect of two silvicultural treatments on soil C dynamics in forested peatlands of northwestern Quebec, compared to unharvested controls: (1) cut with protection of regeneration and soils (CPRS; low soil disturbance), (2) CPRS followed by mechanical site preparation (CPRS+MSP, plowing; severe soil disturbance). The mass loss rate of three bryophytes (*Pleurozium schreberi, Sphagnum capillifolium* and *Sphagnum fuscum*) was measured over two growing seasons together with soil organic C (SOC) stocks. We observed a significant influence of silvicultural treatments, bryophyte species, and soil layer type (fibric, mesic, humic and mineral) on bryophyte mass loss, which was higher in the CPRS+MSP treatment (21.6 ± 0.13 % standard error) than in control sites (9.5 ± 0.21 %); CPRS alone resulted in an intermediate mass loss of 11.6 ± 0.23 %, for Sphagnum mosses. Bryophyte mass loss was significantly higher in fibric than humic layer. SOC stocks in the uppermost organic soil layer (fibric) were lower in the CPRS+MSP group than in the control group, while the CPRS group was intermediate; however, differences were not statistically significant for the other soil layer and for total SOC. We conclude that while CPRS+MSP accelerates Sphagnum moss decomposition in the topsoil layer, it has limited impact on total soil C stocks that are detectable with stock change methods.

Keywords: carbon, management, paludified boreal forest, mosses, site preparation, harvesting, heterotrophic respiration, soil layer.

2.2 Résumé

La forêt boréale détient le plus grand réservoir mondial de carbone (C) du sol. Une grande partie de celui-ci est contenue dans une épaisse couche organique provenant de la lente décomposition des bryophytes. Étant donné qu'une épaisse couche organique ralentit la croissance des arbres, réduit la productivité de la forêt et, par conséquent, l'approvisionnement potentiel en bois, les traitements sylvicoles qui visent à maintenir ou à rétablir la productivité de la forêt après la récolte impliquent souvent une préparation mécanique du site. Cependant, si ces traitements peuvent augmenter la croissance et le stockage du C dans les arbres, ils peuvent également entraîner une décomposition accélérée de la matière organique du sol, réduisant ainsi le stockage du C. Dans cette étude, nous avons évalué l'effet sur neuf ans de deux traitements sylvicoles sur la dynamique du C du sol dans des forêts entourbées du nord-ouest du Québec, en comparaison avec des témoins non récoltés : (1) coupe avec protection de la régénération et des sols (CPRS ; faible perturbation du sol), (2) CPRS suivie d'une préparation mécanique du site (CPRS+PMS, herse ; perturbation sévère du sol). Le taux de perte de masse de trois espèces de bryophytes (*Pleurozium schreberi*, *Sphagnum capillifolium*, et *Sphagnum fuscum*) a été mesuré pendant deux saisons de croissance ainsi que les stocks de carbone organique du sol (COS). Nous avons observé une influence significative des traitements sylvicoles, des espèces de bryophytes et du type de couche de sol (fibrique, mésique, humique et minéral) sur la perte de masse des bryophytes, qui était plus élevée dans le traitement CPRS+MSP (21,6 ± 0,13 % d'erreur type) que dans les sites témoins (9,5 ± 0,21 %); la CPRS a entraîné une perte de masse intermédiaire de 11,6 ± 0,23 %, pour les sphaignes. La perte de masse des bryophytes était plus élevée dans la couche fibrique que dans l'humique. Les stocks de COS dans la couche supérieure du sol organique (fibrique) étaient plus faibles dans le groupe CPRS+PMS que dans le groupe témoin, tandis que le groupe des CPRS était intermédiaire ; toutefois, les différences n'étaient pas statistiquement significatives pour les autres couches de sol et pour le COS total. Nous concluons que si le groupe CPRS+ PMS accélère la décomposition des sphaignes dans la couche supérieure du sol, il a un impact limité sur les stocks totaux de C du sol qui peuvent être détectés par les méthodes de changement de stocks.

Mots clés : carbone, gestion, forêt boréale paludifiée, mousses, préparation du site, récolte, respiration hétérotrophe, couche de sol.

2.3 Introduction

The boreal forest holds almost 60 % of the world's soil organic carbon (C) stock and plays an important role in global climate regulation (Ashton *et al.*, 2012; Dixon, 1994; Pan *et al.*, 2011). The large size of this reservoir is largely due to the slow decomposition rate of plant organic matter (OM) that is prevalent in this cold environment (Clymo *et al.*, 1998; Nilsson *et al.*, 2008). On poorly drained sites of the boreal forest region, the progressive SOM accumulation, originating primarily from bryophytes, can lead to the formation of a peat layer, which can reach several meters thick (Lecomte *et al.*, 2006; Sousa *et al.*, 2021). This peat accumulation may result in the formation of forested peatlands through the paludification process (Charman, 2002; Payette and Rochefort, 2001). Two types of paludification can occur, edaphic paludification initiated in wet topographic depressions, and successional paludification initiated on relatively well-drained soils, which is related to forest succession, and particularly to the influence of black spruce (*Picea mariana* [Mill.] BSP) (Fenton *et al.*, 2005b; Simard *et al.*, 2007).

In the long term, paludification slows tree growth and causes a decline in forest productivity and has been documented in regions around the boreal such as Alaska, Minnesota, Sweden, Finland, and Canada (Hånell, 1991; Lavoie *et al.*, 2005; Nuutinen *et al.*, 2000; Simard *et al.*, 2007). For instance, in the Clay Belt of eastern Canada , paludification results in a 50–80 % reduction in black spruce productivity at the landscape scale (Simard *et al.*, 2007). In fact, the presence of peat modifies soil conditions (e.g., low soil temperature, high water table level and soil moisture), and reduces OM decomposition and nutrient availability for trees (Lafleur *et al.*, 2011a; Payette and Rochefort, 2001; Thiffault *et al.*, 2013). Consequently, it is a poor growing medium, and with competition from peat-forming plants (e.g., Sphagnum species and ericaceous shrubs), seedling and tree development is limited (Lafleur *et al.*, 2011a; Lavoie *et al.*, 2006a). Despite the fact that edaphic and successional paludification may be confounded at the landscape scale, successional paludification can be reversible either by severe fires consuming a significant proportion of the peat layer, or by intensive mechanical soil disturbance both of which stimulate decomposition. In contrast, edaphic paludification is very resistant to change (Laamrani *et al.*, 2014; Lavoie *et al.*, 2005; Lecomte *et al.*, 2006).

Decomposition of soil OM is controlled by various factors that interact together: substrate type, and environmental conditions such as soil temperature, water content, pH, and soil aeration (Cornwell *et al.*, 2008; Fenton *et al.*, 2010; Preston *et al.*, 2012; Scanlon and Moore, 2000). *Sphagnum spp* L. decompose more slowly than other bryophyte such as feathermosses (Bonan and Shugart, 1989; Charman, 2002; Lang *et al.*, 2009). In Sphagnum dominated stands, low soil temperatures and high soil moisture slow down microbial activity and consequently OM decomposition (Payette and Rochefort, 2001; Shetler *et al.*, 2008). Moreover, decomposition rate decreases with depth in the peat layer and is particularly low below the water table level, in the catotelm (*i.e.*, the anaerobic zone with low microbial activity) (Belyea, 1996; Charman, 2002; Wieder and Vitt, 2006). These factors controlling soil OM decomposition can be modified by silvicultural treatments (Lavoie *et al.*, 2005; Mayer *et al.*, 2020; Payette and Rochefort, 2001). For instance, clearcutting (e.g., cut with protection of regeneration and soils (CPRS)) and partial cutting (CP) may reduce the forest cover and change the community of Sphagnum species present, thus changing substrate type, soil microclimate, and C accumulation rates (Fenton and Bergeron 2007; Halpern *et al.*, 2005). However, these effects are generally less pronounced after CP than after CPRS because of the low levels of soil disturbance (e.g., less area covered by machinery tracks and less canopy removal) (Fenton and Bergeron 2007; Halpern *et al.*, 2005). Removing the forest cover temporarily increases soil temperature and modifies soil moisture levels, therefore potentially influencing the soil OM decomposition (Hribljan 2012; Mayer *et al.*, 2020; Payette and Rochefort, 2001). However, silvicultural treatments such as CPRS, or winter harvesting have a low impact on the soil and the peat layer and can accelerate paludification by increasing light and water availability to Sphagnum for growth (Fenton *et al.*, 2010; Lafleur *et al.*, 2010). Therefore, intensive silvicultural treatments that severely disturb the soil, expose deeper soil layers and reduce peat thickness, such as mechanical site preparation (MSP, plow or disc trenching) or prescribed fire, are recommended following CPRS to foster organic layer decomposition, nutrient availability, and tree growth (Henneb *et al.*, 2019; Lafleur *et al.*, 2011a, 2011b; Renard *et al.*, 2016; Thiffault *et al.*, 2013). While intensive silvicultural treatments can increase growth and C storage in trees, they could also lead to C emission by accelerating organic matter decomposition (Lavoie *et al.*, 2005; Trettin *et al.*, 1996; Uzun *et al.*, 2020). As a result, assessing or forecasting the C balance of paludified forests requires an understanding of the amount of peat C that is lost through the decomposition process (Bona *et al.*, 2016; Charman, 2002; Lavoie *et al.*, 2005).

Despite the importance of the soil C stock contained in forested peatlands, few studies have assessed the effects of silvicultural treatments on this reservoir. Notable exceptions are Trettin *et al.* (1992), and McLaughlin *et al.* (2000) showing a negative impact on soil C after MSP in the short term. However, this latter study used a standard substrate (i.e., cellulose) to assess decomposition rate and it could be relevant to use local substrates considering that bryophytes show contrasting decomposition rates amongst species (Fenton *et al.*, 2010; Fissore *et al.*, 2019; Lang *et al.*, 2009). Several studies have focused on the effects of silvicultural treatments on forest tree productivity (Hagemann and Moroni, 2015; Henneb *et al.*, 2019; Lafleur *et al.*, 2011b), others on peatland forest development (*i.e.*, forest composition, structure, tree biomass and OM accumulation) over time since the last fire (Lecomte *et al.*, 2006; Lecomte *et al.*, 2006; Wang *et al.*, 2003), as well as climate change effects (Terrier *et al.*, 2012; Terrier *et al.*, 2013; Väliranta *et al.*, 2020; Zhang *et al.*, 2020). Likewise, the effects of drainage which are generally positive on tree growth (Ashton *et al.*, 2012; Laurén *et al.*, 2021; Roy *et al.*, 1999; Wieder and Vitt, 2006) and negative on soil C were also largely addressed (Turetsky *et al.*, 2011; Wickland *et al.*, 2010) and synthesized by the IPCC (Hiraishi *et al.*, 2014).

The objective of this study was to compare the nine years effect of two silvicultural treatments on soil C stocks and on organic layer decomposition rates in paludified forests of northwestern Quebec: (1) CPRS and (2) CPRS+MSP by plowing (or harrowing) against unharvested controls. More specifically, we aimed to:

- (a) quantify moss decomposition rates of three bryophytes species (*Pleurozium schreberi, Sphagnum capillifolium, and Sphagnum fuscum*) over two growing seasons,

- (b) identify the relationships between the treatments, micro-environmental conditions, and organic matter decomposition rate, and

- (c) assess the soil C stocks following the different silvicultural treatments.

We hypothesized that: (i) *P. schreberi*, an early-successional feathermoss species, would decompose more rapidly than the species *S. capillifolium* (an early succession sphagna) and *S. fuscum* (a late succession sphagna) (Fenton *and* Bergeron, 2006); (ii) CPRS+MSP would increase bryophyte tissue decomposition rate through habitat alteration in comparison with CPRS and controls; (iii) contrary to decomposition rate, soil C stocks would be higher in the control than in CPRS and CPRS+MSP.

2.4 Methods

2.4.1 Study area

The study area (49° 28' N, 78° 30' W) is located in the Clay Belt region of northwestern Quebec (Canada), in the black spruce-feathermoss bioclimatic domain (Figure 11). Forests are dominated by black spruce (*Picea mariana* [Mill.] BSP), ericaceous shrubs (mainly *Kalmia angustifolia* L. and *Rhododendron groenlandicum* (Oeder) Kron & Judd), feathermosses (*Pleurozium schreberi* [Brid.] Mitten), and Sphagna (*Sphagnum fuscum* [Schimp.] Klinggr*., Sphagnum capillifolium* [Ehrh.] Hedw.) (Fenton and Bergeron, 2011; Harper *et al.*, 2003). The Clay Belt originates from the sedimentary clay deposits of pro-glacial lake Barlow-Ojibway (Daubois *et al.*, 2015). Incorporated with gravel in the northern part due to a glacial re-advancement, the Clay Belt forms a compact deposit called Cochrane till on its northern edge (Daubois *et al.*, 2015). Soils are classified as Luvisols, Gleysols, and organic soils (*i.e.*, >40 cm thick) (Canadian Agricultural Services Coordinating Committee; Soil Classification Working Group, 1998). The cold climate is subpolar and subhumid continental with a short growing season length of 150-160 days between May and August, a mean annual temperature of 0 °C and precipitations of 909.1 mm (Joutel station, 1981-2010 period, Environment and Climate Change Canada, access 2021). In this region, poorly drained clay soil, low topographic relief, cold and moderately humid climate, and prolonged absence of fire, promote paludification (Bergeron *et al.*, 2001; Fenton *et al.*, 2005a; Payette and Rochefort, 2001).

2.4.2 Experimental design

The study is part of a larger project with the objective of better understanding the effects of silvicultural practices and environmental conditions on paludification (Laamrani *et al.*, 2014). It consists of nine experimental units, averaging 32 ha each, harvested by CPRS in the fall of 2010 (Henneb *et al.*, 2020). Mechanical preparation was applied to six of the units in the fall of 2011, with three using (MSP- plowing or harrowing) and the other three using disc trenching. In this study, because only plow was selected as MSP method, a total of six units are included in this study (Figure 11). While CPRS soil disturbance was restrained to the trails of harvester and forwarder machines (Harvey and Brais, 2002), a plow equipped with 0.91 m-diameter disks deeply mixed the soil all over the harvested area in two parallel passes (Henneb *et al.*, 2015). Plow treatment showed greater reduction of the organic layer thickness compared to other MSP treatments used in this area (Henneb *et al.*, 2015). Additionally, four experimental control units of 10 ha (i.e., without harvest or MSP) were selected in the surrounding forests in areas showing similar sites conditions i.e., black spruce-feathermoss or sphagnum dominated stands, relatively flat topography, and 60-80% tree density based on a forest inventory caried out in 2010 before harvest. A predictive map, representing the conditions that prevail before the harvest of the forest (i.e. before 2010), indicated an organic layer that ranged from 27 to 62 cm in thickness and stand volumes of 104 to 207 m^3 ha^{-1} (Laamrani *et al*, 2014). Low forest productivity was found on site with a very deep organic layer (>85 cm) (Laamrani *et al*, 2014). The predictive map was based on field measurements of the organic layer along several transects across the landscape, including the experimental units (Laamrani *et al*, 2014). In total, the experimental design included three CPRS, three CPRS+MSP and four control experimental units (Figure 11). To take into account the large variability within experimental units, some experimental units (two CPRS, two CPRS+MSP and one control) were divided in sub-units that were 100 m apart in average. In each unit or sub-unit, three 10 m x 10 m (100 m²) sampling plots, at least 50 m apart, were randomly established in the summer of 2019, for a total of 45 plots (3 treatments * 5 units * 3 sampling plots) (Figure 11). Decomposition rate of bryophytes moss bags, environmental habitat factors and soil C stocks were recorded in each sampling plot.

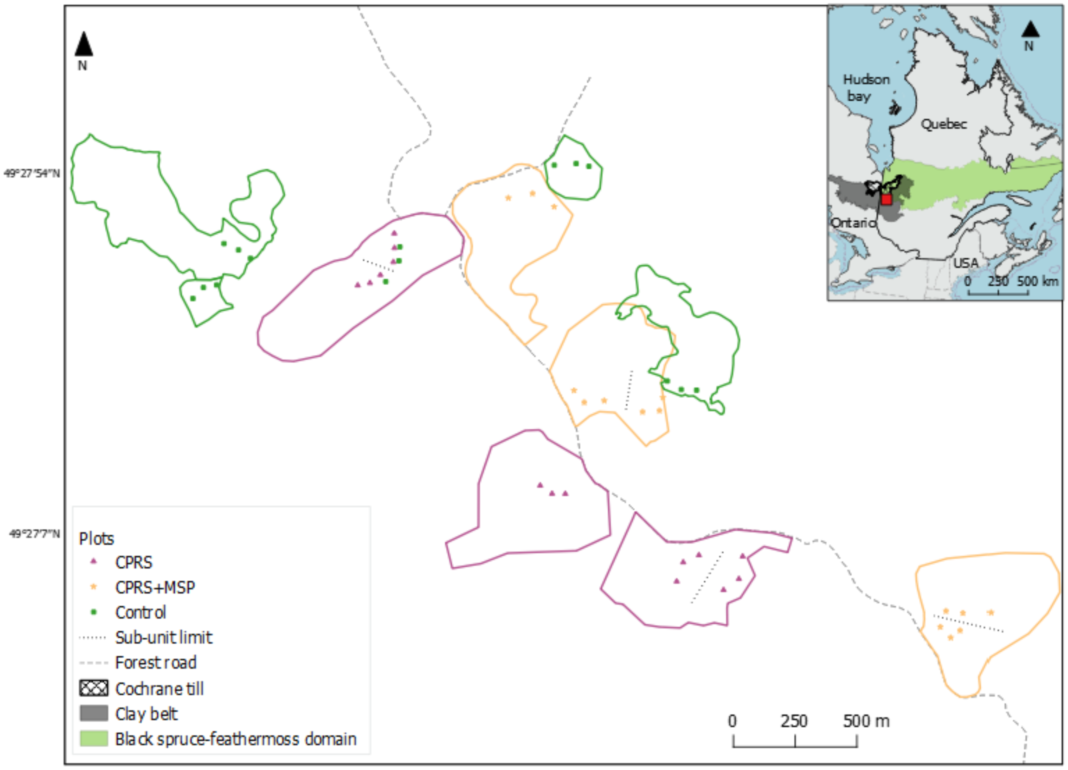

Figure 11
Location of the study area, treatments (cutting with soil protection and regeneration (CPRS), CPRS+MSP mechanical soil preparation (MSP) by plowing, and controls).

2.4.3 Field sampling and measurements

Decomposition rates . We estimated the decomposition rates (mass loss) of three bryophytes species (*Pleurozium schreberi, Sphagnum capillifolium, and Sphagnum* *fuscum*) for one and two growing seasons (*i.e.*, 4 months and 16 months, June to October 2019 and June 2019 to October 2020). Each bryophyte species sample was inserted into individual decomposition bags (10 cm x 10 cm) made of 1 mm nylon mesh. For the Sphagnum species, the top of living stem (the capitulum), was previously removed from the sample to avoid their continuous growth once buried (Faubert and Rochefort, 2002; Fenton *et al.*, 2010). A total of 540 decomposition bags (3 treatments * 5 units * 3 plots * 3 bryophytes groups * 2 depths * 2 seasons) were installed at two depths, *i.e.*, 10 cm and 20 cm from the ground surface. Half of the bags (270) were used to estimate mass loss during the first growing season (*i.e.*, 2019) while the other half were left in place to estimate decomposition over two growing seasons (*i.e.*, 2019 and 2020). The bags for the second sampling period were placed at a short distance (less than 2 m) from the first season bags to avoid soil disturbance while recuperating the first set of bags.

Sample initial and final dry masses were used to estimate mass loss (Equation 1); however, the initial drying of samples to be incubated is likely to damage the tissue of the substrate and modify their decomposition rate (Moore *et al.*, 2007). The initial dry mass of the substrates incubated was thus determined using a linear regression between dry and wet masses from other unincubated samples for each bryophyte species (Appendix A, chapter 2) for the regression curves). The wet mass of samples was determined using the salad spinner method used in other studies (Fenton *et al.*, 2010; Frego and Carleton, 1995). It consists of soaking the samples in water to saturation, spinning the samples to extract all the surface water and to obtain a final constant wet mass. The wet unincubated samples were then dried at 30° C to constant mass, weighed, and a relationship between the wet and dry masses thus obtained was established. At the end of the growing season 2019 and 2020, bags were retrieved, cleaned to remove all new material (*e.g.*, roots, mud, other organic materiel), dried and weighed. The final dry mass of the remaining substrates was determined to evaluate mass loss and rate of decomposition (Equation 1).

$$Decomposition\ rate\ = \frac{(Initial\ dry\ mass\ -\ \ Final\ dry\ mass)}{Initial\ dry\ mass}$$

Equation 1

Moreover, the percent mass loss over time (year) was fitted to an exponential model (non-linear regression with “drm()” function of “aomisc” package, R Core Team 2018) to estimate the exponential mass loss constant or decay constant rate k for each treatment and moss type. The exponential mass loss constant is the fraction of the stored quantity that is lost per unit time, and it was used to model mass remaining in soil reservoir (Lang *et al.*, 2009; Olson, 1963; Trofymow *et al.*, 2002).

Soil carbon stocks . In September 2019, organic and mineral soil volumetric samples were collected within each 100 m² sampling plot (Figure 11). Organic soil layers were classified according to the Von Post humification scale: fibric (Of, Von Post 0 to 3, non or only slightly decomposed organic matter, where the plant material is highly distinguishable); mesic (Om, Von Post 4 to 7, moderately decomposed organic matter); and humic (Oh, Von Post 8 to 10, nearly complete decomposition). One sample of each organic soil layer (fibric, mesic, humic) was collected using a template (15cm*15cm*sample thickness) and one sample from the top of mineral horizon was collected with a cylindrical sampler (height = 5 cm, inner diameter = 5 cm). The individual thickness of each sample as well as the entire depth of the organic layer were recorded.

For thick organic layers (*e.g.*, more than 50 cm in depth), samples not accessible with the template were collected with a half-russian corer (height = 50 cm, inner diameter = 5 cm) or with an auger (height = 20 cm, inner diameter = 5 cm) (Heinemeyer *et al.*, 2018; Hribljan *et al.*, 2015). Moreover, the mineral horizon could not be reached in plots where the peat (organic) layer was thicker than 1.5 m (the length of auger).

In the laboratory, a total of 152 samples were dried and sieved following standard procedure. Organic samples (Of, Om, Oh) were dried in an oven at 65°C for 2 days and passed through a 6 mm sieve to remove coarse elements (roots, buried wood, rock fragments, etc.). Mineral samples were air dried, mixed, and passed through a 2 mm sieve. Sieved organic and mineral samples were weighed to assess their density, and representative individual sub-samples of each were passed through a LECO TruMac CNS combustion analyzer (LECO Corp., St. Joseph, MI, USA) to determine the organic C concentration. The C stocks for each of the profiles were calculated as the sum of the stocks for each sample (Equation 2).

$$C\ stocks\ (g/cm^2) = C\ Concentration\ (\%) * Bulk\ density\ (g/cm^3) * thickness\ (cm)$$

Equation 2

Micro-environmental conditions. The treatments that were applied can affect the soil micro-environmental conditions by impacting forest cover and displacing and mixing soil layers. Thus, several micro-environmental conditions were assessed. For instance, the soil layer types (*i.e.*, the environment in which the decomposition bags are in contact with) were classified using the peat Von Post humification scale as Of, Om or Oh (https://soilsofcanada.ca/soil-formation/horizons.php) or as mineral soil. Air and soil temperature at two depths were also measured (four measures/day) using temperature loggers (thermochron iButton; Maxim Integrated, San Jose, California, United States) during the first and second incubation seasons. In each replicated treatment, four loggers were installed (two at 10 cm depth in two different plots, one at 20 cm depth and another for air temperature in the third plot), for a total of 60 loggers (3 treatments * 5 units * 4 loggers).

To measure the range of water table fluctuations, 90 1.2-meter-long steel rods (2 rods * 3 plots * 3 treatments * 5 units) were installed for three months (10 June to 10 September 2019). The depth of the oxic zone was defined as the distance between the ground surface to the bottom of the orange or brown rust on the rods (Fenton and Bergeron, 2006; Lavoie *et al.*, 2006b). Rust indicates metal corrosion due to the presence of oxygen and water-humidity, and marks the zone of movement of the water table level (Owens *et al.*, 2008).

In September 2019, the organic layer thickness in each replicated treatment was estimated by averaging 20 measurements along 2 orthogonal transects per plot, for 900 measurements in total (3 treatments * 5 units * 3 plots * 20 measurements). Finally, canopy cover (%) was determined at three different places in each plot using a densitometer (3 treatments * 5 units * 3 plots * 3 places = 135 measurements).

2.4.4 Statistical analyses

A mixed linear model was used to determine the effects of bryophyte species, treatments (CPRS, CPRS+MSP, control) and micro-environmental conditions (*i.e.*, soil temperature, water table level, depth (10 cm versus 20 cm), canopy cover and soil layer type) on moss decomposition rate. In this model, treatment and bryophyte types are considered as fixed effects, while micro-environmental variables (soil layer type and depth) are considered as covariates and experimental units and sub-units are considered as random effects. The final decomposition model was identified using the stepwise of variables selection (backward-forward selection method) based on the Akaike Information Criterion (AIC). We used a Principal Component Analysis (PCA) to evaluate and visualise the correlations between micro-environmental variables. We used causal path analysis to evaluate the direct and indirect causal links between micro-environmental variables and treatments, and their effect on decomposition rate. The structural model fitted was validated based on path analysis constraints (i.e., P-value (Chi-square) >0.05, Root Mean Square Error of Approximation < 0.05; Comparative Fit Index > 0.98; Comparative Fit Index > 0.98, Standardized mean square residual < 0.05). Prior to path analysis model specification, normality test was verified (Mardia kurtosis coefficient < 20).

The effect of treatment on total soil C stocks was determined using a mixed linear model with experimental units and sub-units as random effects and treatments as a fixed effect. Log-transformation was applied to the C stock values to better meet model assumptions (normality and equality of variance). Initial organic layer thickness (*i.e.*, before harvesting in 2010) was included as a covariable. The initial organic layer thickness data was extracted from the 10 m resolution organic layer thickness map produced by Laamrani *et al.* (2014) (see Appendix B, chapter 2). For controls units (surrounding harvested sites), we assumed that organic layer thicknesses measured in 2019 were like the initial organic layer in 2010. However, because control units tended to be more paludified than harvested units, we selected control plots in control areas that supported a forest canopy similar to that of pre-harvested areas based on canopy openness (<80%). Four control plots out of 15 presented a high degree of paludification and canopy openness and were thus excluded (Appendix B, chapter 2). The analyses of treatment effects on soil C stocks were performed for total soil profile, individual soil layers, soil bulk density, C concentration and layers thickness. Log-transformation was applied to the bulk density and layer thickness values. Because the current layer thickness depended on the initial thickness, the latter was also used as a covariate in the organic layer thickness model.

Differences among treatments were tested by a post-hoc analysis using a Tukey criterion for multiple comparisons. All results were considered significant at $p \leq 0.05$. These analyses were performed with the R software (v. 3.5.0; R Core Team 2018) using the following packages and functions: lme4 and lmerTest for linear mixed models, function bootMer() to get confidence intervals around modelled data from mixed model, emmeans for multiple comparisons, FactoMineR for PCA, MASS for model selection by Stepwise method and lavaan and sem() for path analysis (Rosseel, 2012).

2.5 Results

2.5.1 Decomposition of bryophytes

Mass loss was significantly influenced by treatment type ($p = 0.0236$, df=2), bryophyte type ($p < 0.0001$, df= 2), soil layer type ($p = 0.0335$, df=5), and the interaction between bryophyte and treatment types ($p = 0.0016$, df=4) (Tableau 7).

The average percent bryophyte mass loss varied among treatments with 21.6 % ± 0.13 (standard error) in CPRS+MSP; 11.6 % ± 0.23 in CPRS and 9.5% ± 0.21 in controls for two growing seasons (16 months). However, only the CPRS+MSP treatment showed significantly greater mass loss than the control treatment. Also, *P. schreberi* decomposed significantly faster than sphagnum species (*S. capillifolium* and *S. fuscum*). A similar trend was indicated by the average annual exponential mass loss constant k which was greater from CPRS+MSP treatment (0.118 ± 0.022) than in CRPS alone (0.061 ± 0.011), or in controls (0.0476 ± 0.0327). In terms of bryophyte, *P. schreberi* had a faster rate (0.101 ± 0.021) than *S. capillifolium* (0.085 ± 0.026) or *S. fuscum* (0.032 ± 0.022) with an average for both sphagnum of 0.058 ± 0.019. The mass loss of *P. schreberi* and *S. fuscum* did not vary with treatment, only the mass loss of S. *capillifolium* was significantly greater in CPRS+MSP than in controls while it was intermediate in CPRS (Figure 12).

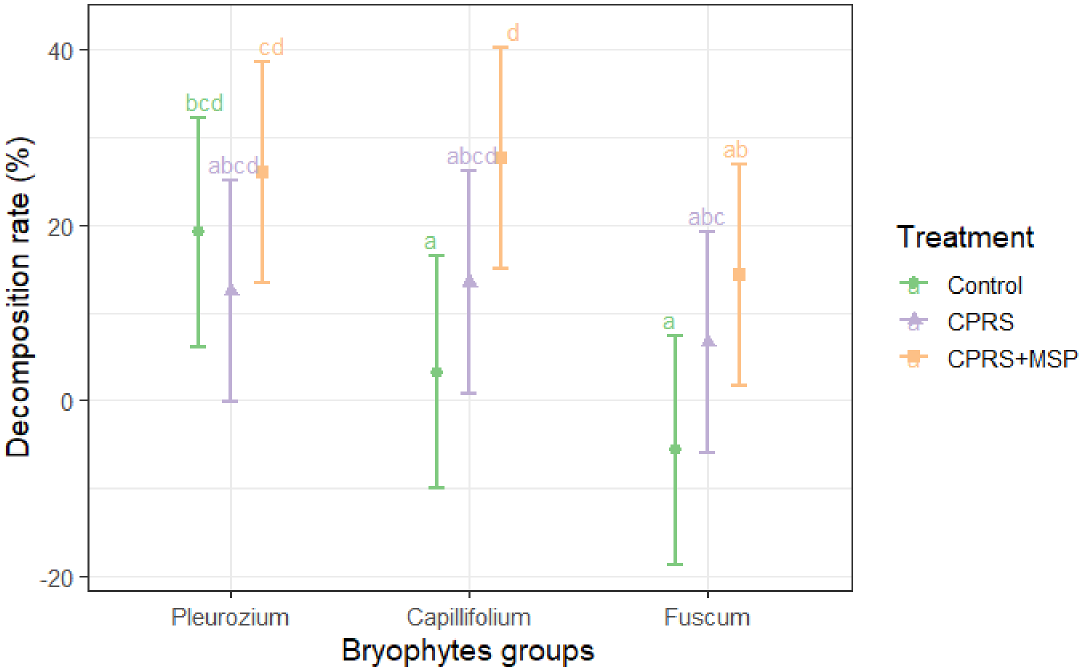

Figure 12
Percent mass loss (least-squares means of the model) for bryophyte species: *Pleurozium schreberi, Sphagnum capillifolium, Sphagnum fuscum*) in interaction with treatments (CPRS, CPRS+MSP and control) over two growing seasons. Bars represent 95% confidence intervals. Letters indicate statistical differences (Tukey's range test), effects that do not share superscripted letters are significantly different.

Furthermore, mass loss was significantly influenced by soil layer type (P= 0.0335, df=5). After two incubation seasons, mass loss in the fibric soil layer was higher than in the humic soil layer (Tableau 7 and Figure 13). Despite mass loss appearing to be higher at the 10 cm depth than at 20 cm, the effect of depth was not statistically significant. The increase in soil temperature and water table level tended to increase bryophyte mass loss but were not statistically significant.

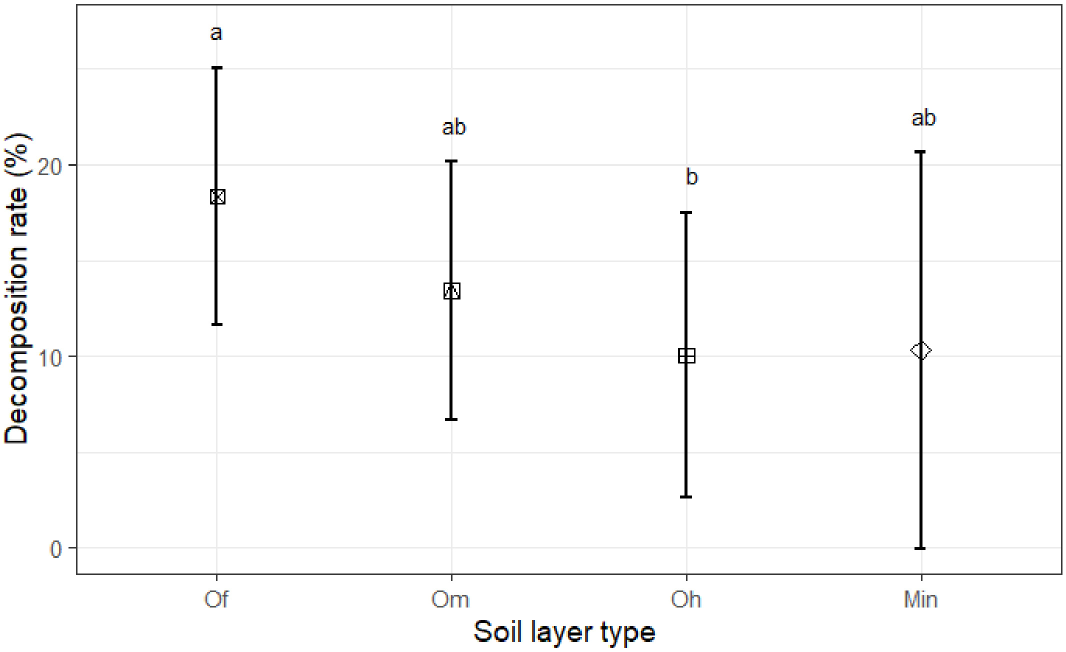

Figure 13
Percent mass loss (least-squares means of the model) according to placement in different soil layer type (Fibric Of, Mesic Om, Humic Oh and Mineral Min) for two growing seasons data. Bars represent 95% confidence intervals. Letters indicate statistical differences (Tukey's range test), effects that do not share superscripted letters are significantly different.

Tableau 7
ANOVA Summary of the best mixed model (selected by stepwise based on AIC values) for the effect of variables on decomposition rate. 40% of decomposition rate was explained by this model (0.42 Conditional R^2, fixed and random effects; 0.27 Marginal R^2, fixed effects only). Significant effects: ↑ positive effect, > greater.

Source	Sum Sq	Mean Sq	NumDF	DenDF	F value	Pr(>F)	Significant effects
Treatment	1961.634	980.817	2	6.937	6.743	0. 0236	CPRS+MSP ↑ > Control
Bryophyte	9664.644	4832.322	2	449.534	33.220	<0.0001	Pleurozium> Sphagnum
Soil layer type	1776.714	355.343	5	457.005	2.443	0.0335	Fibric> Humic
Depth	67.542	67.542	1	450.908	0.464	0.4959	None
Treatment:Bryophyte	2575.049	643.762	4	449.525	4.426	0.0016	CPRS+MSP:Sphagnum> Control: Sphagnum

2.5.2 Relationships between OM decomposition, treatment, and micro-environment

Generally, CPRS and CPRS+MSP were characterized by higher soil temperatures and a shallower water table, while the control was characterized by a greater thickness of the peat layer (Figure 14).

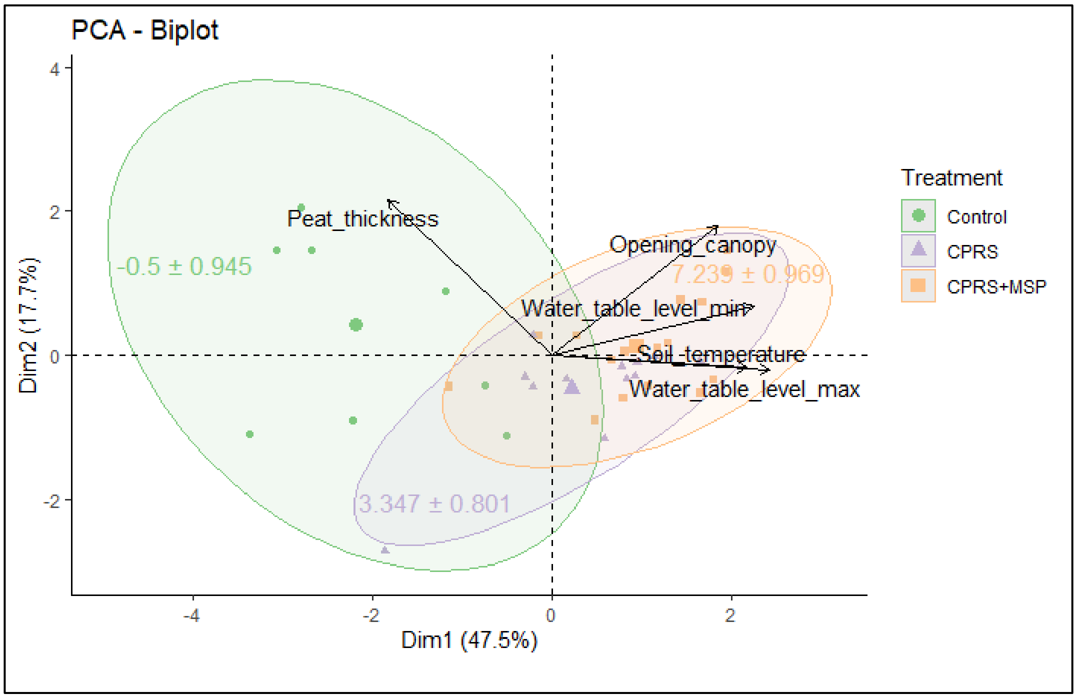

Figure 14
Principal component analysis (PCA) showing the relationships between the different habitat variables, with the first two axes explaining 65.2% of the total variation in the decomposition data. Values were the decomposition rate (%) average of three bryophytes according to treatment for one season, because most of the habitat factors were measured during the first growing season.

Overall, as suggested by the path analysis (Figure 15), treatments and micro-environmental conditions (soil layer type, water table and depth) had direct effects on bryophyte decomposition rates. The direct treatment effect was significant for Sphagnum species only. Soil layer type was the only direct effect that was significant for the decomposition rate of *Pleurozium schreberi*.

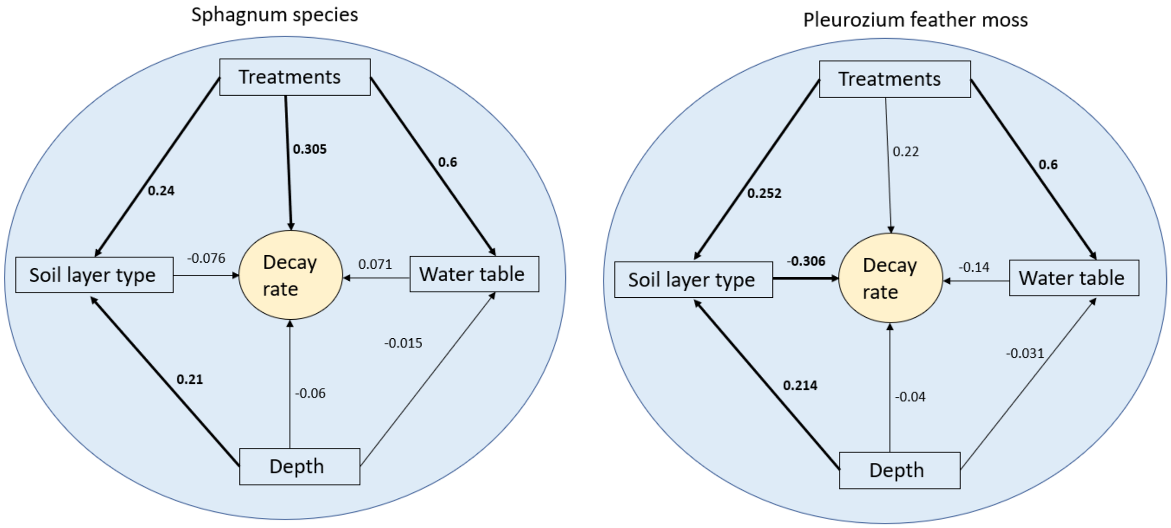

Figure 15
Path analysis reflecting the direct and indirect correlations influencing decomposition rate. Model specification, in R softward, was based on lavaan model structure with regression formulas among the observed variables, and model fitting with the sem(). The analysis was performed for two seasons of data and split into different bryophytes groups: Sphagnum species (*S. capillifolium* and *S. fuscum*) on the left graph, and feather moss *Pleurozium schreberi* on the right. 12% of the variance of the decomposition rate was explained by the variables. Bold lines represent significant correlations and thin lines non-significant ones. The coefficients represent the importance of each correlation.

2.5.3 Soil carbon stocks

Total organic soil C stocks were significantly related to the covariate, organic layer thickness before treatment (p= 0.002, df=1, see Tableau 8 and Figure 16) and the interaction between the treatments and the covariate was not significant. Therefore, the covariate could be used in the analysis which revealed no significant effect of treatments on total organic soil C stocks (Tableau 8).

The analysis was also performed by soil layer and revealed a statistically significant effect of treatment only for the fibric layer (Tableau 8). Deeper organic layers and the mineral layer C stocks were not significantly different among the treatments. Significant C stocks reduction in the fibric layer was found after CPRS+MSP treatment (P= 0.0284, df=2). This reduction was related to significantly reduced C concentration and soil thickness compared to the control in the fibric layer (Tableau 9).

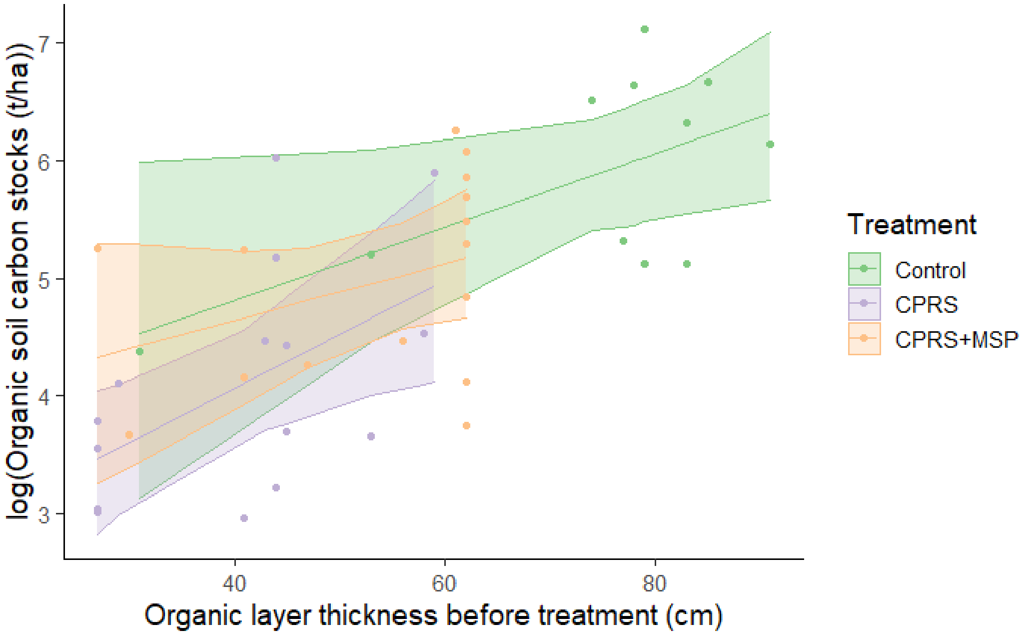

Figure 16
Organic carbon soil with confidence intervals according to initial layer thickness (before treatment in 2010) and treatments predicted by mixed model and based on fixed effects.

Tableau 8
ANOVA Summary of the mixed model for the effect of variables on the total organic carbon stocks in 2019 and different soil layers. 40% of total soil C stocks was explained by the model (0.37 Marginal R^2 only fixed effect; 0.4 Conditional R^2 fixed and random effects). significant results in italic.

Source	Sum Sq	Mean Sq	NumDF	DenDF	F value	Pr(>F)
Total organic layers						
Treatment	0.9870	0.4935	2	32.6034	0.8014	0.4574
Initial_thickness	*6.9244*	*6.9244*	*1*	*33.6009*	*11.2438*	*0.0020*
Treatment * Initial_thickness	0.4340	0.2170	2	33.6124	0.3524	0.7056
Fibric layer						
Treatment	*4.6204*	*2.3102*	*2*	*29.7183*	*4.0227*	*0.0284*
Initial_thickness	0.1999	0.1999	1	29.0116	0.3481	0.5598
Treatment * Initial_thickness	1.7732	0.8866	2	28.6928	1.5438	0.2308
Mesic layer						
Treatment	1.0343	0.5171	2	34	0.9616	0.3924
Initial_thickness	0.6406	0.6406	1	34	1.1911	0.2828
Treatment * Initial_thickness	1.4963	0.7482	2	34	1.3911	0.2626
Humic layer						
Treatment	0.1004	0.0502	2	23.2938	0.0425	0.9584
Initial_thickness	3.2843	3.2843	1	23.1021	2.7829	0.1088
Treatment * Initial_thickness	0.0612	0.0306	2	22.5832	0.0259	0.9744
Mineral layer						
Treatment	0.1060	0.0530	2	25.7146	0.1657	0.8482
Initial_thickness	0.0447	0.0447	1	28.5206	0.1398	0.7113
Treatment * Initial_thickness	0.0426	0.0213	2	28.2105	0.0667	0.9357

Tableau 9
Mean ± SE of Carbon (C) concentration, bulk density, and soil layer thickness among treatments (control, CPRS and CPRS+MSP) in fibric soil layer. Letters indicate statistical differences (Tukey's range test), effects that do not share superscripted letters are significant.

	Control	**CPRS**	**CPRS+MSP**
Fibric layer			
C concentration (%)	0.493 ± 0.0224 ^{a}	0.454 ± 0.0275 ^{ab}	0.353± 0.025 ^{b}
Bulk density (g/cm^3)	0.0351± 0.0092^{a}	0.0416± 0.0133^{a}	0.0410± 0.0117^{a}
Thickness (cm)	16.18 ± 5.205 ^{a}	4.41 ± 1.338 ^{ab}	3.28 ± 0.743 ^{b}

2.6 Discussion

Mechanical site preparation generally has a beneficial effect on tree growth and may be essential to regenerate the forest following harvesting, especially in boreal forested peatlands (Henneb *et al.*, 2019; Lafleur *et al.*, 2011a, 2011b; Thiffault *et al.*, 2013). In these environments, previous studies have shown the importance of disturbance whether by fire (Lecomte *et al.*, 2006; Renard *et al.*, 2016) or by mechanical site preparation (Henneb *et al.*, 2019) for establishing a productive forest. However, the impact of these treatments on soil organic C stocks and on soil organic matter decomposition rates are not well known and are a matter of concern. Our results showed that forest harvesting, and site preparation (CPRS+MSP) have an influence on bryophytes decay rates, however, no effects could be detected on total SOM stocks.

2.6.1 Decomposition of bryophyte

Our results showed that the decomposition rate of the bryophytes was dependent on the interacting effects of bryophyte type and silvicultural treatments as well as on the type of organic soil layer where they were incubated. The CPRS+MSP showed significantly greater mass loss than the control treatment, only for *S. capillifolium* (an early-successional sphagnum species) although a similar trend was also found for S. *fuscum* (Figure 12). As we expected, *P. schreberi* decomposed significantly faster than sphagnum species (*S. capillifolium* and *S. fuscum*) which is consistent with previous studies (Fenton *et al.*, 2010; Lang *et al.*, 2009). However, the decay rate of *P. schreberi* was not affected by the treatment to the contrary of Sphagnum species. The soil layer type (fibric, mesic, humic and mineral) significantly affected the decomposition rate of bryophytes and decomposition was faster in the fibric layer. Similarly, Belyea and Warner (1996), found that peat decomposability generally decreases with increasing humification. The higher microbial biomass found in the uppermost organic layers (Li *et al.*, 2020) may explain these results. The high degree of soil disturbance in CPRS+MSP treatment compared to CPRS created conditions more favorable for OM decomposition. For instance, the CPRS+MSP treatment can mix soil over 40 cm of depth (Henneb *et al.*, 2019) and improve soil temperature and aeration (McLaughlin *et al.*, 2000; Trettin and Jurgensen, 1992; Trettin *et al.*, 1996). The greater soil aeration can thereby increase microbial activity and shift anaerobic microbial populations to aerobic ones (Mallik and Hu, 1997; McLaughlin *et al.*, 2000). Studies using cellulose decomposition have shown that decomposition rates were greater on sites treated with bedding plow (100% disturbance of the soil surface), compared to trenched sites (about 45% disturbance of the soil surface) and whole-tree harvested system without mechanical site preparation respectively (McLaughlin *et al.*, 2000; Trettin and Jurgensen, 1992). Our results suggested that CPRS had a moderate effect on bryophytes decomposition rate because no significant differences were found both with Control and CPRS+MSP treatments. Fenton *et al.* (2010), found no significant effects of CPRS and partial cut (CP) on bryophytes decomposition rate. However, we found that sphagnum species (*S. capillifolium*) mass loss increased in CPRS+MSP treatment. In fact, the tissues of slow decaying Sphagnum species were more sensitive to environmental conditions such as temperature fluctuation than the more easily decomposable substrates (Conant *et al.*, 2008; Hobbie, 1996; Li *et al.*, 2020). This may explain the direct effect of treatments only on sphagnum species (*S.*

capillifolium and *S. fuscum*) in the path analysis (Figure 15) and a greater mass loss in CPRS+MSP compared to the control treatment for Sphagnum species than for *Pleurozium schreberi*.

Contrary to expectations, we found no effect of depth of moss decomposition bag placement on decomposition rates. Previous studies have shown greater decomposition near the surface (e.g. 5 to 10 cm) than deeper in the soil (e.g. >20 cm), potentially due to lower soil temperatures (Fenton *et al.*, 2010; Trettin *et al.*, 1996) and anoxic conditions. In our study, all bags were positioned in the surface layer at 10 or 20 cm and were all within the oxic zone, potentially explaining the absence of difference. The surface layer (0–20 cm), usually slightly or moderately humified (fibric and mesic), is associated with aerobic conditions and high microbial activity in peatlands (Maslov and Maslova, 2022). The changing redox condition due to the fluctuation of the water table level has been found to be more important in stimulating microbial activity than was temperature in peatlands (Hribljan 2012; Limpens, 2008; Yan *et al.*, 2020). For example, the Alkane-degrading microbes are known to be sensitive to the redox conditions in acidic peatlands (Yan *et al.*, 2020).

2.6.2 Soil carbon stocks

Nine years after silvicultural treatments, we found no significant differences between treatments in total soil profile C stocks (cumulated organic and mineral soil layer). Reduction of C stocks was only found for the fibric organic soil layer after CPRS+MSP compared to the Control, while CPRS treatment was intermediate. The other deeper organic layers (mesic and humic) and the upper mineral soil (a depth of 5 cm) were not affected.

Trettin et al. (2011) found an important reduction of SOC stocks in the upper layer (25cm) in the first year following harvesting and MSP in a forested mire. However, 11 years later, no difference was found in the whole soil C stocks (from forest floor to 1.5 m in the mineral soil) between whole-tree harvesting, site preparation (disk trenching and bedding) and the unharvested controls, suggesting a recovery of C stocks (Trettin *et al.*, 2011). Similarly, one decade after MSP, in conifer plantations, no MSP treatment effects were apparent on the total soil C stocks in Sweden (Nordborg *et al.*, 2006) and on several soil properties (bulk density, forest floor mass and depth) in British Columbia (Hope, 2007).

Significant reductions in forest floor C stocks were reported after 25 years MSP (ploughing, mounding or disc trenching) in a conifer forest site in Sweden but MSP did not affect the soil C stocks down to 30 cm (Mjöfors *et al.*, 2017). Nevertheless, other studies mentioned possible significant C stock reduction in very deep organic layers (100+ cm) or mineral soil or whole soil profile after silvicultural treatments depending on forest and soil type (James and Harrison, 2016). In our study, only the upper mineral soil layer was measured due to sampling constraints. However, because of the high relative importance of the organic layer for total soil C stocks in these ecosystems, the absence of change in the C stock of the organic layer may be a good indication of the lack of significant net SOC net changes. As indicated in meta-analyses (James and Harrison, 2016; Nave *et al.*, 2010); changes caused by harvesting and MSP, when present, are mostly observed in the organic layer.

While a wide range of outcomes have been observed for the effect of MSP on SOC stocks (Mayer *et al.*, 2020), a rapid recovery has generally been observed in forested peatland (within the first decade after harvesting) (Trettin *et al.*, 2011). Generally, organic C losses are observed in uppermost organic layer while the intensity of loss is related to disturbance intensity. Our results are consistent with this trend. Moss decomposition rate was faster in the CPRS+MSP compared to Control treatments in the surface layer (10-20 cm) and was more important in the fibric layer than in the deeper layers. Our results suggest that the effect of CPRS+MSP on SOM decomposition persists for several years after treatment in the fibric layer. The maintenance of SOC stocks may be explained by the rapid recovery of ground cover vegetation (graminoids, shrubs and bryophytes) (Anderson *et al.*, 2007; Shetler *et al.*, 2008; Trettin *et al.*, 2011).

We acknowledge that our estimates of SOC stocks are prone to large uncertainties (see confidence intervals of Figure 16) due to the heterogeneity of the site (Kristensen *et al.*, 2015; Laamrani *et al.*, 2014) and to the difficulties associated with the sampling of deep organic layers. According to a power analysis using the R function ‘power.t.test’, the magnitude of the effect between silvicultural treatment (CPRS, CPRS+MSP) and control should be three time higher for a difference to be detectable with a power of 80%. Given the large size and variability of SOC stocks in these forests, direct measures of net CO2 fluxes using eddy covariance as well as assessing dissolved organic C losses would definitely be a worthwhile complement to studies such as ours that assess SOM stock change.

2.6.3 Implication

Based on our results, CPRS+MSP had no impact on SOC stocks a decade after treatment. However, the faster decomposition rate that was observed in the uppermost layer of this treatment confirmed that the C cycle is impacted by the degree of disturbance to the soil. Our study suggests that the positive impact of MSP to limit paludification and to maintain tree growth and forest productivity in the Clay-Belt region (Henneb *et al.*, 2019; Lafleur *et al.*, 2011b) is not accompanied by important SOC losses. Our results show that harvesting followed by mechanical site preparation mostly affects the decomposition of hummock forming Sphagnum species and not Pleurozium mosses. Some studies have suggested that enhanced tree growth following MSP can offset the increased SOC losses that may occur (Lavoie *et al.*, 2005; Mjöfors *et al.*, 2017). Investigations on the overall impact of treatments on both soil and vegetation C stocks, as well as on biodiversity should provide potentially additional perspectives on the sustainably of intensive forest management in peatland forestry.

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3. MODELLING THE IMPACTS OF SILVICULTURAL PRACTICES AND MANAGEMENT ON C DYNAMIC IN BOREAL FORESTED PEATLAND

Article en préparation : Ange-Marie Botroh, David Paré, Xavier Cavard, Yves Bergeron.

3.1 Abstract

Boreal forested peatland contributes to climate change mitigation by accumulating large amounts of carbon (C) in moss-derived peat. They also represented an important wood supply in many areas in the world. However, because moss-derived peat slows down tree growth and threatens the potential wood supply, intense natural or silvicultural disturbances may reduce peat thickness and restore forest productivity. However, while these treatments can increase growth and C storage in trees, they may also reduce soil C storage. In this study, we used a modelling approach to explore the effects of short and long term of silvicultural treatments, management strategies as well as wildfire on ecosystem C dynamics in forested peatlands of northwestern Quebec. we compared, the effects of six scenarios: (1) low severity fire (LSF), (2) high severity fire (HSF), (3) Partial cutting (PC, 40% harvested), (4) total cut with protection of regeneration and soils (CPRS), (5) CPRS followed by mechanical site preparation (MSP) with plantation, (6) total cut followed by prescribed burning (PB) with plantation for two main forest types found in this landscape (black spruce-Sphagnum forest prone to paludification and black spruce-feathermoss). At the landscape level, we compare three forest management regimes with varying proportions of different silvicultural practices with a non-harvest scenario using the current fire cycle: (a) 400-years fire recurrence, (b) current even-age management (Business as usual, low rate of PC and high rate of total cut), (c) intensive alternative even-age management with total harvested rates in MSP and (d) low intensity alternative management with total annual harvested rate in PC. We found contrasting results between stand types. For black spruce-feathermoss stands, severe disturbance had greatest total C storage over the 250-year than low severity disturbance events (HSF>MSP>LSF=PB>CPRS>PC) because tree component is the main driver of the overall trend. Conversely, in black spruce-Sphagnum, low severity disturbances led to the greater C storage (CPRS> HSF=LSF=MSP>PB>PC) due to the dominant role of moss derived C stocks. Low C stocks in PC were observed for both forest types and is probably related to high tree mortality. The delay for the switch from source to sink following disturbance was reduced under severe disturbance for both forest types (in order) : PB (14 years), MSP (15 years), HSF (17 years), PC (17 years) , CPRS (26 years), LSF (30 years) in black spruce-Sphagnum, and HSF (21 years), MSP (28 years), and PB (29 years), CPRS (31 years) , LSF (42 years), PC (remained a source) in black spruce-feathermoss. The contrasting results of the effect of PC in BSSP compared to BSFM forests are due to a major role of moss in the former. These findings suggest that in sites with very low productivity, the contribution of mosses is essential to offset reduced C inputs from trees. In paludified areas, intensive silvicultural with plantation practices may offer a practical compromise for stands with relatively high productivity, while in very low-productivity sites, conservation strategies may be more appropriate. Our study may guide management decisions in categorizing practices according to initial forest condition and type as well as climate change mitigation strategies.

Key words: anthropogenic disturbance, black spruce, bryophyte, carbon storage and flux, fire, peat.

3.2 Résumé

Les forêts boréales entourbées contribuent à l'atténuation du changement climatique en accumulant de grandes quantités de carbone (C) dans la tourbe dérivée des mousses. Elles représentent également un important approvisionnement en bois dans de nombreuses régions du monde. Cependant, comme la tourbe ralentit la croissance des arbres et menace l'approvisionnement potentiel en bois, des perturbations naturelles ou sylvicoles sévères peuvent réduire l'épaisseur de la tourbe et restaurer la productivité de la forêt. Cependant, si ces traitements peuvent augmenter la croissance et le stockage du C dans les arbres, ils peuvent également réduire le stockage du C dans le sol. Dans cette étude, nous avons utilisé une approche de modélisation pour explorer les effets à court et à long terme des traitements sylvicoles, des stratégies de gestion et des feux de forêt sur la dynamique du C de l'écosystème dans les forêts entourbées du nord-ouest du Québec. Nous avons comparé, les effets de six scénarios : (1) feu de forte intensité (HSF), (2) feu de faible intensité (LSF), (3) coupe partielle (PC, 40% d’arbres récoltés), (4) coupe totale avec protection de la régénération et du sol (CPRS), (5) CPRS suivie par la préparation mécanique du terrain avec plantation (MSP), (6) coupe totale suivie de brûlage dirigé avec plantation (PB) dans deux principaux types de forêts du paysage (épinette noire - sphaigne et épinette noire - mousse hypnacée). Au niveau du paysage, nous comparons trois régimes d’aménagement avec des proportions variables de différentes pratiques sylvicoles avec un scénario de non-récolte utilisant le cycle actuel des incendies : (a) récurrence des incendies de 400 ans, (b) gestion équienne actuelle (faible taux de PC et taux élevé de coupe totale), (c) gestion alternative intensive avec des taux de récolte totale en MSP et (d) gestion alternative à faible intensité avec un taux de récolte annuelle totale en PC. Nous avons trouvé des résultats contrastés entre les types de peuplements. Pour les peuplements d’ épinette noire - mousse hypnacée, les perturbations sévères ont entraîné le plus grand stockage total de C sur 250 ans en comparaison avec des perturbations de faible intensité (HSF > MSP > LSF =PB > CPRS > PC), car la composante arborée est le principal facteur expliquant la tendance générale. Inversement, dans la forêt d’ épinette noire - sphaigne, les perturbations de faible intensité ont entraîné un stockage de C plus important (CPRS > HSF = LSF =

MSP > PB > PC) en raison du rôle dominant de la tourbe. Le délai de passage de source de C au puits de C à la suite d'une perturbation a été réduit en avec les perturbations sévères pour les deux types de forêts (dans l'ordre) : PB (14 ans), MSP (15 ans), HSF (17 ans), PC (17 ans), CPRS (26 ans), LSF (30 ans) dans l' épinette noire - sphaigne, et, HSF (21 ans), MSP (28 ans), et PB (29 ans), CPRS (31 ans), LSF (42 ans), PC est resté une source) dans l'épinette noire - mousse hypnacée. Ces résultats suggèrent que dans les sites à très faible productivité, la contribution des mousses est essentielle pour compenser les apports réduits de C des arbres. Dans les zones paludifiées, les pratiques de sylviculture intensive et de plantation peuvent offrir un compromis pratique pour les peuplements à productivité relativement élevée, tandis que dans les sites à très faible productivité, les stratégies de conservation peuvent s'avérer plus appropriées. Notre étude peut guider les décisions de gestion en classant les pratiques en fonction de l'état initial et du type de forêt, ainsi que les stratégies d'atténuation du changement climatique.

Mots clés : bryophytes, feu, épinette noire, tourbe, perturbation anthropique, stockage et flux de carbone.

3.3 Introduction

Boreal forests store almost half of global forest carbon (C) in living vegetation and soil (Ashton *et al.*, 2012; Dixon, 1994; Pan *et al.*, 2011). Soil pool represents more than two thirds of this boreal C stocks, due notably to peat formation (Dixon, 1994). Paludification is the most common peat (*i.e*, soil organic layer thickness greater than 40 cm) formation in the boreal forest (Charman, 2002; Payette and Rochefort, 2001). It occurs when the productivity of plants, mainly bryophytes (*i.e.,* peat-moss such as *Sphagnum spp.*), is greater than the decomposition of organic matter in cold and poorly drained soils (Charman, 2002; Payette and Rochefort, 2001). Apart from the effect of edaphic conditions (*i.e.,* wet lowlands), paludification related to forest succession (i.e., successional paludification), may occur in uplands in prolonged absence of fire (Lecomte *et al.*, 2006; Roulet *et al.*, 2007; Sousa *et al.*, 2021). However, paludification by modified forest conditions (e.g., low nutrient availability, poorly drained soil, underground plant competition) slow tree growth and reduce forest productivity in long term.

In many parts of boreal area, particularly in Fennoscandia, Russia and Canada, forested peatlands are managed for wood production (Hånell, 1991; Nuutinen *et al.*, 2000; Payette and Rochefort, 2001). The development of silvicultural practices that maintain or restore forest productivity (e.g., by fostering organic matter decomposition and nutrient cycling) like drainage, more widespread in Russia, Finland, and Sweden, prescribed fire or mechanical site preparation is recommended (Lafleur *et al.*, 2016; Laurén *et al.*, 2021; Lavoie *et al.*, 2005; Paavilainen and Päivänen, 1995).

The Clay belt of northeastern Canada in Quebec is prone to paludification due to the low water permeability of clay soil, flat topography, cold climate, and the prolonged absence of fire (Bergeron *et al.*, 2001; Fenton *et al.*, 2010). The relative longer fire cycle (400-years), fosters low-severity soil burns that consumes partially the organic layer, leads to successional paludification between fire events (Fenton *et al.*, 2005; Lecomte *et al.*, 2006). In this landscape, successional paludification, through the deepening of the organic layer, reduces tree growth and can decrease black spruce productivity by 50–80% over the long term. (Simard *et al.*, 2007). It has been also suggested that low soil disturbance after harvest, such as careful logging around advanced growth (CLAAG; also called cut with protection of regeneration and soils (CPRS) in Quebec), Partial cut (PC) or winter harvesting, may accelerate paludification and reduce forest productivity (Fenton *et al.*, 2010; Lafleur *et al.*, 2010). Then, successional paludification can be reversible by either severe fires consuming a great amount of ground cover vegetation and peat or potentially by prescribed fire or intensive mechanical soil disturbance that deeply disturb the soil after harvesting (Henneb *et al.*, 2019; Laamrani *et al.*, 2014; Lavoie *et al.*, 2005).

According to the degree of soil disturbance, disturbance type (natural or silvicultural practices), by fostering organic matter decomposition (i.e., nutrient availability), reducing plant competition (moss species, shrub), can create favorable environmental conditions for the establishment more productive forest (Nguyen-Xuan *et al.*, 2000). For instance, high fire severity in the first century, can lead to productive and dense black spruce forest stand with thin OM layer and more feather moss than sphagnum (Lecomte *et al.*, 2006). In the short term (e.g., 20-30 years), tree growth was higher after HSF than CPRS (Lafleur *et al.*, 2016) and prescribed fire was higher than CPRS (Renard *et al.*, 2016) and seedling growth was better after mechanical site preparation with the reduction of OM layer (Henneb *et al.*, 2019). Then, these disturbances increase C storage in trees through tree growth, but they can have some opposite effect on C stocks. They could lead to direct C emission through OM combustion, or C indirect emission by organic matter decomposition, which can result in a negative C balance (Lavoie *et al.*, 2005; Trettin and Jurgensen, 1992; Uzun *et al.*, 2020). The increasing of tree growth can offset the losses of soil C, but whole ecosystem C offsetting is yet to be addressed (Trettin *et al.*, 1992; Williams *et al.*, 2016). There are few studies on the global C sequestration potential of these practices and their long-term trend on C storage in forested peatland. An overall negative impact of drainage on soil C, a well-known practice in peatland forested, remain largely addressed and soil OM losses could be compensated at long term by the tree growth in the some very poor peatland forested (Hommeltenberg *et al.*, 2014; IPCC, 2014; Turetsky *et al.*, 2011; Wickland *et al.*, 2010). Then, for sustainable management and climate change mitigation, forested peatland management in clay belt requires a thorough understanding of how various silvicultural practices influence C fluxes and stocks.

Modelling is a useful tool for the forecasting forest C dynamic and different trajectories under various disturbance events. There are models which simulate the C dynamics both in soil and vegetation (e.g., Biome-BGC, Forest-DNDC, 3-PG, STANDCARB, MASSIMO, SILVA, TRIPLEX-FLUX, EFISCEN, CO2FIX, CBM-CFS3) and others which are more specific in soil (e.g., YASSO, CENTURY, ROMUL, RothC) (Deluca and Boisvenue, 2012; Peltoniemi *et al.*, 2007). However, peatlands (mosses contribution and peat layer), despite the important contribution to the C cycle, do not account. The Carbon Budget Model of the Canadian Forest (CBM-CFS3) is adapted to forest management planning and consistent with the Intergovernmental Panel on Climate's (IPCC) C account guidelines (Kull *et al.*, 2014; Kurz *et al.*, 2009) . For this software has a beta-version of a moss module (MOSS-C; Bona et al. 2016) that can be used to model upland forested peatland dynamics.

The objective of this study was to assess by modeling the impacts of various silvicultural and management practices on C flux and stocks in boreal forested peatlands. Particularly, in the two main forest types found in this landscape (black spruce-Sphagnum forest (BSSP) and black spruce-feathermoss (BSFM), we compared at stand level, two natural disturbances and four different silvicultural practices: (1) low severity fire (LSF), (2) high severity fire (HSF), (3) Partial cutting (PC, 40% harvested), (4) total cut with protection of regeneration and soils (CPRS), (5) CPRS followed by mechanical site preparation (MSP) with plantation, (6) total cut followed by prescribed burning(PB) with plantation. At the landscape level, we compared three forest management regimes with contrasting proportion of some silvicultural practice to the current fire cycle : (a) 400-years current cycle, (b) current even-age management or business as usual (BAU, lower rate of PC and higher rate of total cut), (c) more intensive alternative management with total harvested annual rate by MSP and (d) low intensity alternative management with total annual harvested rate by PC. The following questions were addressed: (1) how do C flux and stocks vary among different scenarios? (2) Are the potential C losses by intensive silvicultural practices offsetting by C gain in tree and in when delay? (3) Is the goal of maintaining forest productivity in forested peatland by intensives management practices conflicting with climate change mitigation (sustainable management-based C sequestration)?

We hypothesized that total C stock increased more in low severity disturbance than high severity disturbance in the following order: CP > CPRS > CPRS + mechanical site preparation (MSP) > LSF > CPRS+ followed by prescribed burning (PB) > HSF. At the landscape level, the current fire cycle with no harvesting results in higher C storage followed by PC-management, current even-age management (business as usual (BAU)) and MSP-management. We applied the research version of CBM-CFS3 software on paludified or liable to paludification forests from forest inventory data of a forest management unit (FMU) located in Québec's clay belt zone. The moss extension for the non-forested peatland to this CBM- CFS3 research version was re-implemented and modified using R-language (R version 4; R Development Core Team 2021).

3.4 Methods

3.4.1 Study area

The study area corresponds to Forest Management Unit (FMU) 085-51 (between 50°09'N and 48°50'N latitudes, and 79°31'W and 78°05'W). FMU085-51, with an area of 991 616.3 ha, lies in the physiographic region of the Clay Belt, in the black spruce-feathermoss bioclimatic domain (Figure 17). Forests are dominated by black spruce (*Picea mariana* [Mill.] BSP), ericaceous shrubs (mainly *Kalmia angustifolia* L. and *Rhododendron groenlandicum* (Oeder) Kron & Judd), feathermosses (*Pleurozium schreberi* [Brid.] Mitten), and *Sphagnum* spp. (e.g., *Sphagnum fuscum* [Schimp.] Klinggr*., Sphagnum capillifolium* [Ehrh.] Hedw.) (Fenton *et al.*, 2010; Harper *et al.*, 2003). In the northern part of our study area, clay is incorporated with fine till that forms a compact deposit called Cochrane till (Daubois *et al.*, 2015). This northern part of the Clay Belt is characterized by a high risk of paludification whereas the southern part has medium and low risk of paludification (Mansuy *et al.*, 2018). The topography is flat, and soils are mainly classified as Gleysols, Luvisols, or Organic (Canadian Agricultural Services Coordinating Committee ; Soil Classification Working Group, 1998). The cold climate is subpolar and subhumid continental with a short growing season length of 150-160 days between May and August, a mean annual temperature of 0.6°C and precipitations of 907.6 mm (Joutel, Lebel Sur Quevillon, Lac Berry stations, 1981-2010 period, Environment and Climate Change Canada, accessed 2021). This cold and moderately humid climate associated with the poorly drained clay soil, the low topographic relief, the prolonged absence of fire and the abundance of *Sphagnum* promote paludification in this area (Bergeron *et al.*, 2001; Fenton *et al.*, 2010; Payette and Rochefort, 2001).

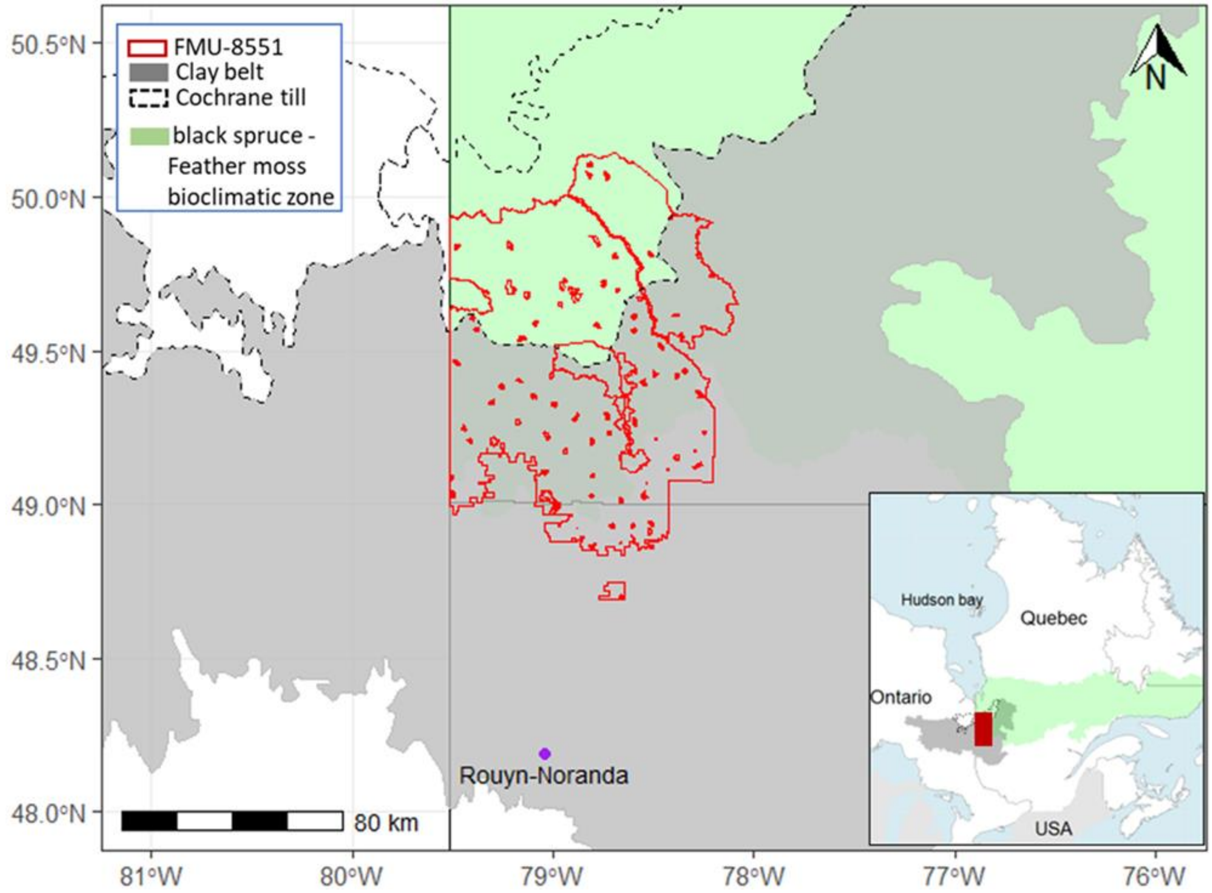


Figure 17
Location of the study area: Forest Management Unit 085-51, in the black spruce-feathermoss bioclimatic domain.

3.4.2 Model description

We used the operational carbon budget model of the Canadian Forest Sector version 3 (CBM-CFS3) to assess C stocks and flux after fire and sylvicultural practices over 250-year simulation period.

CBM-CFS3 structure. CBM-CFS3 track C stocks and stock changes in annual time-steps (Kurz *et al.*, 2009). C pools in CBM-CFS3 can be grouped into two groups: biomass and soil (Figure 5 in chap 1). Soil pools contain dead organic matter (DOM, Litter + debris + dead wood), organic (FH horizon) and mineral soil layers. We used a moss module (MOSS-C; Bona *et al.*, 2016) included in the research version of CBM. This module tracks the growth of live moss biomass and their turnover into two soil organic layers with contrasting decay rates (fast and slow decay) for two types of bryophytes (feather mosses and *Sphagnum* spp.). To increase the usability and flexibility of the moss module the MOSS-C module was re-built in R programming language (R version 4; R Development Core Team 2021). For more information about the model see Kurz *et al.* (2009) and Bona *et al.* (2013,2014, 2016).

Vegetation growth. The dynamic of living C biomass was based on stand growth simulation. CBM uses allometric equations developed for specific Canadian forest types to transform merchantable wood volume from yield curves to aboveground biomass and to link aboveground biomass to belowground biomass (Boudewyn *et al.*, 2007; Kurz *et al.*, 1996; Li *et al.*, 2003). The wood biomass of non-merchantable and sapling size trees is estimated by curve-smoothing algorithms (Kurz *et al.*, 2009). Net Primary Productivity (NPP) is used as C annual input. Moss productivity depends on moss ground cover and NPP (Bona *et al.*, 2016). The merchantable tree volume was used to predict tree canopy openness, which was then used to model bryophyte ground cover and NPP rates for each type of bryophyte. A delay in tree regeneration of 10 years was applied after fire and zero year for harvested practices. In this study, we adapted and developed different relationships and distinguished the effect of fire severity on moss ground cover based on field data collected by Fenton et *al.*, 2005 in the Clay Belt. Multiple regression models were applied to different variables (*e.g.*, canopy closure, basal area) to predict *Sphagnum* species ground cover according to fire severity (Botroh *et al*, 2024, appendix E, chapter 1). The ground cover of feathermoss was deducted from the *Sphagnum* species ground cover because we assume that feathermoss and *Sphagnum* species represent 100% of the ground bryophyte cover. The regression models indicate a higher feathermoss cover after HSF than after LSF in young stands, but feathermosses were gradually replaced by *Sphagnum* species at long term. The bryophyte cover model after LSF was used as reference for silvicultural practices with low severity on organic soil layer (CPRS, PC) and model after HSF as reference for high severity silvicultural practices (MSP, PB). Specific constant NPP to feathermoss and *Sphagnum* cited in Bisbee *et al.*, 2001 (Botroh *et al*, 2024, appendix C, chapter 1) were multiplied by the proportion of each type of bryophyte ground cover to estimate the annual inputs NPP for biomass. For more details see Botroh *et al.* (submitted).

Turnover and decomposition. Living biomass C is transferred to soil pools through turnover (*i.e.*, biomass mortality) and litterfall rates (Figure 5 in chap 1). Soil pools vary as C is transferred between different soil pools and or lost to the atmosphere through decomposition (Kurz *et al.*, 2009). Decomposition is simulated by applying decay rate that depends on the mean air temperature, the specific base decay rate at a reference temperature of 10 ◦C (Q10) (Kurz *et al.*, 2009). The mean air temperature of 0◦C and precipitations of 909 mm were used in our simulation (Joutel station, 1981-2010 period, Environment and Climate Change Canada, access 2021). Soil pools at the beginning of a model run were initialize by fire, the historical disturbance events. Parameters related to turnover, decomposition (base decay rate, Q10, etc.), C transfer, and disturbance matrices were modified based on published literature.

Disturbance matrices. Disturbance matrices, specific to Canadian ecozones, describe the proportions of C transferred between C pools, atmosphere, and forest products sector following disturbance events. Default fire and harvesting matrices and parameters of the spatial unit of the eastern boreal shield were used and modified to include variation in disturbance severity (appendix A, chapter 3). The proportion of burnt C after fire were calibrated using values cited in Shetler *et al*. (2008), and other studies (Andrieux *et al.*, 2018; Lecomte *et al.*, 2006) (appendix A, chapter 3). Low severity fires (LSF) tended to have lower impact on biomass and partially consume the organic layer while high severity fires (HSF) can almost completely consume the organic matter layer (Kasischke and Bruhwiler, 2002; Simard *et al.*, 2007). For instance, HSF consumed more living biomass (100% tree crown, 100% feather moss and 90% *Sphagnum*) and organic soil layer (45% in average) than low fire severity with low biomass consumption (83% tree crown, 100% feather moss and 77% *Sphagnum*, 20% organic soil layer in average) (Andrieux *et al.*, 2018; Benscoter and Vitt, 2008; Kasischke and Bruhwiler, 2002; Shetler *et al.*, 2008). The default fire matrice was used to reflect HSF while the proportion was halved to represent LSF (Kasischke and Bruhwiler, 2002; Simard *et al.*, 2007). For CPRS, the default CBM matrice remained unchanged. However, the CPRS matrice was modified for MSP, especially on soil pool based on previous empirical studies (Kpodo and Thiffault, 2014; Trettin *et al.*, 1992). In addition, the default partial cut matrice of 50% of the merchantable biomass harvesting was adjusted to 40% (appendix A, chapter 3). The proportion of C from bryophytes also were calibrated according to the different disturbance (see scenario and assumptions section).

Outputs. Some outputs can be used to evaluate simulation scenarios at the annual time step: total ecosystem C stocks, C stock changes (delta ecosystem), and emissions. Total ecosystem C storage was estimated as the sum of C in all the live biomass and soil pools (litter, debris, dead wood and organic (FH horizon) soil layer), mineral soil was excluded. We also focused on live biomass and organic soil layer pools. We presented all outputs in Mg C ha^{-1} (1 Mg C ha^{-1} = 1 t C ha^{-1} = 10^6 g C ha^{-1}) for stand level and Mt C (1 Mt C =10^6 t C = 10^6 Mg C = 10^{12} g C) for landscape level. A positive value for the annual change in C stock indicates a net gain in C stocks (“sink”), a negative value indicates a net loss (“source”) (IPCC, 2006; Kull *et al.*, 2014).

3.4.3 Data

We used forest inventory data of FMU 8551 completed in 2010 (Québec government agency Ministère des Forêts, de la Faune et des Parcs, 2010) and merchantable volume yield curves from Québec government agency Bureau du Forestier en Chef (BFEC). In this study, natural black spruce stands that were paludified or liable to paludification were selected in the forest inventory based on: fire origin, ecological forest type (black spruce-feathermoss-BSSFM and black spruce-*Sphagnum* dominated stand -BSSP), forest density (high to low), drainage class (moderate to poor) and surface deposits (*e.g.*, organic, till, glaciolacustrine). A total of 32, 987 black spruce stands originated from fire representing an area of 179, 526.4 ha were selected for the simulations (18% of FMU area). BSSP forests represented 82% and BSFM forests 18% of selected stands (Tableau 10). These selected stands presented various age classes but were dominated by the older age class of 120+ years. The uneven-aged stands, accounting for only 11% of the selected area, are included in the age class of 120+ years because CBM does not explicitly consider multiple cohorts.

Different black spruce yield curves, from less to high productive relatively, reflecting the difference in ecological type (BSSP vs. BSFM), were assigned to stand and disturbance groups (see section scenario and assumptions, and Botroh *et al*, 2024, appendix G, chapter 1, not yet published). These yield curves (volume-age) were made with forest growth simulation model NATURA (stand-level) and ARTEMIS (tree scale) (Auger, 2017). The evolution curve is constructed from a regression done on the same permanent and ecologically similar plots of the black spruce-feather-moss bioclimatic subdomain. However, the yield curve of wooded peatland (unproductive black spruce stands) used, provided by CBM team and was developed on empirical data of natural black spruce peatlands in Ontario and ecologically similar regions in Quebec (Penner *et al.*, 2008). We adjusted the curves to align with our simulation assumptions (Botroh *et al*, 2024, appendix G, chapter 1, not yet published). The selection and analysis of yield curves were based on site quality index (SQI) from previous studies (Garet *et al.*, 2009; Simard *et al.*, 2007). We maintain the beginning of the curve until the maximum volume, and after the inflection point, apply a decreasing exponential function as described in Lecomte et al., 2006 (Botroh *et al*, 2024, appendix G, chapter 1, not yet published).

Tableau 10
Selected stands of FMU 8551 in the 2010 forest inventory data.

Forest group	Characteristics	Area (ha)
Black spruce-*Sphagnum* (BSSP)	dominant moss species: *Sphagnum* low forest density: dominant cover density between 40% and 60% poor drainage class	148738 (82% selected stands)
Black spruce-feathermoss (BSFM)	dominant moss species: feathermoss intermediate forest density: dominant cover density between 60% and 80% relatively well drained to moderate drainage class	32762,6 (18% of selected stands)

3.4.4 Scenarios and assumptions

We simulated fire and harvesting scenarios over 250 years at stand and landscape level to compare the C dynamic trend (Tableau 11). Model initialization was performed with the historical fire cycle of 140 years (Bergeron *et al.*, 2004).

Stand level . At the stand level, scenarios included HSF, LSF, CPRS, CPRS, MSP, PB and PC were applied and compared separately in the two-forest type found in the landscape: black spruce-*Sphagnum* dominated stands (BSSP), and black spruce-feathermoss dominated stands (BSFM). Our simulation hypothesis was primarily based on the dominance of ground cover by bryophytes, which can reflect the initial site's quality (productivity and soil condition) (Tableau 10, Appendix F, chapter 3).

Because *Sphagnum* species decompose more slowly than feathermoss and modify site drainage (Benscoter and Wieder, 2003; Fenton *et al.*, 2010; Lang *et al.*, 2009), we assume that BSFM were more productive and well drained than BSSP stand. Moreover, BSSP stand was initialized, and originated from low fire severity (LSF) while BSFM stand was initialized by high severity fire (HSF) based to resistant of *Sphagnum* species to fire compared to feather mosses (Benscoter *et al.*, 2011; Benscoter and Wieder, 2003; Terrier *et al.*, 2014). The simulation was done on an initial stand age of 120 years old because it is the dominant landscape age classes of UAF-85-51.

Tableau 11
Scenarios at the stand and landscape levels.

Scenarios	Description
Stand-level	
BSSP stand	
Scenario 1	LSF in BSSP stand
Scenario 2	HSF in BSSP stand
Scenario 3	CPRS in BSSP stand
Scenario 4	MSP in BSSP stand, Mechanical site preparation after clearcut with plantation
Scenario 5	PB in BSSP stand, Prescribed burning after clearcut, with plantation
Scenario 6	PC in BSSP stand, Partial cut with 40% of commercial stems extraction
BSFM stand	
Scenario 7	HSF in BSFM stand
Scenario 8	LSF in BSFM stand
Scenario 9	CPRS in BSFM stand
Scenario 10	MSP in BSFM stand, Mechanical site preparation after clearcut with plantation
Scenario 11	PB in BSFM stand, Prescribed burning with plantation
Scenario 12	PC in BSFM stand, 40% of commercial stems extraction

LSF: low severity fire

Tableau 11 (Suite)
Scenarios at the stand and landscape levels.

Landscape-level	
Scenario 1	400-year fire cycle (current cycle, 0.25% annual burn rate) with the occurrence of low fire severity LSF,
Scenario 2	Current even-age management or business as usual (BAU), 0.90% annual harvest rate: 0.855 % total cut CPRS (0.38 % of CPRS, and 0.48 % of MSP followed clearcut) and 0.045 % PC
Scenario 3	Alternative management MSP, total annual rate on 0,9 % in MSP: increase the proportion of MSP (clearcut + MSP + plantation)
Scenario 4	Alternative management PC, total annual rate on 0,9 % in PC, increase the proportion of PC

HSF: high severity fire
CPRS: cut with protection of regeneration and soils
MSP: mechanical site preparation
PB: prescribed burning
PC: partial cut
BSSP: black spruce-*Sphagnum* dominated stands
BSFM: black spruce-feathermoss dominated stands

Fire scenario. Fire consumes a proportion of trees, bryophytes and upper organic soil. HSF and LSF differ according to the proportion pools burned and is related to forest type. The proportion of C burning for living biomass and organic soil pools, according to fire severity were calibrated the cited values provided in Shetler *et al.*, (2008) see appendix A, chapter 3. Higher organic C soil layer proportion burned was applied in BSFM stand as the HSF effect reference, and the lower C proportion burned in BSSP stand as the LSF effect reference. A factor of approximately 1.71 (Lecomte *et al.*, 2006) was applied to the fire reference severity on the organic layer to determine the LSF or HSF, in each forest type (appendix A, chapter 3). Predictive relationships and different yield curves were used to reflect moss dynamic and tree growth reduction in BSSP stand and higher site productivity in BSFM over time (Andrieux *et al.*, 2018;

Belleau *et al.*, 2011). We assumed that HSF led to higher site productivity than LSF (Lecomte *et al.*, 2006; Simard *et al.*, 2007). In BSSP, more prone to paludification, two possible successional trajectories were presented: a productive one after HSF, starting with productive mature stands that transitioned to open and less productive stands over the long term, as feathermoss was gradually replaced by *Sphagnum*, and a second, less productive trajectory after LSF, beginning with open mature stands already abundant in *Sphagnum*. In BSFM stands less prone to paludification, includes more productive stands that remained productive, where feathermoss was not replaced by *Sphagnum* spp. over the simulation period. We used soil parameters (Q10, decay rate constant) based on published literature to reflect the thickness of organic soil layer in BSSP compared to BSFM stands. A regeneration delay of 10 years after fire was applied in fire scenario (Zoltai *et al.*, 1998). Natural fire scenario was already explored in our previous study (Botroh *et al*,2024, not yet published).

Harvest CPRS. CPRS is the clearcutting with 97% of the merchantable volume removing and transferred for wood product. The remaining tree biomass whereas foliage, branches, and coarse and fine roots were left on site and transferred to sol pool. Moss ground cover and organic soil layer were less impacted. This moss live portion was transferred in upper organic layer of soil pool. For instance, CPRS reduce 60% of living feathermoss cover and 50% of *Sphagnum* cover and only 20% of uppermost organic layer (Kpodo and Thiffault, 2014; Lafleur *et al.*, 2016; Mayer *et al.*, 2020). C proportion impacted was mentioned in appendix A, chapter 3. In uppermost organic layer, CPRS accelerates decomposition rate in the early-stage stand (up to 15 years) but at a lower level than fire scenario (appendix A, chapter 3). Here, although disturbance severity may vary by forest type, as seen in fire scenarios, all harvested scenarios maintained the same impact magnitude across BSSP and BSFM stands. Because of this little impact on the soil organic layer and the understory vegetation, we assumed that CPRS were closer to the LSF tree and moss growth dynamic. It maintained stand to the lower initial productivity state in BSSP and relatively higher site productivity in BSFM. A regeneration delay was not applied.

Harvest PC. In the partial cut scenario, we tested 40% of the merchantable volume removing. Based on previous studies in the clay belt (Fenton and Bergeron 2007), lower proportion of moss live and organic soil layer were reduced in PC compare to CPRS (Fenton and Bergeron 2007) (see appendix A, chapter 3). The moss live portion affected was transferred in upper organic layer. In uppermost organic layer, PC accelerates decomposition rate in the early stage stand at the same level than CPRS on the area disturbed (Tableau 12, see appendix A, chapter 3). We assumed that PC were closer to the LSF tree and moss growth dynamics like CPRS, and did not improve tree growth (Moussaoui *et al.*, 2019; Roy Proulx *et al.*, 2021). A regeneration delay was not applied.

Mechanical site preparation MSP + plantation . Clearcutting followed MSP impacts moss live ground, organic layer and wood debris. The upper organic layer (acrotelme) was much more reduced than harvesting only based on the previous studies (Botroh *et al*, 2023, Trettin *et al*, 1992, 2011). We used an average of the C disturbed proportion of several MSP technique, such as disc trenching and plow, provided in Kpodo and Thiffault (2014), Trettin *et al.* (1992), Mayer *et al.* (2020) (see appendix A, chapter 3). The moss live proportion reduced was transferred in upper organic layer and the proportion of upper layer affected was transferred in underlayer. Decomposition rates were accelerated both in upper and deeper organic layer at different rates in disturbed year (Trettin *et al.*, 1992) and the following years (Botroh *et al*, 2023), at the early stage stand followed disturbance (15 years). Wood debris was mixed, and a proportion was transferred in the deeper soil layer. Also, to represent mixed soil layer effect of mechanical site preparation, we transferred also some proportion of deeper layer in uppermost layer. These adjustments aimed to simulate the 30% C loss in the upper organic layer observed in first year followed MSP in previous studies (Mayer *et al.*, 2020; Trettin *et al.*, 1992). We assumed that mechanical site preparation with plantation brings higher site productivity and promotes tree and moss growth patterns closer to those observed under HSF. However, we assumed that tree growth in MSP + plantation remains somewhat relatively lower than in HSF, though higher than that in CPRS stand. For example, MSP with conifer plantation increased tree height by approximately 15–25% compared to the harvesting only (Mjöfors *et al.*, 2017; Sikström *et al.*, 2020). Then, in BSSP initiated by LSF, yield curve chosen for MSP+plantation exceeded the lower initial site productivity, while in BSFM stands initiated by HSF, the yield curve fell below the productive initial productivity level. A regeneration delay was not applied.

Prescribed burning (PB) . Clearcutting followed PB consumes vegetation (tree and bryophytes) and upper organic soil (appendix A, chapter 3). However, the impact of PB on organic soil layer was lower than HSF (Lafleur *et al.*, 2016; Nave *et al.*, 2010) and snags proportion is more reduced in PB because they are exported for forest product. We assumed that PB with plantation brings higher site productivity and promotes tree and moss growth patterns closer to those observed under HSF. PB with conifer plantation increased tree height by approximately 12,5 % compared to the harvesting only (Renard *et al.*, 2016). PB in BSSP brings lower stand productive dominated by Sphagnum to the productive stand at the mature state while in BSFM stand remains productive dominated by feathermoss. A regeneration delay was not applied.

Tableau 12
Summary of main assumption about disturbance effect on C pool and process.

Pool	Fire	CPRS	MSP+plantation	PB+plantation	PC	Reference
Biomass Tree growth and moss	-Tree growth higher in HSF than LSF Regeneration delay -Sphagnum cover higher in HSF than LSF	-Same growth than LSF No delay regeneration -Same moss dynamic than LSF	-Higher than CPRS MSP= 1,22*CPRS No delay regeneration -Same moss dynamic than HSF	-Higher than CPRS PB= 1,12 *CPRS No delay regeneration Same moss dynamic than HSF	-Same growth than LSF No delay regeneration -Same moss dynamic than LSF	Lafleur *et al.*, 2016; Moussaoui *et al.*,2019; Mjöfors *et al.*, 2017 ; Proulx *et al.*, 2019; Renard *et al.*, 2016.
	Tree growth CPRS=PC=LSF<PB<MSP<HSF					
Organic layer	Uppermost Decomposition ++ LSF<HSF	Uppermost lower impact Decomposition +	Uppermost higher impact Deeper lower impact Decomposition +++	Uppermost higher impact Decomposition ++ but 26% less of HSF	Uppermost lower impact Decomposition +	Botroh *et al.*, 2023; Kpodo and Thiffault, 2014; O'Neill *et al.,* (2006); Renard *et al.*, 2016 ; Trettin,*et al.*, 1992
	Decomposition at short term PC<CPRS<LSF<PB<HSF<MSP					

LSF: low severity fire

HSF: high severity fire

CPRS: cut with protection of regeneration and soils

MSP: mechanical site preparation

PB: prescribed burning

PC: partial cut

Landscape level. At the landscape level, we simulated four scenarios that include the most common disturbances, as following: (1) the baseline scenario is long fire cycle of 400 years (400-year cycle, current cycle), (2) current management dominated by CPRS with lower MSP proportion (Business as usual management, 99% CPRS with 50% MSP and 1% PC), (3) alternative management with the increasing of MSP (MSP-alternative, 100% CPRS were followed by MSP), (4) alternative UAF8551 management with the increasing of PC (PC-alternative, 100% PC) (Tableau 11). The 400-year fire cycle was simulated by randomly burning on average 0.25% of total landscape area per year (see Botroh *et al*. unpublished for more details). In the management scenario, an annual harvest rate of 0.9% was used, and the mature or oldest stands were first harvesting. Management scenarios showed different levels of annual proportion of MSP and PC compared to the current management scenario as mentioned in Tableau 11. All Scenarios were tested on the same initial landscape condition. At the initial simulation conditions (time step = 0), the landscape dominated by oldest ages classes (120 + years) was characterized by 82% of BSSP more prone to paludification and 18% of BSFM (Tableau 10). 82% of the landscape was initiated LSF, while 18% of the landscape with HSF.

3.5 Results

3.5.1 Stand level

Over the 250-year simulation, in BSSP, C stocks (excluding mineral soil) increased in all scenarios, with the greatest accumulation in CPRS compared to the other scenarios (Figure 18 a). Initial C stocks of 224 Mg C ha^{-1} increased respectively to: 252 Mg C ha^{-1} in CPRS, 249 Mg C ha^{-1} in HSF, 247 Mg C ha^{-1} in both LSF and MSP, 238 Mg C ha^{-1} in PB and 231 Mg C ha^{-1} in PC. In BSFM, they increased at a lower rate than BSSP with slight accumulation in HSF compared to the other scenarios: from 194 Mg C ha^{-1} (initial stocks) to 195 Mg C ha^{-1} in HSF, 186 Mg C ha^{-1} in MSP, 180 Mg C ha^{-1} in LSF, 179 Mg C ha^{-1} in PB, 173 Mg C ha^{-1} in CPRS, and 125 Mg C ha^{-1} in PC.

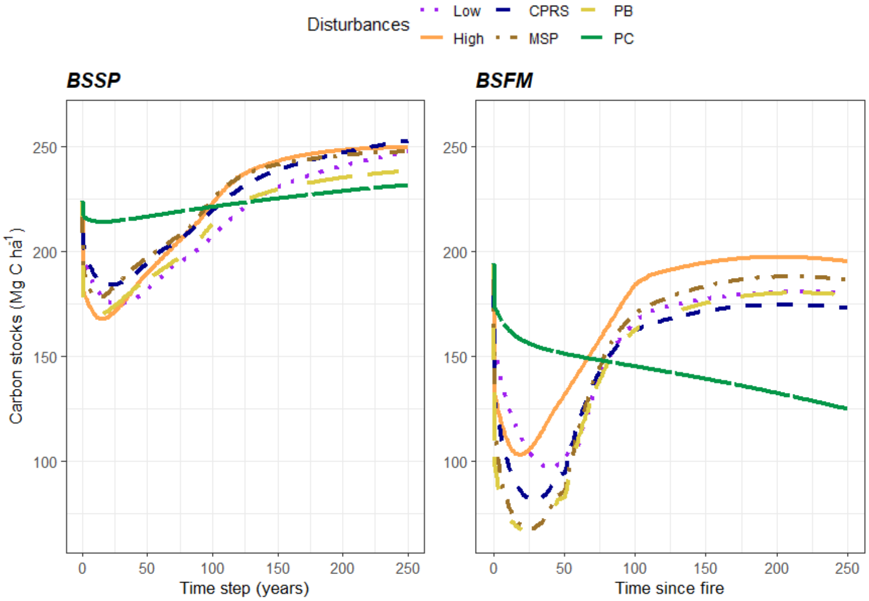
Immediately after disturbance, total C stocks showed greatest reductions after PB and HSF respectively compared to the other scenarios (in the order: PB > HSF > MSP > LSF > CPRS > PC) in BSSP. MSP was intermediate. In contrast, in BSFM, PB and

MSP caused the largest reductions in total ecosystem C (PB ≥ MSP > CPRS > HSF > LSF > PC). During the first 100 years following disturbance, in BSSP, total ecosystem C was highest in PC, followed by MSP, CPRS, and HSF, which were relatively close, while PB and LSF resulted in lower C stocks (PC>MSP≥CPRS≥HSF>PB≥LSF). Thereafter, total ecosystem C became the lowest in PC across all scenarios and HSF and MSP the highest due to the tree dynamic. At the end of simulation, total C stocks after CPRS tended to surpass HSF and MSP scenarios due to organic layer (Figure 18 a). While the reduction in C storage in PC is due reduction in the tree component, C storage the C lost in PB is mostly from the organic layer. In BSFM, during the first phase after disturbance (approximately 60 years), PC had the highest total ecosystem C, followed by HSF, LSF, MSP, CPRS, and PB remained closer together in terms of C stocks over the first 60 years of the simulation. However, after this period, HSF had the highest total C stocks, followed by MSP, while PC had the lowest. PB, similar to LSF, was closer to CPRS. Over the entire simulation period, the total C stocks of PC declined toward the end of the simulation period, while it remained steady with HSF due to tree biomass dynamics (Figure 18 a, b). Overall, despite that the total C dynamics in BSSP did not exhibit a clear trend due to the trade off between tree biomass and moss derived stocks, a lower disturbance level of the organic layer led to the highest total stocks. In contrast, in BSFM, severe disturbances promoting C accumulation in tree biomass resulted in greater C storage. Faster recovery in living biomass was observed as our simulation assumption in HSF, MSP, and PB while LSF and CPRS exhibited similar patterns independently of forest type (Figure 18 b). Despite having the largest C stocks in the early stages of simulation, PC showed the slowest recovery over the simulation period due to the tree mortality.

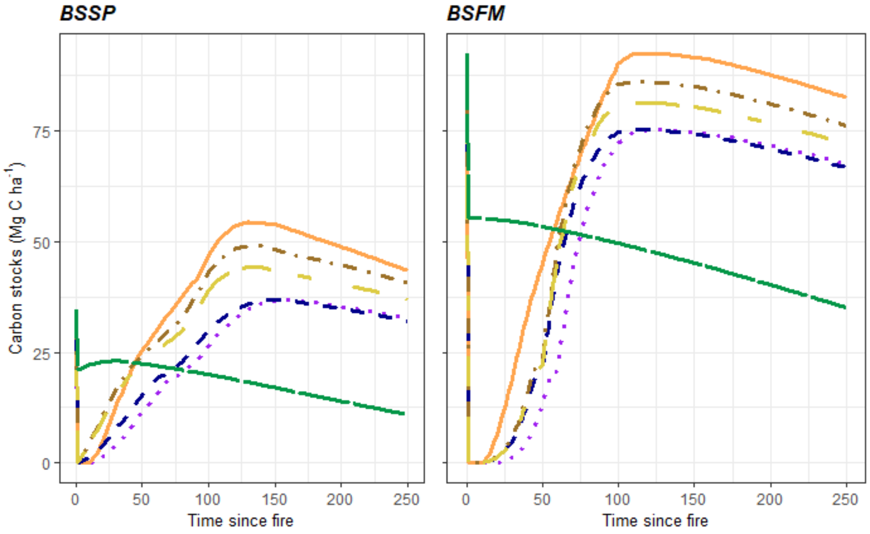
Regarding organic soil layer C stocks, fire scenarios (PB, HSF and LSF respectively) exhibited a greater reduction than in other disturbance scenarios immediately after disturbance independently of forest type (Figure 18 d). However, over the simulation period, in BSSP, organic soil C stocks increased in all scenarios at the higher rate than BSFM. In BSSP, PC showed the greater accumulation in organic layer stocks followed by CPRS, LSF, MSP, HSF, and PB (PC>CPRS>LSF>MSP>HSF>PB, *Figure 18*, appendix B, chapter 3). PC and CPRS and LSF reached the pre-disturbance soil layer stocks level in the first century (25 years for PC, 75 years in CPRS and 100 years about for LSF) while MSP, PB, and HSF reached the pre-disturbance soil layer stocks at 200 years about (Figure 18 d). In BSFM, organic soil C stocks increased much more in LSF, followed by HSF, MSP, CPRS, PB and PC (LSF > HSF > MSP > CPRS > PB > PC). The higher rate in LSF in BSFM is due to the relatively higher promotion of sphagnum species cover than HSF. Organic soil C stocks converged from150 years for all scenarios. PC had the lowest organic soil C at the end of the simulation in BSFM stands. Contrary to the organic layer pool, the dead organic matter (DOM, snag, and debris) C pool was more reduced in the sylvicultural scenarios (MSP, PB, and CPRS) compared to wildfire scenarios (Figure 18 c).

Immediately after disturbance, all scenarios shifted to net C sources (Tableau 13). Disturbance effects were more pronounced in BSFM stands than BSSP stands. The smallest source was observed in PC (-6 Mg C ha^{-1} $year^{-1}$ in BSSP and -18 Mg C ha^{-1} $year^{-1}$ in BSFM) compared to other scenarios. The largest C sources were found in PB (-89 Mg C ha^{-1} $year^{-1}$) and MSP (-65 Mg C ha^{-1} $year^{-1}$) in BSFM, and in PB (-46 Mg C ha^{-1} $year^{-1}$) and HSF (-41 Mg C ha^{-1} $year^{-1}$) in BSSP (Tableau 13).MSP in BSSP was intermediate (-27 Mg C ha^{-1} $year^{-1}$). Thereafter, the stands transitioned to C sinks more quickly in PB (14 years), MSP (15 years), HSF (17 years), and PC (17 years) compared to CPRS (26 years) and LSF (30 years) in BSSP (Table). In BSFM, the stands also became C sinks more rapidly with HSF (21 years), MSP (28 years), and PB (29 years), compared to CPRS (31 years) and LSF (42 years), largely due to faster tree growth. In BSFM, PC remained a weak C source because the moss component was insufficient to offset C losses from trees in BSFM compared to BSSP. This result highlights the significant contribution of moss to C sequestration in BSSP stand especially in PC.

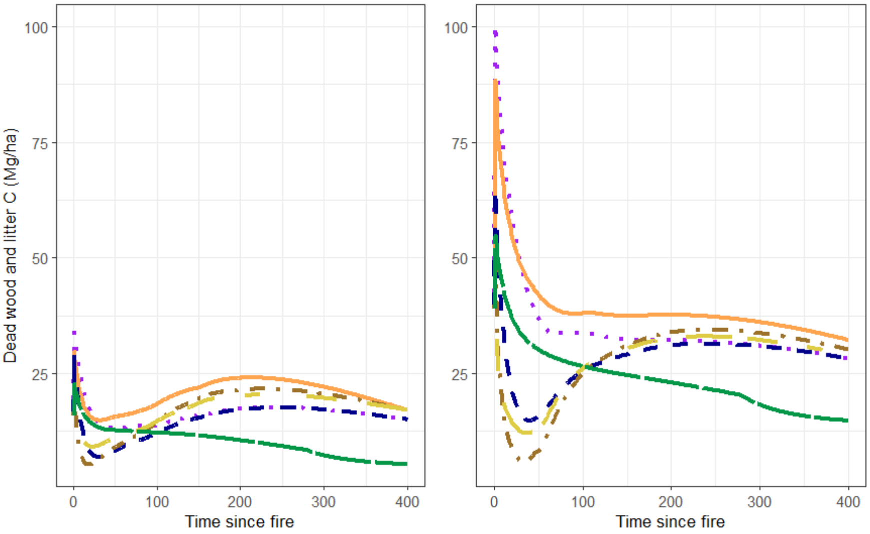
a) Total carbon stocks (excluding the mineral soil) per disturbances and stand type



b) Living biomass per disturbances and stand type



c) Deadwood per disturbances and stand type



d) Organic layer per disturbances and stand type

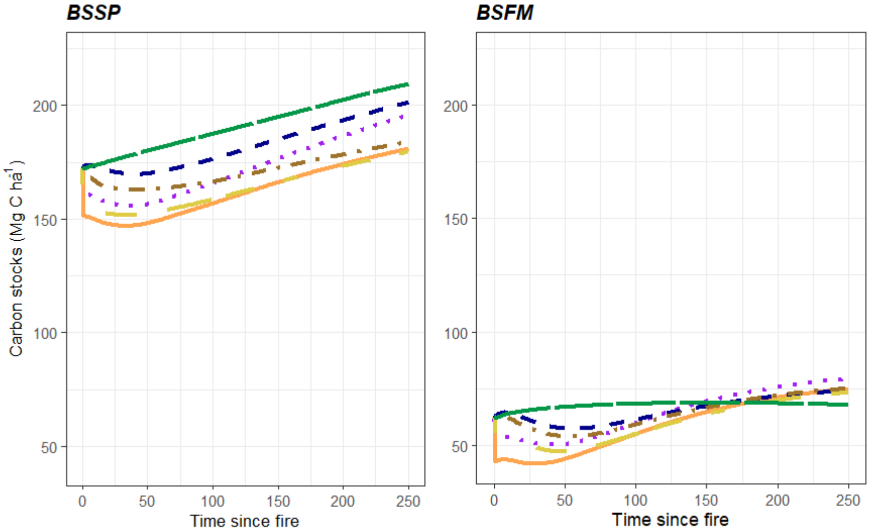


Figure 18

a)Stand level total ecosystem carbon stocks (Mg C ha^{-1} , live biomass + litter + debris+ dead wood + organic soil layer) ; b) live biomass, c)deadwood, d) organic layer evolution with time since disturbance (250 years) for the two stand types (BSSP and BSFM) and the five disturbance scenario (LSF, HSF, CPRS, MSP, PC). HSF in orange, LSF in purple, CPRS in blue, MSP +plantation in brown, PB +plantation in yellow, PC in green. LSF: low severity fire ; HSF: high severity fire; CPRS: cut with protection of regeneration and soils; MSP: mechanical site preparation; PB: prescribed burning; PC: partial cut; BSSP: black spruce-Sphagnum dominated stands; BSFM: black spruce-feathermoss dominated stands.

Tableau 13
Summary of total carbon balance (Mg C ha-1 year-1 = 100g C m^{-2} y^{-1}) per disturbance scenario: carbon balance at disturbance year, the first positive value and the delay to switch to a sink for all tree compartment and moss compartment (live biomass and moss-derived C) contributions.

Disturbance	at disturbance year	time step to switch (years)	sink
BSSP stand			
HSF	**-40.699**	17	0.042
LSF	-18.617	30	0.011
CPRS	-18.59	26	0.016
MSP	-27.479	15	0.048
PB	**-46.488**	14	0.022
PC	-5.913	17	0.033
BSFM stand			
LSF	-34.289	42	0.03
HSF	-55.925	21	0.041
CPRS	-58.969	31	0.001
MSP	**-64.995**	28	0.004
PB	**-89.396**	29	0.053
PC	-18.038	NA	-0.09 at the end closer to neutral

LSF: low severity fire
HSF: high severity fire
CPRS: cut with protection of regeneration and soils
MSP: mechanical site preparation
PB: prescribed burning
PC: partial cut
BSSP: black spruce-*Sphagnum* dominated stands
BSFM: black spruce-feathermoss dominated stands

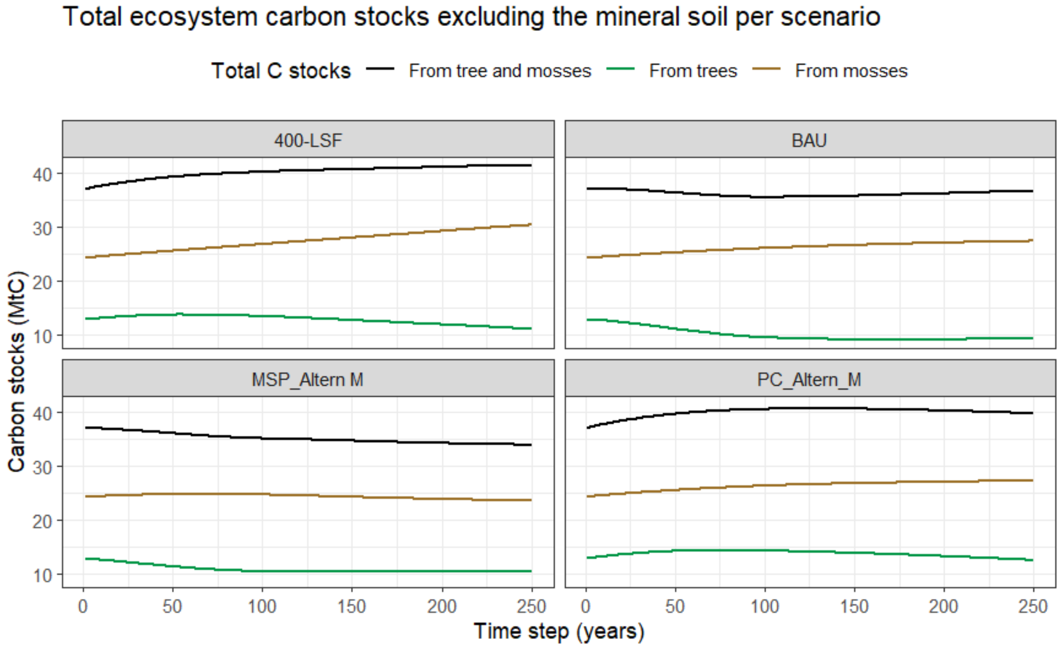
3.5.2 Landscape level

At the landscape level, total C stocks excluding mineral soil were higher in the fire scenario (current fire cycle) than in the managed scenarios (Tableau 14 ,Figure 19a). Over the 250-year simulated period, total C stocks increased from 37 MtC (initial landscape C stock) to 41 M tC in current fire, to 38.7 in PC-alternative management. In contrast, they decreased to 33.1 MtC under the MSP-alternative management scenario and to 36.6 MtC under the business as usual (BAU) management scenario. This trend is primarily driven by C storage in the organic soil layer rather than in tree biomass (Figure 19b, c).

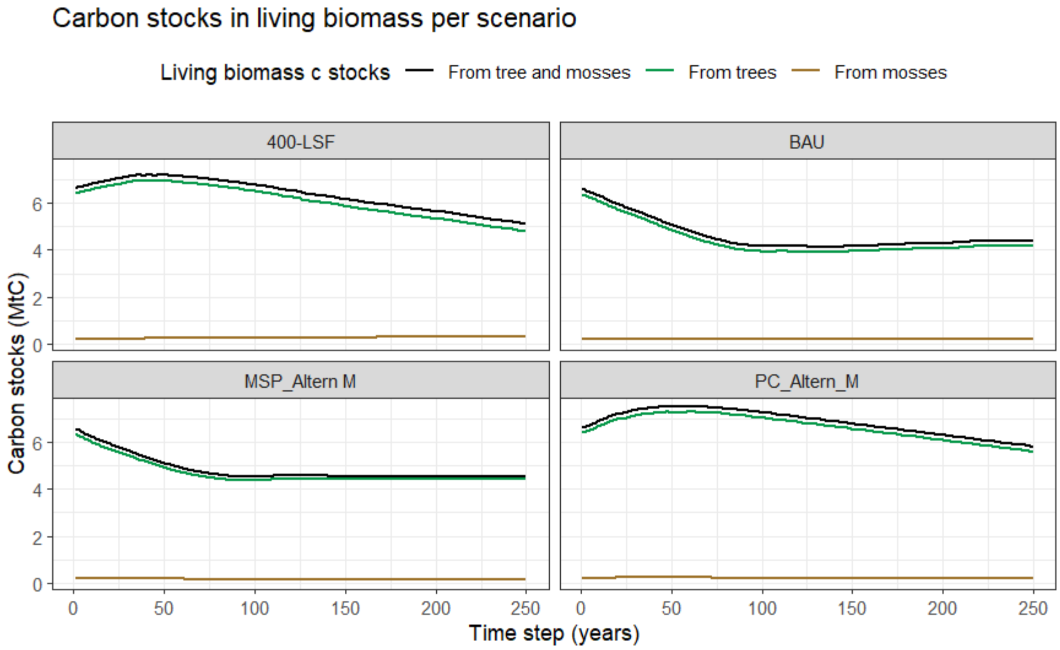
Forests remained a C sink along the 400-year fire cycle scenario through the 250-year simulation period compared to management scenario. In the 400-year fire cycle scenario, the landscape was a strong sink for the first 100 years, then it converged to an almost neutral C balance for the remaining period. However, the greatest sink was found in CP-alternative management scenario at the early stage of simulation due to a greater tree biomass. Thereafter, the landscape transitioned to a source until 200 years. The landscape was a large source immediately after treatment in MSP-alternative management and BAU respectively. Until 100 years, it switched to being a sink both in BAU and MSP-alternative scenarios. PC-management remained the greatest sink for the remaining simulation period while MSP-alternative MSP remained closer to neutral at the end of simulation (Figure 20). 400-year fire cycle scenario exhibited the greatest C emissions to the atmosphere than management scenario (see appendix B, chapter 3). However, MSP-alternative management showed in average relatively lowest C emission than usual management and PC-alternative management due to the thin C stocks stored in the ecosystem (Figure 19c, appendix B, chapter 3).

C storage and balance at the landscape level were linked to the class age distribution. Initially, the landscape was dominated by older age classes (120+ years) dominated the landscape at the initial conditions. By the end of simulation, MSP-alternative management and the BAU scenario resulted in a higher proportion of younger age classes (<120 years). In contrast, PC-alternative management and 400-year fire cycle maintained a greater proportion of oldest age classes (>120 years) (appendix B, chapter 3).

a)


Total ecosystem carbon stocks excluding the mineral soil per scenario
Total C stocks
From tree and mosses
From trees
From mosses
400-LSF
BAU
MSP_Altern M
PC_Altern_M
Carbon stocks (MtC)
10
20
30
40
0
50
100
150
200
250
Time step (years)

b)


Carbon stocks in living biomass per scenario
Living biomass c stocks
From tree and mosses
From trees
From mosses
400-LSF
BAU
MSP_Altern M
PC_Altern_M
Carbon stocks (MtC)
0
2
4
6
0
50
100
150
200
250
Time step (years)

c)

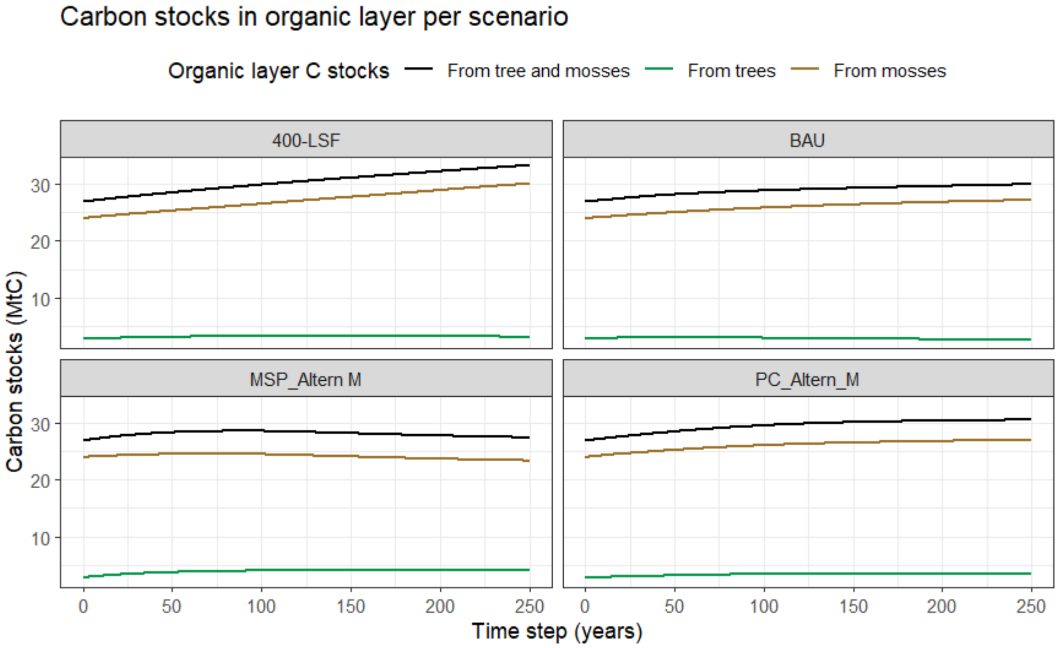


Figure 19
Landscape simulated carbon stocks dynamic (MtC; 1012 gC or 106 tC, without mineral soil) for 400-year fire cycle and the two management scenarios (BAU current or business as usual, MSP-alternative and PC-alternative): a) Total ecosystem C stocks excluding the mineral soil (live biomass + litter + debris+ dead wood + organic soil layer), b) living biomass C stocks (from trees, from mosses) and c) organic layer (moss-derived and from trees). C stocks from tree in green, C stocks from moss in brown, and total (moss + tree) in black.

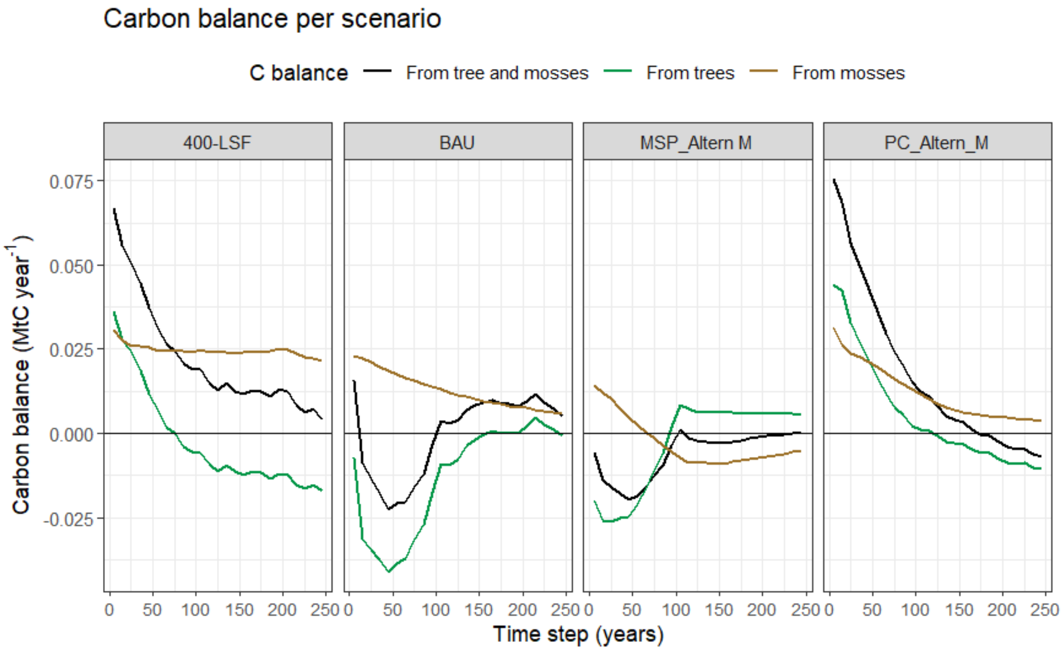


Figure 20
The landscape carbon balance over 250-year simulation period (MtC year^{-1}; 10^{12} gC or 10^{6} tC by year) in 400-year fire cycle scenario and in three management scenarios (BAU current or business as usual, MSP-alternative and PC-alternative): tree contribution in green, moss in brown, and combined moss and tree in black. The annual carbon balance values were smoothed averages over 10 years.

Tableau 14
Landscape total carbon stocks (MtC; 10^{12} gC or 10^{6} tC, without mineral soil) and differences between the last year of simulation and the first year of simulation per disturbance scenario on 250 year-simulation period for 400-year fire cycle and three management scenarios (BAU current or business as usual, MSP-alternative and PC-alternative). Ratio in percent of the difference between final and initial values relative to the initial value in percentage. SD standard deviation.

Scenario	**Average**	**SD**	**Ratio (%)**
400-year fire cycle	40.703	1.186	14.523
BAU management	37.607	0.337	0.756
MSP-alternative	34.433	1.059	-10.374
PC-alternative	39.578	0.765	4.969

3.6 Discussion

3.6.1 Stand level

Our study explores the effect of various silvicultural practices on C trends in boreal forested peatland and compared them with a scenario of forests impacted by wildfires. Total trend in C stock changes varied according to disturbance type but also to forest type. In BSFM stands, severe disturbances events that promote tree growth resulted in faster C accumulation in biomass and to higher overall C storage. On the opposite, in the BSSP stands, no distinct pathways were clearly observed due to a tradeoff between tree and moss biomass accumulation over the 250 years simulation period. The main result indicated that a lower disturbance severity level, such as CPRS, led to the greater C accumulation over the simulation period in BSSP due to peat accumulation. We found that peat accumulation was responsible for the rapid recovery of the pre-harvested soil layer stocks after PC and CPRS in BSSP stand, in less than 100 years. This finding is consistent with some previous studies in paludified stand (Lafleur *et al.*, 2016; Lavoie *et al.*, 2005; Lavoie *et al.*, 2005). CPRS and PC favor the accumulation of a thick layer and accelerated paludification. In fact, CPRS and PC in paludified stand conserved Sphagnum moss and enhanced sphagnum productivity by increasing light and water table level after harvested events (Fenton and Bergeron 2007; Fenton *et al.*, 2010; Lafleur *et al.*, 2010; Lavoie *et al.*, 2005). In our simulation, PC led to the higher C accumulation in organic layer than CPRS due to the lower area disturbed. However, some studies suggested that the relatively closed forest canopy in PC would reduce the proliferation of *Sphagnum* mosses compared to CPRS and would accumulate peat at a lower rate than CPRS (Fenton *et al.*, 2009; Lavoie *et al.*, 2005). In our simulation, a greater long-term C accumulation in peat after PC in BSSP stands compensated for the reduction in C accumulation in tree biomass while this trend was not observed for BSFM stands. This finding suggests a greater contribution of sphagnum species in the C sequestration in site with low tree productivity (Bonan and Shugart, 1989; Zoltai and Martikainen, 1996). A greater C storage in PC due to the remaining tree compared to stand-replacing disturbances is well documented (Lee *et al.*, 2002; Peng *et al.*, 2002; Taylor *et al.*, 2008). In our simulation, PC showed the greatest C storage in the early phase of the simulation, but thereafter C storage decreased due to reduced growth rate of the remaining old trees (Ameray *et al.*, 2021; Audet-Giroux, 2021). In the long term, stand growth declined significantly because the stand was harvested at 121 years of age, which corresponds to a period of negative biomass increment according to yield curves (see Appendix C, chapter 3). Our simulation assumptions do not promote tree growth following PC. The benefit of PC for tree growth is not clear and depends on pre-harvest structure and site characteristics. For instance, PC was found to increase post-harvested black spruce forest productivity on site prone to paludification if the organic layer is thin, under 17 cm (Moussaoui *et al.*, 2019). Moreover, the regrowth of removed trees is not accounted for in the CBM-SCF3 model, potentially leading to an underestimation of biomass carbon during the early phase of the simulation.

Furthermore, we found that intensive disturbance led to a greater C loss in the short term but to a rapid recovery of the C balance due to enhanced tree growth. PB showed the greatest C loss due to the cumulative negative effects of total harvesting, temporary C reduction by decomposition and direct emission from fire, compared to the unburned silvicultural practices and natural fire scenario (Magnani *et al.*, 2007; Mayer *et al.*, 2020). Furthermore, in our study, we found that the magnitude of C loss after disturbance was greater in BSFM than BSSP. Initial C stock size and forest type had the strongest influence on the magnitude of disturbance C loss (Mayer et al., 2023). The intermediate impact of MSP in BSSP compared to BSFM observed in our study suggests that paludified stands are more resistant to intensive mechanical silvicultural practices. While MSP stimulated decomposition more than burning, it left on BSSP site a greater proportion of the disturbed organic layer by mixing and burying it into deeper layers. In contrast, fire scenarios (PB, HSF, and LSF) released a proportion of the disturbed organic layer into the atmosphere through direct emissions, unlike unburned disturbances. These findings support the view that severe disturbances, such as fire or other methods that physically remove the organic soil layer, may significantly reduce total C stocks in paludified stands.

The recovery of C loss following an intensive disturbance was faster in BSSP (e.g. 14, 15, 17 years in PB, MSP and HSF respectively) than in BSFM due to a faster recovery of tree growth which underscores the importance of the advanced growth of plantation or natural regeneration. The rapid increased in living tree biomass after severe disturbance (HSF, MSP, PB) reflects the well-known positive effect of the reduction of the organic layer thickness on tree growth on paludified sites (Lafleur *et al.*, 2016; Renard *et al.*, 2016). However, the success of plantation can sometimes be overestimated. Barrette *et al.* (2024) showed that in black spruce plantation, realized yield was consistently lower (-60%) than anticipated yield projected by models for ages 26–34. In our study, we applied a tree growth rate that is an average of the ones found with natural disturbance and with silvicultural practices on sites with similar conditions (Lafleur *et al.*, 2016; Mjöfors *et al.*, 2017). Similarly to the effect of a severe wildfire, rapid recovery was also found with PC (17 years) in BSSP stands while BSFM stands after PC remained a source. The growth of the remaining moss cover rapidly compensated for the early losses following PC in BSSP. Although the post-PC growth of removed trees is not accounted for, this finding suggests that moss-derived C stocks play a significant role in C recovery within these forests when C sequestration from trees is reduced.

Delay to switch from source to sink after disturbance was complex. It varied according to magnitude of disturbance, to the initial state, to the forest regrowth capacity and to methods used. Nevertheless, our findings are in line with previous modelling studies that modelled the evolution of C stocks with time since harvesting (12-26 years in Paradis *et al.,* 2019; 20 years in Kurz *et al.,* 1999). Gaboury (2006) found that black spruce forest plantations switch from source to sink at 27 years, but their study did not include mosses. Nevertheless, Korkiakoski *et al.* (2023) study indicated a very rapid recovery within five years after harvesting using eddy covariance and an even more rapid recovery with PC (2-3 years) for drained forested peatland.

3.6.2 Landscape level

At the landscape level, we tested the effect of increasing the proportion of an intensive silviculture practice (MSP), as well as that of a low intensity disturbance (PC) that we compare to the current management scenario (BAU). BAU includes a lower proportion of those practices, MSP and PC, for the same annual harvested rate. Because our region is dominated by sites prone to paludification (82% of BSSP stands and 18% BSFM), the trend in C storage considerably impacts the dynamics of the organic layer. We found that the C storage increased by 15% and 5% respectively by the end of the simulation period with the current fire scenario and with the PC-alternative management while it decreased by -10% with the MSP-alternative management scenario and - 1% with BAU. This finding supports the results of previous studies that showed that landscapes that are under a natural disturbance (fire) regime leeds to increased total C stocks compared to the management BAU scenario. For instance, Boisvenue *et al.* (2014), studying the same area as our study showed that a 400-year fire cycle increased the total ecosystem by 5% while even age management with rate of 1% of clearcutting decreased C stocks by -17%. The different results between this study and ours can be attributed to the inclusion of the moss C dynamics in our study, because the area considered, and the harvest rates were similar. C storage is lowest in even-aged management (BAU and MSP alternative management scenarios) because these practices rejuvenate the landscape more than the fire and the PC scenarios do (Ameray *et al.*, 2021; Kellomäki *et al.*, 2023; Paradis *et al.*, 2019; Peng *et al.*, 2002). The lower intensity of harvest in PC promotes a greater proportion of older stands in the landscape as does the natural disturbance regime scenario. Older stands store more C compared to younger stands because they have a large C storage within the dead organic matter pools (wood and moss) and a higher proportion in peat. The greatest reduction in C storage in MSP-alternative management compared to BAU, despite a similar proportion of younger stands (appendix C, chapter 3), indicates that this pattern is due to a reduction in *Sphagnum* abundance in MSP-management scenario. However, the fast growth of young stands contributed to maintain the long-term C balance in even-management scenario (BAU and MSP-alternative scenarios). Boisvenue *et al*, 2014 showed that postharvest emissions can result in ten times increase in the C emissions surpassing those of a 100-year fire cycle. Furthermore, some studies showed a substantial benefit of even-aged forest management toward climate change mitigation when forest products substitute emission intensive products such as steel or concrete (Boisvenue *et al.*, 2014; Paradis *et al.*, 2019). Moreover, there is a greater sensitivity of forest ecosystem C stocks and flux to forest management under climate change (Ameray *et al.*, 2024; Valipour *et al.*, 2021). Valipour *et al.,* 2021, showed that tree harvesting decreased total ecosystem C stocks from between 7 and 36% under present climatic conditions and between 7–60% under changing climate, respectively, with greater reductions for shorter logging rotation lengths and greater landscape cutting intensities. Harvesting under climate change could increase biomass production and hence C storage by CO^2 fertilization, although studies point to the other direction (Girardin *et al.,* 2016) but it could also lead to the higher losses from of soil organic matter decomposition (Valipour *et al.*, 2021). However, in the context of increasing wildfire frequency under warner climate, some intensive management, such as PB, can mitigate wildfire risks and C emission compared to uneven age management (Marrs *et al.*, 2018). In fact, we found that C emission was lower in MSP-alternative scenario than in PC-management scenario supporting findings that emission depends on the amount of C stored in the ecosystem (Boisvenue *et al.*, 2014; Paradis *et al.*, 2019). Old forests with higher C stocks can release more C when disturbed, as opposed to younger ones. Then, intensive forest management may potentially reduce C emission in the context of increased levels of disturbances.

3.6.3 Model limitations and perspectives

Our study explores the effects of silvicultural practices on C dynamics black spruce stand including peat and moss using modelling approach. Despite some variation in specific estimates, the overall trends in C dynamics following disturbances were consistent with previous studies and offered additional insights into the direction of these effects. However, our simulation may have limitations due to various assumptions about forest type conditions, tree growth, and disturbances. For instance, the assumption regarding the success of MSP on tree growth, or the assignment of the same growth after LSF as after PC, and the using of averaged effects for silvicultural practices, may impact greatly the conclusion of model simulations. Moreover, the model does not incorporate the understory regrowth after PC, which represents another source of uncertainty. Forest exhibit large spatial variability, site conditions and response to disturbance within the same scale level. The success and the real effect of silvicultural practices depends on their intensities, the initial forest type characteristics and the methods (Ameray *et al.*, 2021; Gauthier and Vaillancourt, 2008). Incorporating spatial information and accounting for uncertainties related to the magnitude of disturbance impacts on soil and tree growth would improve the accuracy of C dynamics representation. C dynamics after PC can be improved by using alternative simulation approaches with shorter rotation lengths and by addressing the model's limitations regarding post-harvest tree growth. Because our stand dynamic approach, including underground vegetation, was based on tree growth, develops specifics yield curves according to natural disturbance and silvicultural practices could further improve our results. In our study, shifts in *Sphagnum* species composition following disturbance (Fenton and Bergeron, 2007) and the increased abundance of ericaceous shrubs (Lafleur *et al.*, 2016), which could significantly influence C balance recovery (Turetsky et al., 2010), were not included.

In future investigations, simulation included both climate change, fire and management in the same scenario could help improve simulation results. Despite that, the study was based on the previous empirical studies, the validation of model prediction can also increase the accuracy of model prediction. Nevertheless, our study highlighted the critical role of forest stand types and initial site conditions to access the effects of silvicultural disturbances on C stocks.

3.6.4 Implication

Forested peatlands of the Clay Belt area represent an important C reservoir and are also an important source of timber. Silvicultural practices aimed at improving forest productivity on sites prone to paludification have been evaluated (Henneb *et al.*, 2019; Lafleur *et al.*, 2018; Moussaoui *et al.*, 2019). Our study explored for the first time the short and long term impact of silvicultural practices on C storage in our study area.

Based on our simulations, severe disturbance that fosters tree growth had the greater total C storage in BSFM stand over the 250-year simulation period. In BSSP, no distinct pathways were observed due to the tradeoff between tree and moss derived peat. Overall, in BSSP, lower disturbance severity level on soil such as CPRS led to greater C storage over simulation due to the dominant role of moss derived C stocks. Despite the fostering of tree C biomass storage in BSSP, paludification is the main driver of soil C storage in BSSP. An interesting finding of our study is that the greater C loss with intensive disturbance in comparison with low intensity disturbance is rapidly offset by a rapid increase of the C storage in tree biomass. PB and MSP offset C lost in soil in the short term relatively quickly compared to the severe disturbance HSF, Much more in BSSP stand than BSFM stand. Our study suggests that our intensive silvicultural treatment with plantation is not accompanied by a greater C reduction in the organic layer. Moreover, we suggest that paludified stands (BSSP) are more resistant to intensive mechanical site preparation than BSFM. Consequently, in paludified stands, only severe wildfires that cause high losses in combustion may substantially reduce total C storage. On low productivity and highly paludified sites, moss contribution is important to compensate the reduction of sequestration from the tree component and can contribute to maintain an important C sink in paludified forest ecosystem. Intensive silvicultural practices may offer a valuable option in stands with relatively high productivity, while in very low-productivity sites, conservation strategies may be more appropriate. Our study stresses the importance of considering initial forest conditions and site types in managing forests for their multiple values including the provision of biomass, C sequestration and the conservation of biodiversity.

3.7 Acknowledgements

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CONCLUSION

La gestion durable des forêts doit prendre en compte certains aspects importants des services écosystémiques de la forêt, dont la séquestration du C forestier. Les forêts paludifiées ou susceptibles à la paludification dans la ceinture d'argile au nord-ouest du Québec sont une importante source d'approvisionnement de bois. Les opérations forestières doivent permettre de maintenir la productivité des forêts en limitant la paludification, mais cette dernière est un moteur important dans l'accumulation de C dans le sol. L'adoption des stratégies d'aménagement de moindre impact sur la séquestration de C passe aussi par une connaissance approfondie de la dynamique naturelle du C forestier. L'objectif de cette étude était donc de mieux comprendre l'effet des pratiques sylvicoles et des feux (principale perturbation naturelle) sur la dynamique de C en forêts boréales paludifiées. L'un des défis de cette étude était d'inclure la composante mousse. Nous avons concilié des approches empiriques (terrain) et de modélisation afin d'avoir une idée à court et long terme des effets des perturbations sur la productivité forestière, le stockage de C des arbres et des mousses, tant au niveau du peuplement qu'au niveau du paysage.

Dans le chapitre 1, nous avons trouvé que les feux peu sévères, en accord avec les anciennes études sur la dynamique de la biomasse forestière (Lecomte et al, 2006; Simard et al, 2007), ont favorisé l'augmentation des stocks totaux de C dans les peuplements dominés par les sphaignes (BSSP) en raison de l'accumulation de tourbe. Tandis que dans les peuplements dominés par les mousses hypnacées (BSFM) ce sont les feux sévères qui ont entrainé un stockage de C plus important en raison de la biomasse des arbres. La couche organique représentait le principal réservoir de C de l'écosystème dans le BSSP (70 % en moyenne) et une proportion beaucoup plus faible du total dans les peuplements du BSFM (24 % en moyenne). Un des apports de cette étude est qu'indépendamment du type de forêts, la croissance des arbres permet le passage rapide de la forêt de source à puit après perturbation, mais les mousses maintiennent les puits de C sur le long terme. Au niveau du paysage, cette étude confirme l'accumulation continue de C due à la paludification (Gauthier et al, 2006) dans les prochains siècles à moins d'avoir une augmentation de la fréquence des feux sévères. Cette étude suggère qu'une bonne caractérisation des conditions initiales et du type de forêt est essentielle pour mieux prédire les effets des perturbations sur la dynamique du C.

Les pertes de C du sol organique après les pratiques sylvicoles à travers la décomposition de la tourbe demeurent un des éléments les moins connus dans la simulation de la dynamique de C forestier (Lavoie *et al*., 2005; Mayer *et al*., 2016.) et dont l'estimation présente de nombreuses incertitudes (Bona et al, 2016). Le second chapitre nous a montrés que la décomposition, à travers la perte de masse des bryophytes à court terme, dépendait à la fois des traitements sylvicoles, des espèces de bryophytes et du type de couche du sol (fibrique, mésique, humique et minéral). Selon notre approche méthodologique, nous avons trouvé que la coupe suivie de la préparation mécanique du sol (CPRS+ PMS) accélère la décomposition des sphaignes dans la couche supérieure du sol, mais a un impact limité sur les stocks des horizons du sol profond et totaux de C comparé aux zones non coupées, tandis que la CPRS avait un effet intermédiaire. Nous avons également constaté que les stocks de C du sol dépendent aussi des stocks de C initiaux avant perturbations, ce qui rejoint la conclusion du chapitre 1 sur l'importance de la caractérisation des conditions initiales dans l'évaluation des dynamiques des stocks. Ce chapitre a permis de mettre en évidence les aspects à considérer dans la simulation des stocks de C. Une des contributions majeures de cette étude a permis d'avoir une idée de la magnitude des effets des coupes et des préparations mécaniques sur le C du sol dans la ceinture d'argile, ce qui a permis de confirmer par exemple l'importance de la bonne caractérisation des conditions initiales dans la modélisation des pratiques sylvicoles ainsi que l'effet limité à court terme des perturbations sur les stocks de C du sol organique.

Les résultats du chapitre 3 montrent que les pratiques sylvicoles de faible intensité comme les CPRS ont entrainé un stockage plus important que les feux naturels peu sévères dans les zones dominées par les sphaignes (BSSP). Par ailleurs, dans l'immédiat après perturbation, tandis que les pratiques sylvicoles intensives de brûlage dirigé et de préparation mécanique de sol (PB et MSP) entrainaient plus de pertes de C dans les BSFM; dans les BSSP, ce sont les perturbations sévères de feu (PB et le feu naturel sévère HSF) qui ont entrainé le plus de pertes. MSP était intermédiaire dans BSSP. Cela suggère que les BSSP sont plus résistantes à la préparation mécanique de terrain (chapitre 2), et que les perturbations qui consument le sol organique comme les feux sévères et les PB sont susceptibles de réduire considérablement les stocks de C (chapitre 1). Cependant les pratiques intensives suivies de plantation qui favorisent la croissance des arbres tout comme les feux sévères, ont permis le passage rapide de source à puit indépendamment des types de forêts. Par ailleurs, dans les sites à très faible productivité, la contribution des mousses est essentielle pour compenser les apports réduits de carbone C des arbres. Au niveau du paysage, les stratégies d'aménagement qui impliquent des taux élevés de pratiques sylvicoles intensives telles que le MSP entraine une baisse de stockage de C total, mais ont montré un puit de C relativement stable à cause des jeunes peuplements avec une productivité importante.

Trois conclusions principales ressortent de nos résultats :

-Les effets des perturbations peuvent être contrastés selon la dominance du site par les sphaignes ou les mousses hypnacées et le degré de paludification qui en résulte.

- Dans les zones susceptibles à la paludification, les perturbations intensives peuvent maintenir un stock de C important dans la biomasse aérienne au cours du premier siècle, mais au-delà ce sont les mousses qui favorisent la croissance des stocks face à la réduction de la productivité des arbres en raison de la paludification.

- Les perturbations intensives favorisent la séquestration de C dans les arbres, laquelle permet de compenser rapidement les pertes de C occasionnées comparées aux perturbations peu sévères.

- Les perturbations plus sévères et plus fréquentes sont celles qui peuvent compromettre l'accumulation de C total dans le paysage.

Les résultats de cette étude (Figure 21) apportent de nouvelles connaissances scientifiques pouvant avoir des implications pour l'aménagement forestier en zone paludifiée. Dans la ceinture d'argile, l'allongement du cycle de feu dans ces dernières décennies qui se traduit par une réduction de productivité forestière présente un enjeu en termes de services écosystémiques notamment la productivité de bois pour le développement socio-économique et la séquestration de C. Le paysage est donc une mosaïque de peuplements encore productifs, ceux en perte de productivité et ceux en transition, résultante de l'occurrence de feu sévère et peu sévère. Notre étude montre que les CPRS, qui sont la norme d'exploitation forestière au Québec, ne pourront pas maintenir la productivité forestière à long terme contrairement aux pratiques intensives, qui, elles peuvent en contrepartie entrainer une réduction de stockage de C du sol organique au niveau du paysage. Ainsi, les pratiques sylvicoles intensives peuvent répondre au besoin de séquestration C et d'approvisionnement en bois dans les peuplements encore productifs, tandis que dans les sites à très faible productivité, les stratégies de conservation pourraient être envisagées (Figure 22). Cependant, ces conclusions sont essentiellement portées sur le succès des pratiques sylvicoles intensives sur la croissance des arbres et la remise en production forestière effective à travers les plantations. Elles n'incluent pas les échecs susceptibles de ces pratiques mentionnées par de nombreuses études (Barette et al, 2024; Boiffin et al, 2014). Les opérations forestières intensives sont des opérations très couteuses. Ainsi, les aspects techniques et socio-économiques sont des questions qui méritent d'être prises en compte. De plus les opérations intensives soulèvent plusieurs interrogations écologiques. Les émissions de CO_2 liées aux opérations de récolte et préparation de terrain (Boisvenue et al, 2006, Paradis et al, 2016), la modification des habitats des espèces animales et végétales telle que la perte du bois mort sont des aspects qui méritent d'être évalués. Les recherches sur l'impact global des traitements sur les autres services écosystémiques telle que la biodiversité devraient fournir des perspectives supplémentaires sur la durabilité de la gestion forestière intensive dans ces milieux.

En somme, pour répondre aux besoins environnementaux et socio-économiques, les pratiques sylvicoles adaptées à la ceinture d'argile devraient combiner et alterner les pratiques à la fois sévères et peu sévères tout en ciblant les zones en fonction des caractéristiques des territoires selon la nécessité d'appliquer des pratiques intensives. Cela pourrait passer par une mise en place d'un système de suivi spatialisé des peuplements selon leur degré de paludification.

La dynamique de C en forêt paludifiée est un sujet très complexe. Les pistes de recherche futures sur l'évaluation de l'empreinte C de l'aménagement se trouvent à différents niveaux. Ce travail mentionne l'importance de caractérisation des conditions initiales et des types de peuplements dans l'évaluation de la dynamique de C à la suite des perturbations. Ainsi, la simulation directe des épaisseurs de la couche organique et des changements des conditions de sites et de végétation pourrait aider à mieux décrire la dynamique de séquestration des forêts en fonction de leur degré de paludification. L'établissement des courbes de croissances propres aux peuplements perturbés pourrait améliorer les précisions des estimations. Les travaux de mesure de terrain utilisant des méthodes plus adaptées telles que les mesures de flux par Eddy covariance ainsi que des taux de transfert de la végétation au sol, qui sont plus spécifiques aux traitements et à la zone, à court, moyen et long terme pourrait enrichir le paramétrage des modèles. L'inclusion des méthodes de spatialisation au travers de la télédétection et de l'apprentissage profond pourraient s'avérer utile pour la validation des résultats sur le terrain et orienter la planification des opérations forestières. Les travaux de Laamrani et al, 2015, Corredor et al, 2021 et ceux de *Cerrejon et al.*, 2020 peuvent constituer un bon potentiel dans l'amélioration de la représentation spatiale des stocks de C des forêts paludifiées. De plus, l''utilisation de la nouvelle version du modèle de simulation CaMP (Canadian peatlands, Bona et al, 2020 ; Shaw et al, 2016) plus spécifique aux tourbières et forêts entourbées et qui inclue la nappe phréatique pourrait être envisagée pour poursuivre et affiner les études de la dynamique de C dans la ceinture d'argile. Le changement climatique est susceptible de modifier profondément le devenir de C forestier, ainsi une projection future du devenir du C incluant à la fois le changement climatique, les feux et l'aménagement pourraient contribuer à définir une dynamique plus réaliste de la dynamique de C forestier dans la ceinture d'argile.

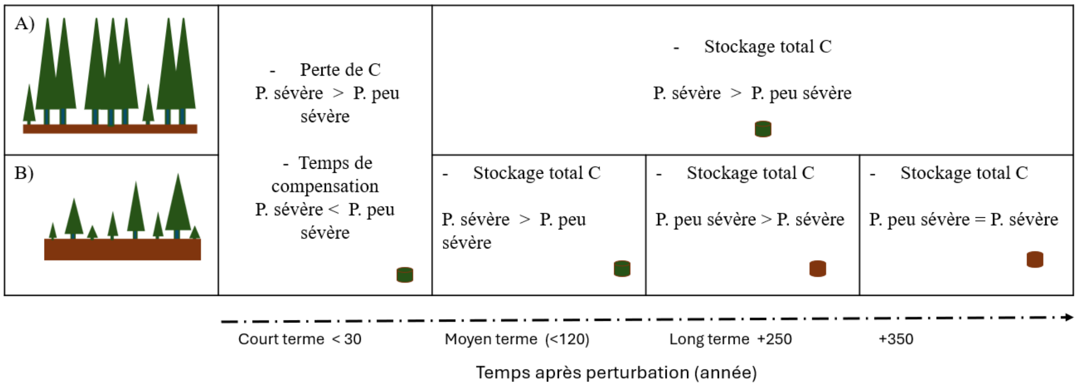


Figure 21 Résumé des résultats de la thèse. Effets des perturbations qui renouvellent le peuplement sur la dynamique de stockage de C dans 2 types de forêts. A et B : conditions initiales - type de forêts , A forêt d'épinette noire à dominance de mousses hypnacées productive avec une couche organique mince, B forêt d'épinette noire à dominance de sphaignes avec une couche organique épaisse, paludifiée ou plus encline à la paludification. P. pour perturbations. Les perturbations sévères naturelles (feu sévère) et sylvicoles (MSP, PB suivie de plantation) qui favorisent la croissance des arbres. Les perturbations peu sévères naturelles (feu peu sévère) et sylvicoles (CPRS). Le remplissage vert indique le compartiment de l'écosystème (arbres) qui domine le stockage totale, et marron pour la tourbe provenant essentiellement des mousses sphaignes. 🟩 🟫

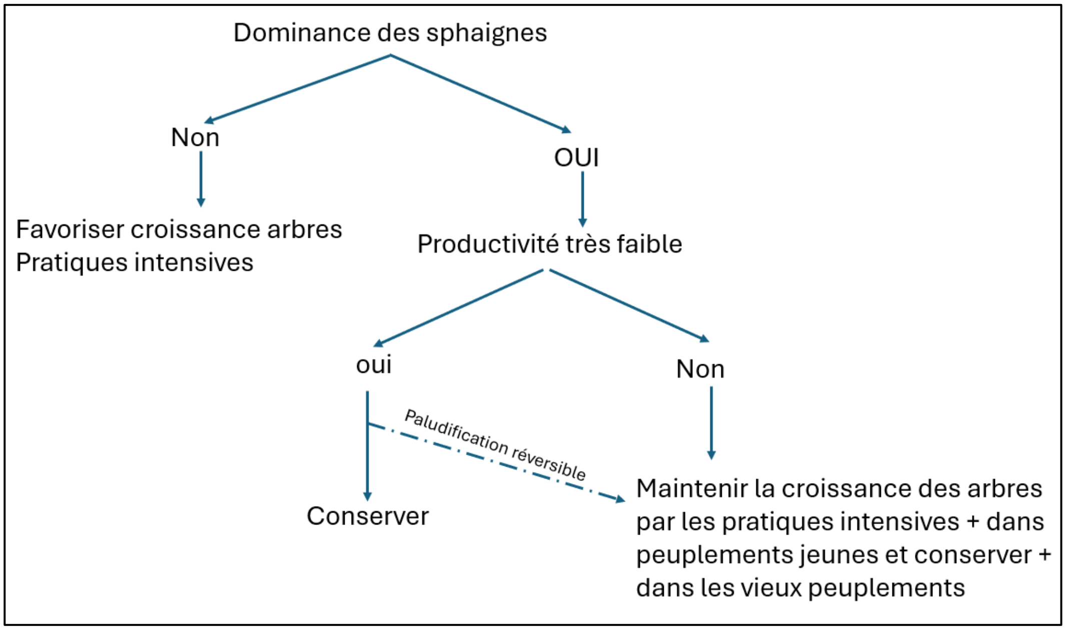


Figure 22 Arbre de décision pour le stockage de C en fonction de la dominance des sphaignes dans le sous-bois des forêts. Les pratiques sylvicoles intensives suivie de plantation (exemple les préparations mécaniques de terrain, le brûlage dirigé) peuvent répondre au besoin de séquestration C et d'approvisionnement en bois dans les peuplements encore productifs, tandis que la conservation peut être envisagée dans les sites à très faible productivité.

APPENDICE A – APPENDICES DU CHAPITRE 1

Appendix A: CBM description

Table A1: Description of carbon pools in the Carbon Budget Model of the Canadian Forest Sector 3-version 1.1 (CBM-CFS3) including the MOSS-C plug-in module. (Kurz et al, 1992; 2009 ; Bona et al, 2016).

Carbon pools	Description
Live biomass	
Merchantable	Live stemwood of merchantable size (DBH > 9.1 cm) + bark
Other wood	Live branches, stumps, and small trees + bark
Foliage	Live foliage
Fine roots	Live roots, approximately <5mm diameter
Coarse roots	Live roots, approximately ≥5mm diameter
Feather live	Feather moss live
Sphagnum live	sphagnum moss live
Dead organic matter (DOM) and soil layer	
Snag stems	Dead standing stemwood of merchantable size + bark
Snag branches	Dead branches, stumps, and small trees + bark
Medium	Coarse woody debris on the ground
Aboveground fast	Fine and small woody debris plus dead coarse roots in the forest floor, approximately ≥5 and <75mm diameter
Aboveground very fast	The L horizon comprised of foliar litter plus dead fine roots, approximately <5mm diameter
Aboveground slow	F, H and O horizons
Belowground fast	Dead coarse roots in the mineral soil, approximately ≥5 diameter
Belowground very fast	Dead fine roots in the mineral soil, approximately <5mm diameter
Belowground slow	Humified organic matter in the mineral soil
Feather fast	Feather moss dead organic matter in acrotelme
Sphagnum fast	sphagnum moss dead organic matter in acrotelme
Feather slow	Feather moss dead organic matter in catotelme
Sphagnum slow	sphagnum moss dead organic matter in catotelme

Appendix B: Disturbance matrices

Disturbance matrix CBM

Disturbance matrice defines the proportions of C transferred between C pools, atmosphere, and forest products sector following disturbance events. Fire matrices and parameters were calibrated using the spatial unit of the eastern boreal shield and modified to include fire severity. Fuel consumption for each pool was calibrated using national forest inventory, fire weather information, and the Boreal Fire Effects model (BORFIRE) (de Groot, 2006; Kurz et al., 2009). We assumed that the default CBM fire matrices were closer to high fire severity because they were calibrated for upland sites. Then we calculated the LSF matrix by applying a 50% reduction rate compared to HSF as per Kashiske et al, 2006.

Table B1 : disturbance matrice used in the simulation

Source pool	Sink pool	Default values proportion (high fire severity)	Modified proportion low fire severity
Softwood Merchantable	Softwood Stem Snag	1	1
Softwood Foliage	CO2	0,9	
Softwood Foliage	CH4	0,01	
Softwood Foliage	CO	0,09	
Softwood Other	Softwood Branch Snag	0,750002	0,79250166
Softwood Other	CO2	0,224998	0,18674834
Softwood Other	CH4	0,0025	0,002075
Softwood Other	CO	0,0225	0,018675
Softwood Submerchantable	Softwood Branch Snag	0,75	0,7925
Softwood Submerchantable	CO2	0,225	0,18675
Softwood Submerchantable	CH4	0,0025	0,002075
Softwood Submerchantable	CO	0,0225	0,018675
Softwood Coarse Roots	Aboveground Fast DOM	0,5	

Softwood Coarse Roots	Belowground Fast DOM	0,5	
Softwood Fine Roots	Aboveground Very Fast DOM	0,454948	0,45859676
Softwood Fine Roots	Belowground Very Fast DOM	0,5	0,50401008
Softwood Fine Roots	CO2	0,040546	0,03365318
Softwood Fine Roots	CH4	0,000451	0,00037433
Softwood Fine Roots	CO	0,004055	0,00336565
Aboveground Very Fast DOM	Aboveground Very Fast DOM	0,031467	0,56416015
Aboveground Very Fast DOM	CO2	0,87168	0,392256
Aboveground Very Fast DOM	CH4	0,009685	0,00435825
Aboveground Very Fast DOM	CO	0,087168	0,0392256
Belowground Very Fast DOM	Belowground Very Fast DOM	1	
Aboveground Fast DOM	Aboveground Fast DOM	0,358407	0,71128315
Aboveground Fast DOM	CO2	0,577434	0,2598453
Aboveground Fast DOM	CH4	0,006416	0,0028872
Aboveground Fast DOM	CO	0,057743	0,02598435
Belowground Fast DOM	Belowground Fast DOM	1	
Medium DOM	Medium DOM	0,60786	0,823537
Medium DOM	CO2	0,352926	0,1588167
Medium DOM	CH4	0,003921	0,00176445
Medium DOM	CO	0,035293	0,01588185
Aboveground Slow DOM	Aboveground Slow DOM	0,909897	0,95945365
Aboveground Slow DOM	CO2	0,081093	0,03649185
Aboveground Slow DOM	CH4	0,000901	0,00040545
Aboveground Slow DOM	CO	0,008109	0,00364905

Belowground Slow DOM	Belowground Slow DOM	1	
Softwood Stem Snag	Medium DOM	1	
Softwood Branch Snag	Aboveground Fast DOM	1	

Empty cells signify unchanged values, thus the same as the default values.

Appendix C: Parameters modified moss module.

Moss parameters

We assume that the default parameters were adapted for site less prone to paludification BSFM stand.

Table C1 : moss parameters used in the simulation

Parameters	Description	Default_value	BSFM stand		BSSP Stand		References
			HSF	LSF	HSF	LSF	
Growth							
FeatherNPPconstant	annual feather moss Net primary productivty rate (Mg C ha−1 yr−1)	based on equation see Bona et al., 2016	0,24	0,24	0,24	0,24	*Bisbee et al, 2001*
SphagnumNPPconstant	annual sphagnum Net primary productivty rate (Mg C ha−1 yr−1)	based on equation see Bona et al 2016	0,77	0,77	0,77	0,77	*Bisbee et al, 2001*
Sphagnum ground Cover (%)	Linear regression depends to canopy openness see appendix table ,,,,	based on equation see Bona et al 2016	Linear regression depends to canopy openness see appendix D				*Fenton et al, 2015*
Feather moss ground Cover (%)	100 % - Sphagnum ground Cover(%)	based on equation see Bona et al 2016	100 % - Sphagnum ground Cover(%)				*Fenton et al, 2015*
NPPconstant* bryophyte ground cover(%)		based on equation see Bona et al 2016					
Decomposition							

kff	Constant decay feather fast kff, base decay rate feather moss fast pool	0,018	0,018	0,018	0,018	0,018	Default value , Bona et al, 2016
Ksf	Constant decay sphagnum fast	0,028	0,028	0,028	0,015	0,015	Default value , Bona et al, 2016 /// paludified Fissore et al, 2019
kfs	Constant decay feather slow kfs	0,006	0,006	0,006	0,006	0,006	*Bona et al, 2016*
m	For Constant decay sphagnum slow kss (formule (LN(MVOL)* m) + n), MVOL maximum of tree volume	0,0003	0,0004	0,0004	0,0003	0,0003	*Bona et al, 2016*
n	For Constant decay sphagnum slow kss (formule (LN(MVOL)* m) + n)	0,00007	0,0004	0,0004	0,00007	0,00007	*Bona et al, 2016*
kss	Constant decay sphagnum slow kss						
MAT	Mean temperature	NA	0	0	0	0	*climate normals of study area*
Q10acrotelme_fast	Q10 for annuel decay applied for fast pool	1	2,278	2,278	2	2	*Unpaludied or upland Bona et al, 2016 based on Hararuk et al,2017 , default values 1 //Scanlon et Moore 2000 ; Payette et al, 2002*
Q10catotelme_slow	Q10 for annuel decay applied for slow pool	1	2,278	2,278	2,7	2,7	*Unpaludied Bona et al, 2016 based on Sasha Hararuk et al, 2017 , default value 1 //Scanlon et Moore 2000 ; Payette et al, 2002*
ak for each bryophyte	applied bryophyte dead organic matter pool applied decay rate, kff (or Ksf, kfs, kfs)*(e^((MAT-10)*(ln(Q10)*0.1)						Bona et al, 2016
Annuel turnover or C transfert							
Moss live	C transfert from the moss live to moss fast dead organic matter; annual turnover 1/3 years	1	0,33	0,33	0,33	0,33	*During et al , 1987 ; During et al , 1979 ; Asada et al, 2003 ; default value = 1*
Ptfast_to_slow	carbon proportion transfert from DOM FAST decayed to Slow in absence of fire	0,15	0,15	0,15	0,15	0,15	*default value , Bona et al, 2016 based on Clymo model, payette et al , 2002*
Ptfast_to_Atmo	carbon proportion transfert from DOM FAST decayed to	0,85	0,85	0,85	0,85	0,85	*default value , Bona et al, 2016 based*

	atmosphere in absence of fire (1-Ptfast to slow)						*on Clymo model, payette et al , 2002*
Ptslow_to_Atmo	carbon proportion transfert from DOM Slow decayed to the atmosphere in absence of fire	1	1	1	1	1	*default value , Bona et al, 2016*
Fire effects Only DOM fast is burn							
Prop_Cburned_in_fire_Fmlive		1	1	1	1	1	default value 1 , Bona et al, 2016
Prop_Cburned_in_fire_Splive		1	0,9	0,77	0,9	0,77	default value 1 , Bona et al, 2016/ modfied based on Shetler et al, 2008
Prop_Cburned_in_fire_FMfast	proportion of Feather moss DOM fast carbon burned to atmosphere	0,9	0,9	0,526315789	0,7695	0,45	*unpaludified default value , Bona et al, 2016 , shetler et al ,2008 // effect divided by 2 for paludified dominated by sphagnum*
Prop_Cburned_in_fire_SPfast	proportion of sphagnum moss DOM fast carbon burned to atmosphere	0,6	0,56	0,32748538	0,3762	0,22	unpaludified default value , Bona et al, 2016 , shetler et al ,2008 // effect divided by 2 for paludified dominated by sphagnum Lecomte et al, 2008
Prop_residuel_livefeather	proportion of Feather moss live biomass non impacted by disturbance, remaining after disturbance	0	0	0	0	0	unpaludified default value , Bona et al, 2016 , shetler et al ,2008
Prop_residuel_livesphag	proportion of Sphagnum moss live biomass non impacted by disturbance, remaining after disturbance	0	0,1	0,23	0,1	0,23	unpaludified default value , Bona et al, 2016 , shetler et al ,2008
Prop_unburned_FM_DOM_FAST	proportion of Feather moss DOM fast carbon unburned (1-prop_Cburned)	0	0,1	0,473684211	0,2305	0,55	unpaludified default value , Bona et al, 2016 , shetler et al ,2008
Prop_unburned_SP_DOM_FAST	proportion of Sphagnum moss DOM fast carbon unburned (1-prop_Cburned)	0	0,44	0,67251462	0,6238	0,78	unpaludified default value , Bona et al, 2016 , shetler et al ,2008
Prop_residuel_FM_DOM_FAST	proportion of Feather moss live biomass non impacted by disturbance , eally remaining in the	0	0	0	0,23	0,461	unpaludified default value , Bona et al, 2016 , shetler et al ,2008

	pool, at year disturbance						
Prop_residuel_SP_DOM_FAST	proportion of Sphagnum moss live biomass non impacted by disturbance , really remaining in the pool, at year disturbance	0	0,1	0,23	0,325	0,65	unpaludified default value , Bona et al, 2016 , shetler et al ,2008
PropC_Fmfast_to_slow	C feathermoss DOM fast residual proportion for slow	0,1	0,1	0,473684211	0,0005	0,089	unpaludified default value , Bona et al, 2016 , shetler et al ,2008
PropC_SPfast_to_slow	C sphagnum moss DOM fast residual proportion for slow	0,4	0,34	0,44251462	0,2988	0,13	unpaludified default value , Bona et al, 2016 , shetler et al ,2008
Decay_multiplier	proportion to multiplier decay after fire before steady state of forest, short time here 15 years	1	4	3	2	1,5	*Unpaludied O'Neill et al, 2006, Paludidied Bond-Lamberty 2004*

Reference, Bona et al, 2016 ; Bisbee et al, 2001; Payette et al, 2001 ; Scanlon et Moore 2000; Harden et al, 1997, Fissore et al, 2019.

Appendix D: Data used for moss module calibration

Experimental design

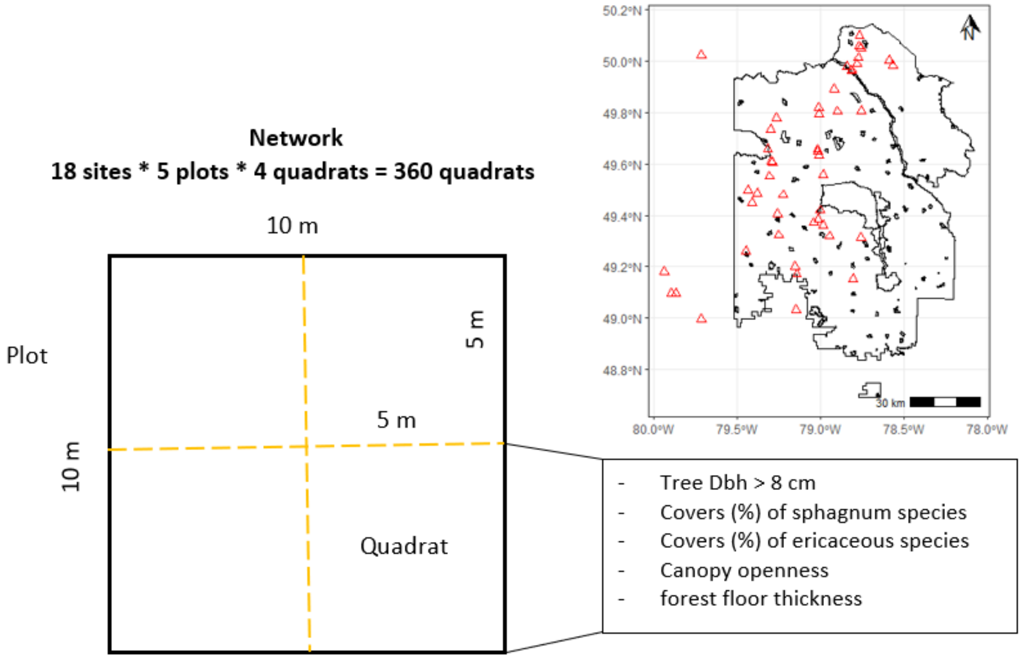
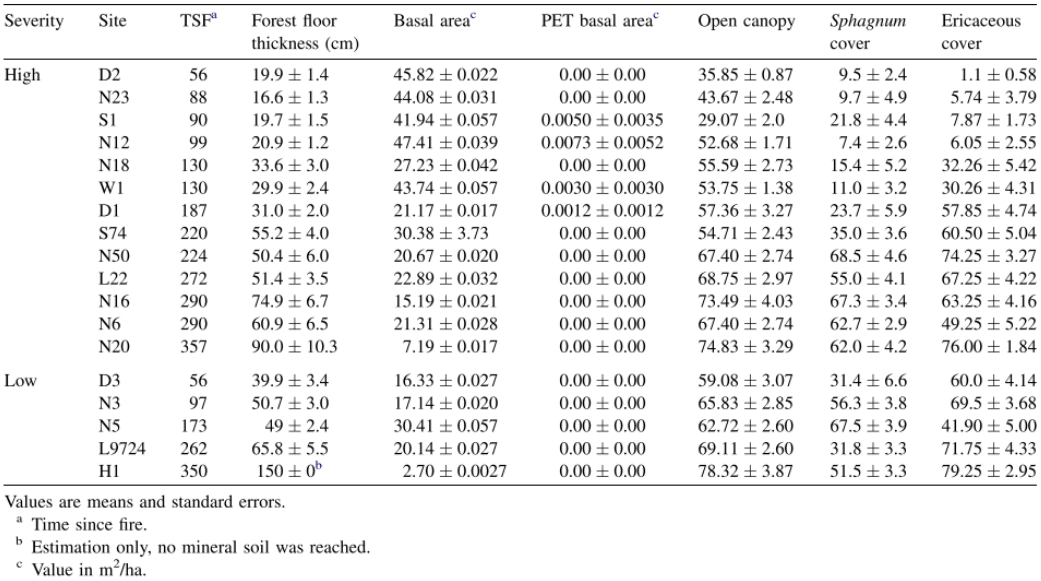


Figure D1: Location of experimental design from Fenton et al., 2005 used for linear regression for mosses cover dynamic and the study area Forest Management Unit 085-5.

Table D1: Description of high and low fire severity sites used in analyses (Fenton et al., 2005)



Severity	Site	TSF[a]	Forest floor thickness (cm)	Basal area[c]	PET basal area[c]	Open canopy	*Sphagnum* cover	Ericaceous cover
High	D2	56	19.9 ± 1.4	45.82 ± 0.022	0.00 ± 0.00	35.85 ± 0.87	9.5 ± 2.4	1.1 ± 0.58
	N23	88	16.6 ± 1.3	44.08 ± 0.031	0.00 ± 0.00	43.67 ± 2.48	9.7 ± 4.9	5.74 ± 3.79
	S1	90	19.7 ± 1.5	41.94 ± 0.057	0.0050 ± 0.0035	29.07 ± 2.0	21.8 ± 4.4	7.87 ± 1.73
	N12	99	20.9 ± 1.2	47.41 ± 0.039	0.0073 ± 0.0052	52.68 ± 1.71	7.4 ± 2.6	6.05 ± 2.55
	N18	130	33.6 ± 3.0	27.23 ± 0.042	0.00 ± 0.00	55.59 ± 2.73	15.4 ± 5.2	32.26 ± 5.42
	W1	130	29.9 ± 2.4	43.74 ± 0.057	0.0030 ± 0.0030	53.75 ± 1.38	11.0 ± 3.2	30.26 ± 4.31
	D1	187	31.0 ± 2.0	21.17 ± 0.017	0.0012 ± 0.0012	57.36 ± 3.27	23.7 ± 5.9	57.85 ± 4.74
	S74	220	55.2 ± 4.0	30.38 ± 3.73	0.00 ± 0.00	54.71 ± 2.43	35.0 ± 3.6	60.50 ± 5.04
	N50	224	50.4 ± 6.0	20.67 ± 0.020	0.00 ± 0.00	67.40 ± 2.74	68.5 ± 4.6	74.25 ± 3.27
	L22	272	51.4 ± 3.5	22.89 ± 0.032	0.00 ± 0.00	68.75 ± 2.97	55.0 ± 4.1	67.25 ± 4.22
	N16	290	74.9 ± 6.7	15.19 ± 0.021	0.00 ± 0.00	73.49 ± 4.03	67.3 ± 3.4	63.25 ± 4.16
	N6	290	60.9 ± 6.5	21.31 ± 0.028	0.00 ± 0.00	67.40 ± 2.74	62.7 ± 2.9	49.25 ± 5.22
	N20	357	90.0 ± 10.3	7.19 ± 0.017	0.00 ± 0.00	74.83 ± 3.29	62.0 ± 4.2	76.00 ± 1.84
Low	D3	56	39.9 ± 3.4	16.33 ± 0.027	0.00 ± 0.00	59.08 ± 3.07	31.4 ± 6.6	60.0 ± 4.14
	N3	97	50.7 ± 3.0	17.14 ± 0.020	0.00 ± 0.00	65.83 ± 2.85	56.3 ± 3.8	69.5 ± 3.68
	N5	173	49 ± 2.4	30.41 ± 0.057	0.00 ± 0.00	62.72 ± 2.60	67.5 ± 3.9	41.90 ± 5.00
	L9724	262	65.8 ± 5.5	20.14 ± 0.027	0.00 ± 0.00	69.11 ± 2.60	31.8 ± 3.3	71.75 ± 4.33
	H1	350	150 ± 0[b]	2.70 ± 0.0027	0.00 ± 0.00	78.32 ± 3.87	51.5 ± 3.3	79.25 ± 2.95

Values are means and standard errors.

[a] Time since fire.

[b] Estimation only, no mineral soil was reached.

[c] Value in m^2/ha.

Appendix E: Equations used

Sites more prone to paludification BSSP

Table E1: The relative bryophytes ground cover mathematical models according fire severity for age>50 . feather moss and sphagnum represent 100% of total moss cover. Relationship depends to canopy openness.

Y	Models Low fire severity	High fire severity	R2
Cover(%) of Sphagnum	b0eqcoverspha + b1eqcoverspha * Canopy_openness + b2eqcoverspha * an b0eqcoverspha <- 31.94607 b1eqcoverspha <- -0.18755 b2eqcoverspha <- 0.19418	b0eqcoverspha <- 31.94607 -25.53726 b1eqcoverspha <- -0.18755 b2eqcoverspha <- 0.19418	0.42
Canopy openness	b0canopy + b1canopyvol * VolumeBois + b2canopyage *an b0canopy <- 52.080550		0.41

	b1canopyvol <- -0.028958 b2canopyage <- 0.090580		
Cover(%) of feathermoss	100- Cover sphagnum	100- Cover sphagnum	NA
NPP	NPPconstant*cover(%)	NPPconstant*cover(%)	NA

This relationship describes a higher feathermoss cover after HSF than after LSF in young stand, but the feather moss cover is replaced by Sphagnum species in sites more prone to paludification. The equations are only valid for ages ≥50 years, so we extrapolated for ages <50 years.

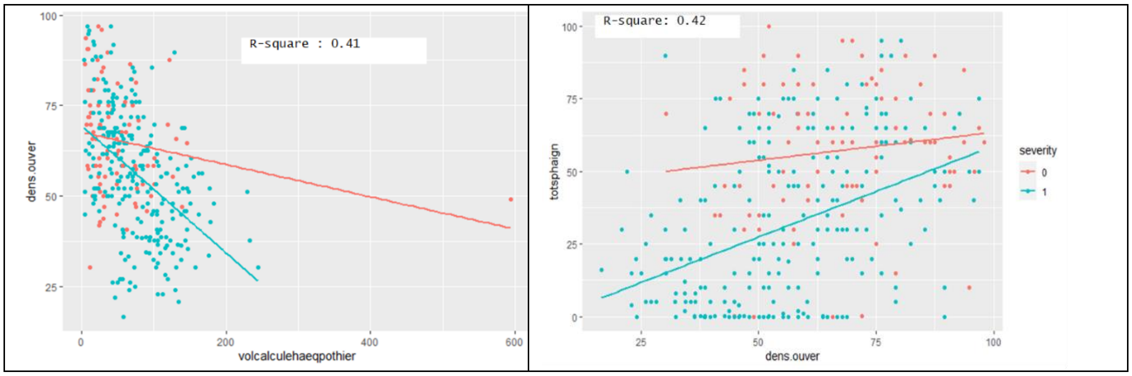


Figure E1: Canopy openness according tree volume and fire severity and sphagnum cover according to canopy openness and fire severity. 0- Low-severity fire / 1- High-severity fire.

Dens.over is canopy openness/ volcalculehaeqpothier is tree volume/ Totsphaign is total sphagnum cover.

Bryophytes cover in non paludified stand BSFM

Sites less prone to paludification were dominated by feathermoss, well drained. Instead, we modified the cover after HSF and LSF to reach a maximum of 15% and

30% Sphagnum cover, respectively, according to published literature on productive black spruce-feathermoss dominated stands (Bona *et al.,*2016; Bond-Lamberty *et al.*, 2004, O'Neill et al., 2006 ; Ward *et al,* 2014).For instance, the maximum bryophytes cover reached in Bond-Lamberty *et al.*, 2004 in well drained black spruce stand is 12% at 151 years and sphagnum cover reached 3-10 % at 140 years after fire in black spruce stands of Alaska .We hypothesized that this can occurring after high fire severity fire because sites were not palufidified. Instead, we took the double of this value for sphagnum cover after LSF in this group of forest (20-30%). Moreover, we used the predictive equation of sphagnum cover according to organic thickness layer in ward et al, 2014.

Equation: Sphagnum sp. (%) -15.100 + 1.121(OTL organic thickness layer) . with the limit of OTL (30-40 cm) to be non paludified stand. We found 18.53% for 30 cm and 29.74% for 40 cm.

According ward et al, 2014, The OLT increased during the first 64 years after fire but stayed statistically constant thereafter. Moreover, the absence of a significant decrease in productivity from 80 to 2000 years post-fire suggests that these characteristics reach a steady state early in the chronosequence and remain stable in the absence of major disturbances or shifts in site conditions.

Appendix F: Landscape description

In this study, natural paludified stands or liable to paludification were selected according to the following criteria: black spruce (dominant species), low forest density (25 % < D, C, B < 80%), Till (1AA), glaciolacustrine (4GA) and organic soil deposit (7 E and 7 T), relatively poorly drainage class (>=30), black spruce-feathermoss, spruce-sphagnum ecological type, and non-anthropogenic disturbances.

Table F1 : Selected stands of FMU 8551 in the 2010 forest inventory data.

Forest group	caracteristics	Area (ha)
Black spruce-sphagnum (more prone to paludification)	dominant moss species : sphagnum low forest density : B , C dominant , D (Csup D, sup C) drainage class : poorly 50-60 surficial deposits (more organic and low glaciolacustrine "7T", "7E">>> "1AA" "4GA")	147 211.6
Black spruce-feathermoss (less prone to paludification)	dominant moss species : feathermoss intermediate forest density : B dominant , C, D drainage class : moderate 30-40 surficial deposits (glaciolacustrine, "1AA" "4GA")	32 314.75

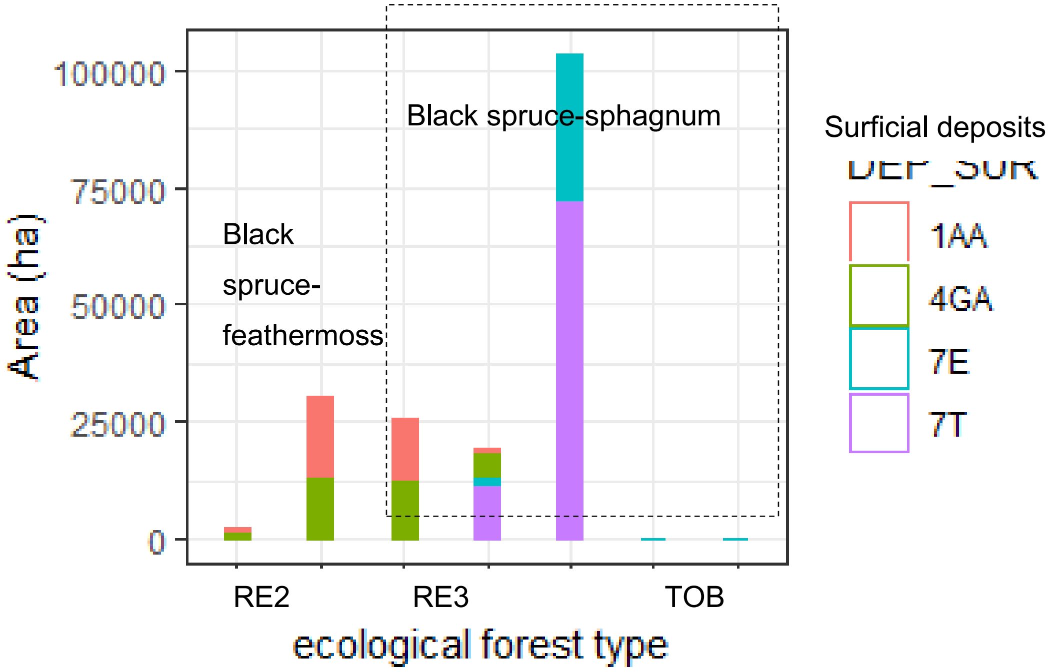


Figure F1: surficial deposits according forest types in forest inventory 2010 . RE2 : Black spruce forest with mosses or ericaceous plants. RE3 : Black spruce forest with sphagnum mosses. TOB : Ombrotrophic bog.

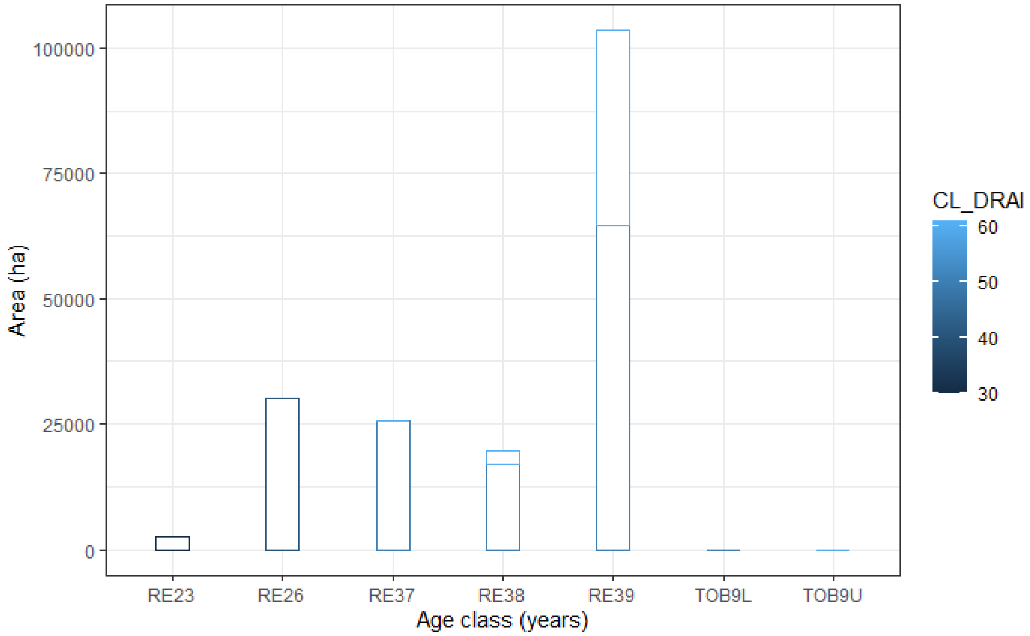


Figure F2: drainage classes according forest types in forest inventory 2010. 30 Moderately well-drained, 40 Imperfectly drained, 50 Poorly drained, 60 Very poorly drained.

Appendix G : growth, yields curves and soil

Growth

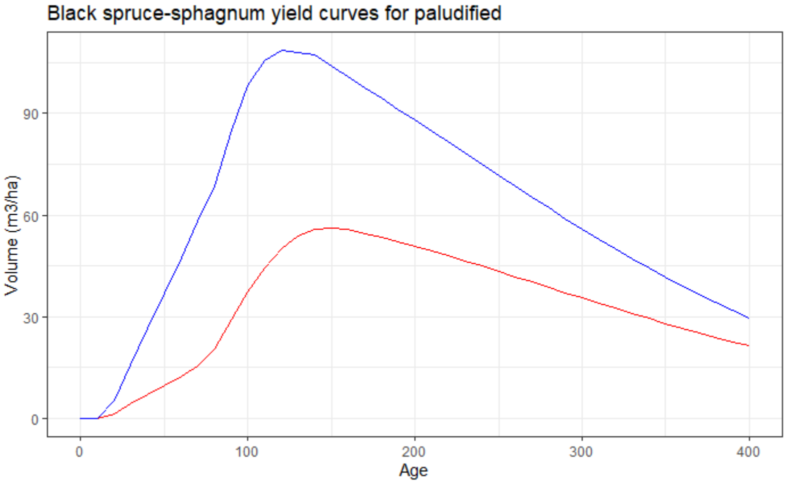
Constant NPP was used. Yield curves were selected based on moss dominance. For upland black spruce feather moss with very high productivity for HSF and lower productivity for HSF but less than HSF, for paludified for HSF moderately productivity of black spruce feather moss followed by low productivity of black spruce sphagnum due to paludification in the absence of fire, for LSF low productivity sphagnum, and very low productivity of black spruce sphagnum dominance. Yield curves we chosen based on studies in the study area (Lecomte et al, 2006, Simard et al ,2007).

We adjusted and extrapolated the curves to align with our simulation duration of 400 years, considering that the curve's original duration was limited to 250 years. The selection and analysis of yield curves were based on previous studies (Garet et al., 2009; Simard et al., 2007).

We maintain the beginning of the curve until the maximum volume, and after the inflection point, apply a decreasing exponential function to represent a logarithmic decline, as described in Lecomte et al., 2006. Different reduction rates were applied at various locations after the inflection point. The goal of the reduction is to achieve a decrease of about half in site prone paludification and in upland , 34% of the volume by the end of the simulation period (400 years) and maintain the logarithmic form of the curve (Andrieux et al., 2018; Lecomte et al., 2006)

Example = volumes [i - 1] * exp(-reduction_rate1 * ages[i]), three or four rate were used.

Example of yield curves extrapolated.



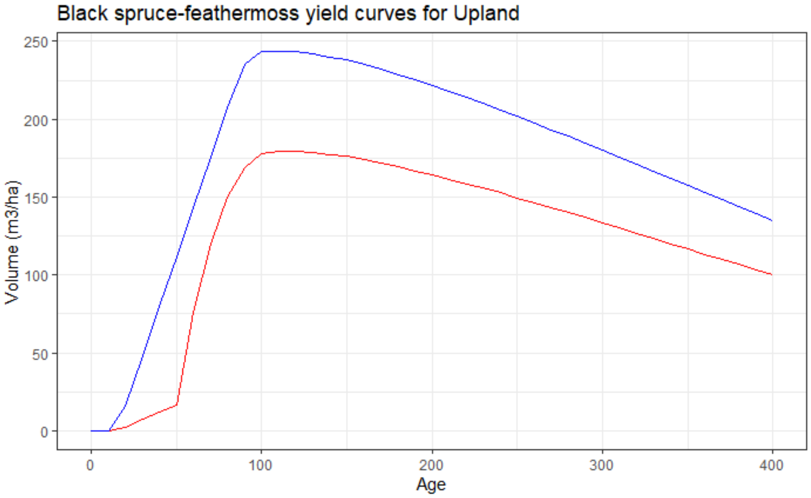
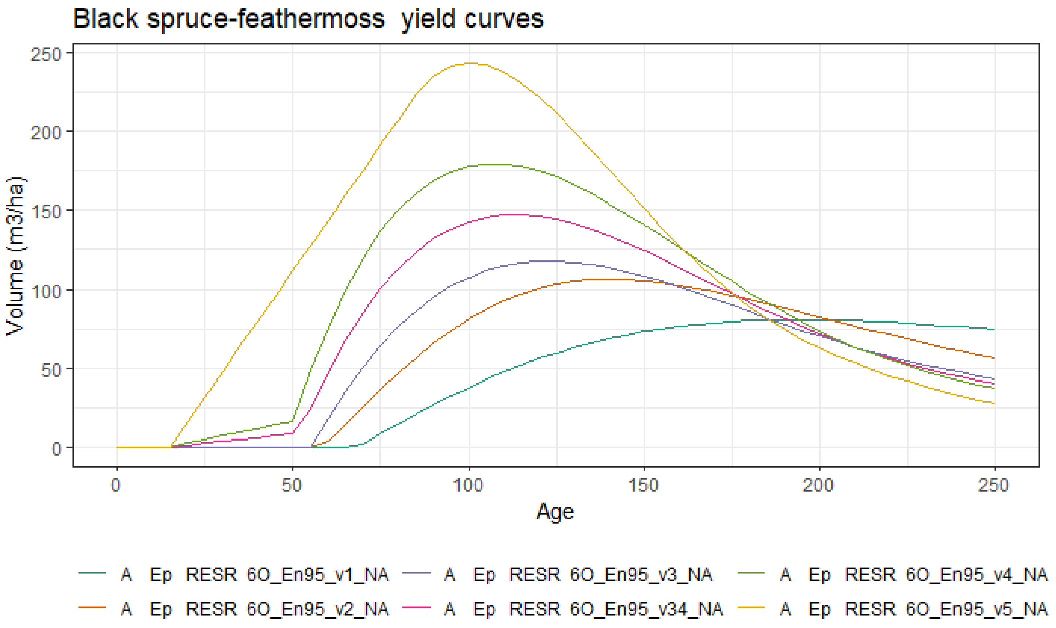


Figure G1: Yield curves according fire severity in paludified stand (Black spruce sphagnum) and non paludified stand (Black spruce feathermoss). Blue line for high severity fire (HSF) and red line for low severity fire (LSF).

Previous yield curves before extrapolation

For black-spruce feathermoss, less prone to paludification



Different curves with productivity index compared to the IQS

For black-spruce sphagnum more prone to paludification

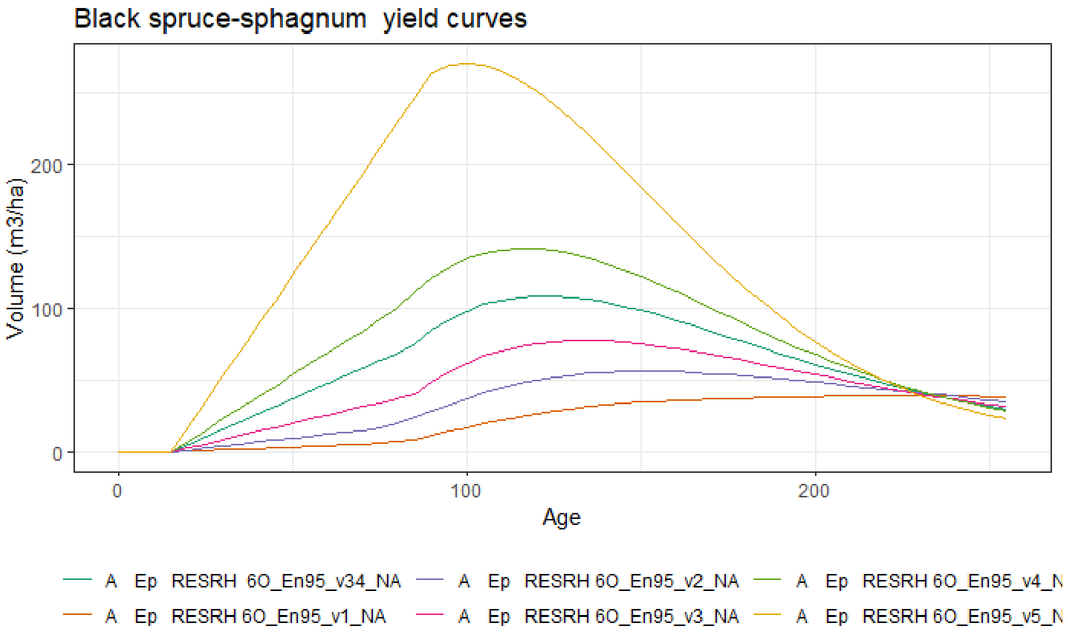


Figure G2 : Previous yield curves before extrapolation

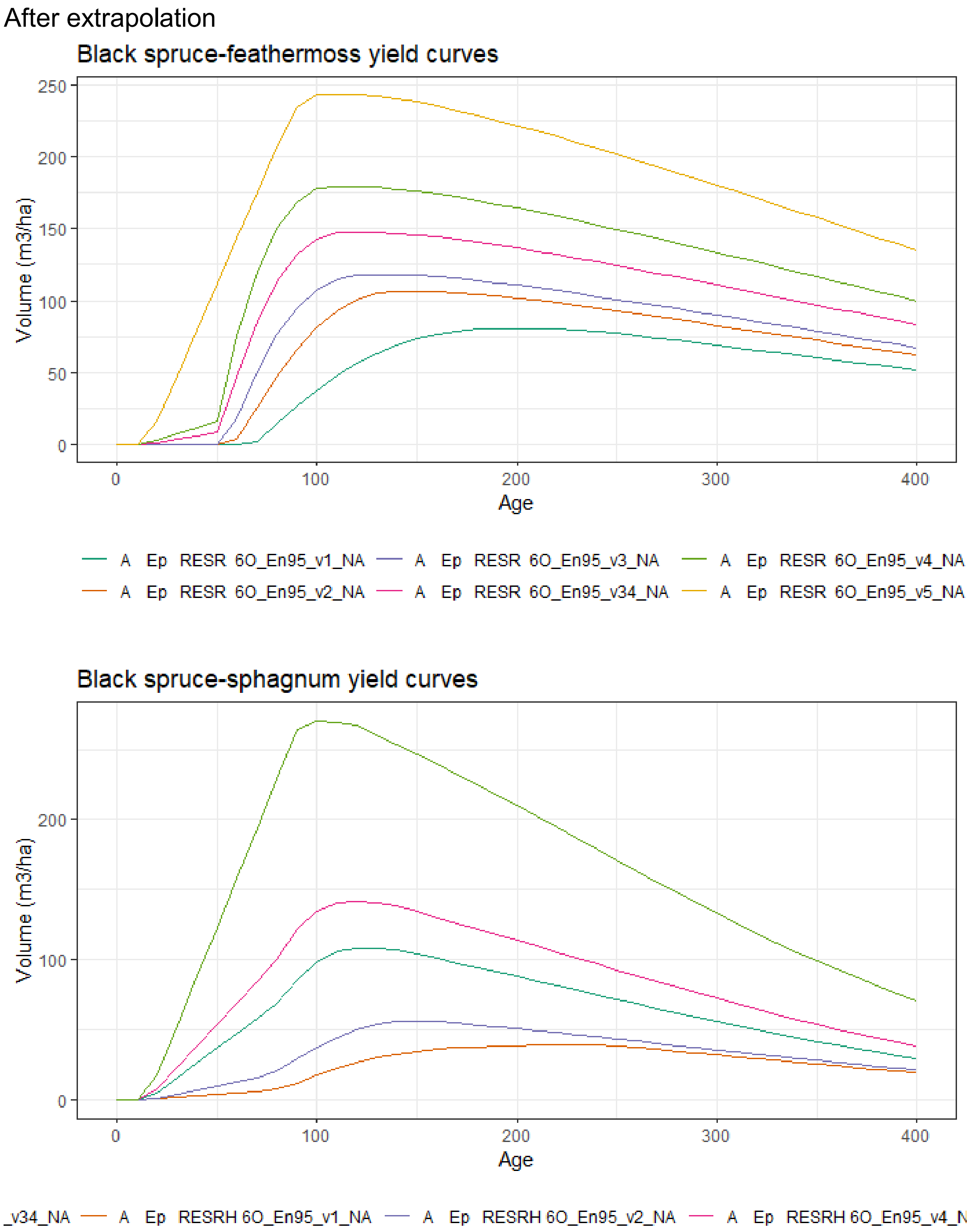


Figure G3 : Yield curves after extrapolation

Moss ground cover linear regression depends on canopy openness, as shown in the equation used. However, for unpaludified stands, linear regression adapted to forests susceptible to paludification was employed. We utilized the moss dynamic equation developed by Kelly Bona et al. for the Clay Belt study area in Quebec and applied the yield curves of unpaludified stands (black spruce feather moss). To align with the key values from the equation of Kelly and the maximum values identified in the literature over the simulation period (400 years), we adjusted the moss dynamic based on the regression equation of Fenton. We applied different reduction coefficients over time.

Moss soil pool

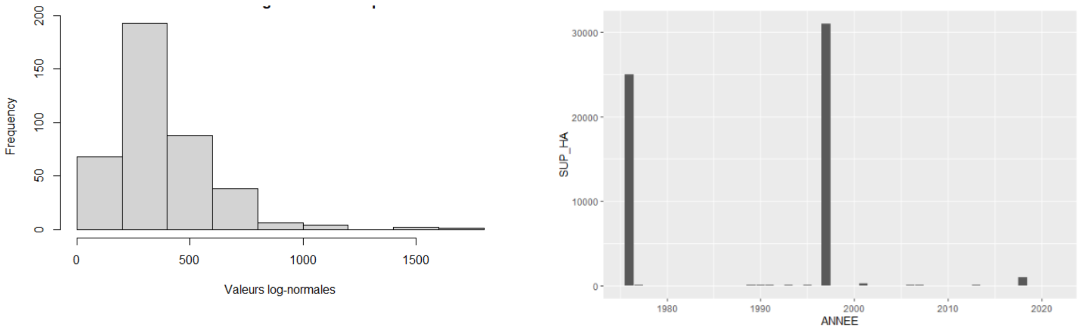
Hypotheses were made on soil organic pool. Because not all living moss was consumed, mainly sphagnum, organic soil fast equivalent to this remaining living moss proportion must also remain. In paludified stand, not all organic fast soil after burn must be transferred to organic soil slow pool because of soil moisture, another proportion of will stay in organic soil fast. Because in the paludified group, the slope of organic soil slow C pool slows down rapidly, the proportion residual of organic fast is taken arbitrarily for Sphagnum species in LSF fire. Besides the correspondence of the live moss remaining after disturbance, for upland or unpaludified stand, we maintain the default hypothesis that all organic soil pool remaining after fire go to the organic soil slow.

Appendix H : Yearly area burnt

Yearly area burnt was randomly selected from a log-normal distribution with an average of log (0.25% or 1% of the land base) and a standard deviation of 0.5. because fire events can show an extreme event, we used this distribution. It has a long distribution tail, which means it assigns a significant probability to values that are extremely distant from the mean. We also compared the distribution log-normal to the real fire distribution in my study area.

Log-normal distribution

Study area AUF 85-51



Area burnt in North region of Quebec (BFEC)

Bergeron *et al.*, 2004 conifer region in northern Quebec

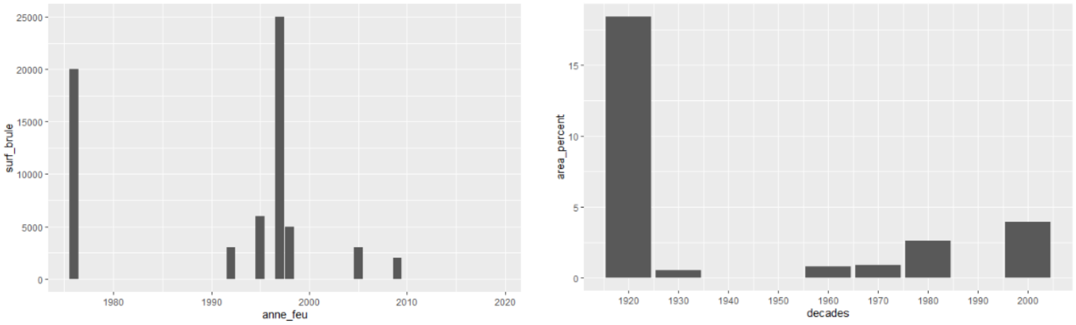


Figure H1: Comparison between the distribution log-normal used in the simulation to the real fire distribution in my study area and other real distribution in the previous studies.

All real fire distributions were characterized by the extreme fire events .

Appendix I =other results

Stand level

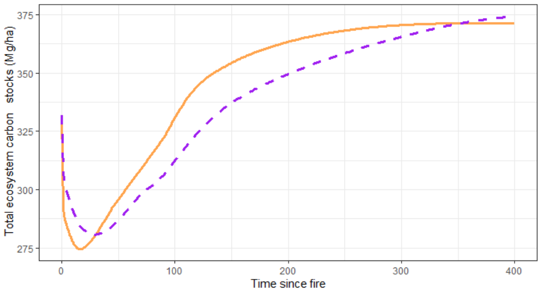
Table I1: Summary of carbon stocks (Mg C ha-1) in total, biomass, and organic soil per disturbance scenario: Initial values, values at the disturbance year, final values, accumulation rates, and average moss proportions.

Values		BSSP		BSFM	
		HSF	LSF	HSF	LSF
Total	Initial	224.31		194.40	
	Distubance year	181.54	196.95	131.58	153.16
	Final	244.95	258.84	173.44	163.51
	Accumultion rate	0.052	0.086	-0.05	-0.08
	Moss proportion	0.67	0.75	0.21	0.28
Biomass	Initial	35.98		93.39	
	Distubance year	0.1	0.23	0.02	0.05
	Final	29.33	25.61	64.75	53.94
	Accumultion rate	-0.017	-0.026	-0.07	-0.1
	Average moss proportion	0.03	0.05	0.012	0.018
Organic soil layer	Initial	172.46		62.01	
	Distubance year	151.72	161.35	43.03	52.97
	Final	198.54	218.26	76.46	81.37
	Accumultion rate	0.065	0.115	0.036	0.048
	Average moss proportion	0.89	0.91	0.56	0.62

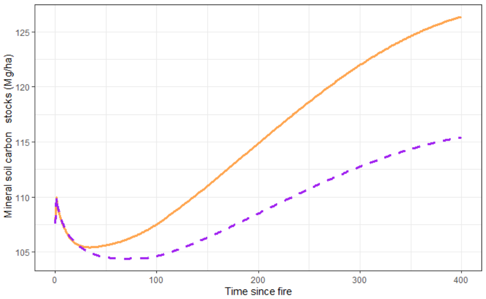
Total ecosystem stocks C with mineral soil from tree

Paludified

Total

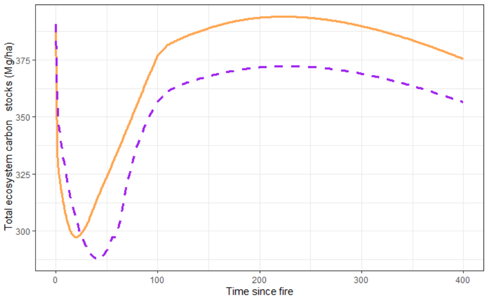


Mineral soil



Non paludified

Total



Mineral soil

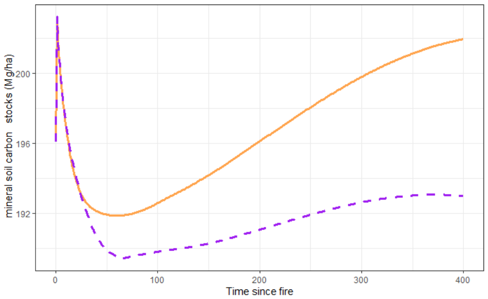
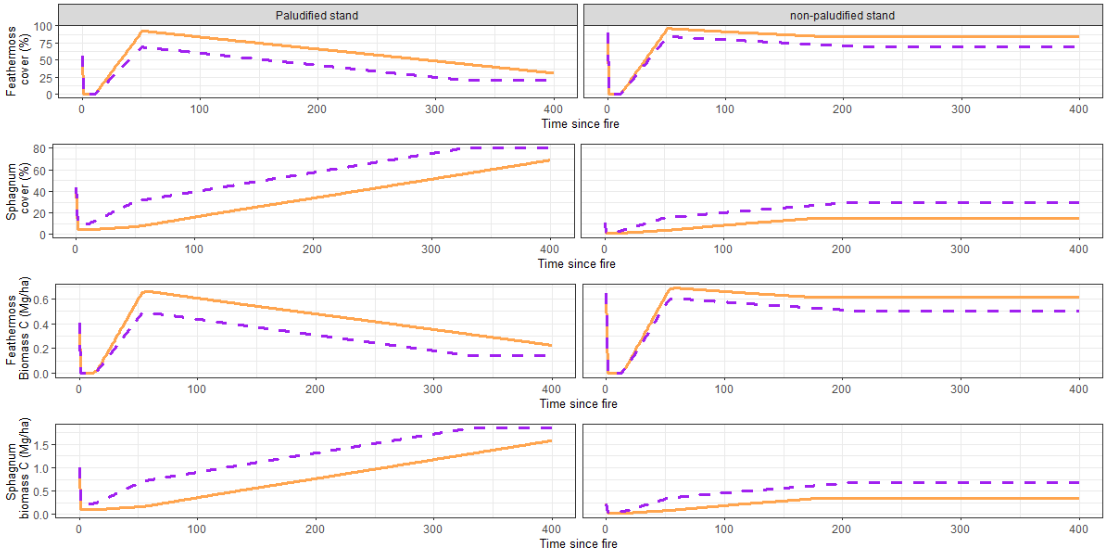


Figure I1 : Details of C storage in tree component(Total C stocks including mineral soil, mineral soil). LSF in dashed purple line and HSF in orange line.

Moss details : Paludified stand =BSSP and non-paludifed stand =BSFM. LSF in dashed purple line and HSF in orange line.



- SOIL

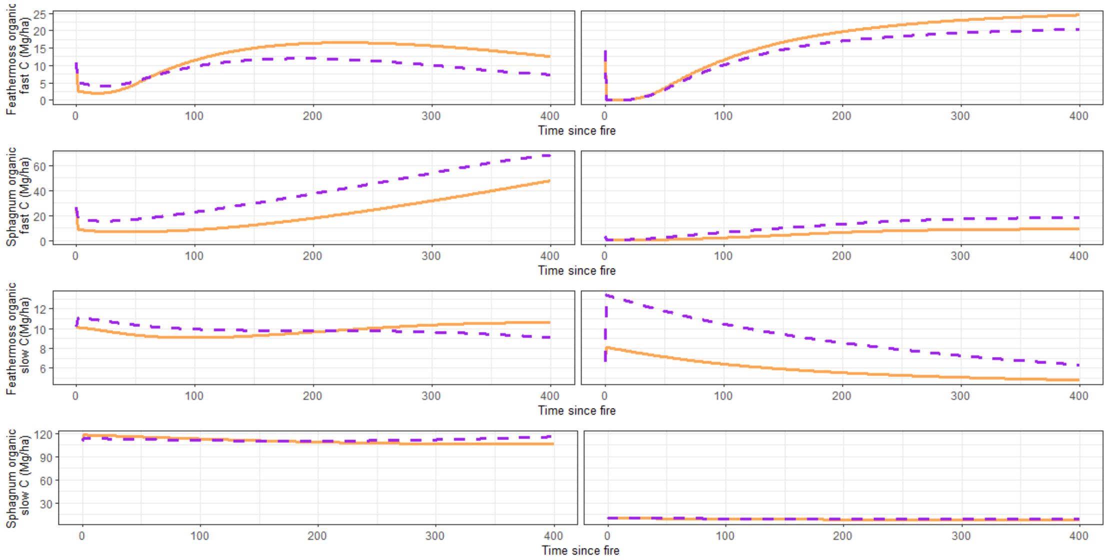


Figure I2: Details of C storage in moss component (moss cover, living biomass and soil according moss types)

Sphagnum cover increase while feathermoss decrease in paludified stand , in non plaudifed stand , feathermoss dominated over and remain steady in prolonged absence of fire .

Sphagnum increases soil stocks much.

- Moss proportion on the total without mineral soil throw the simulation.

Paludified stand Non paludified stand

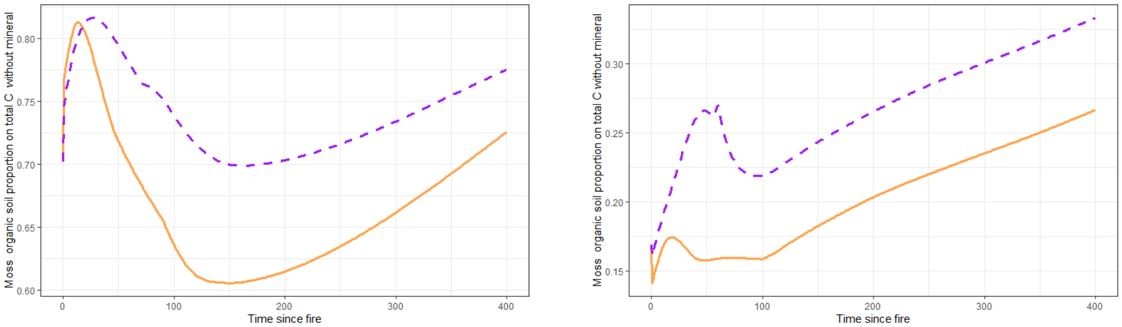
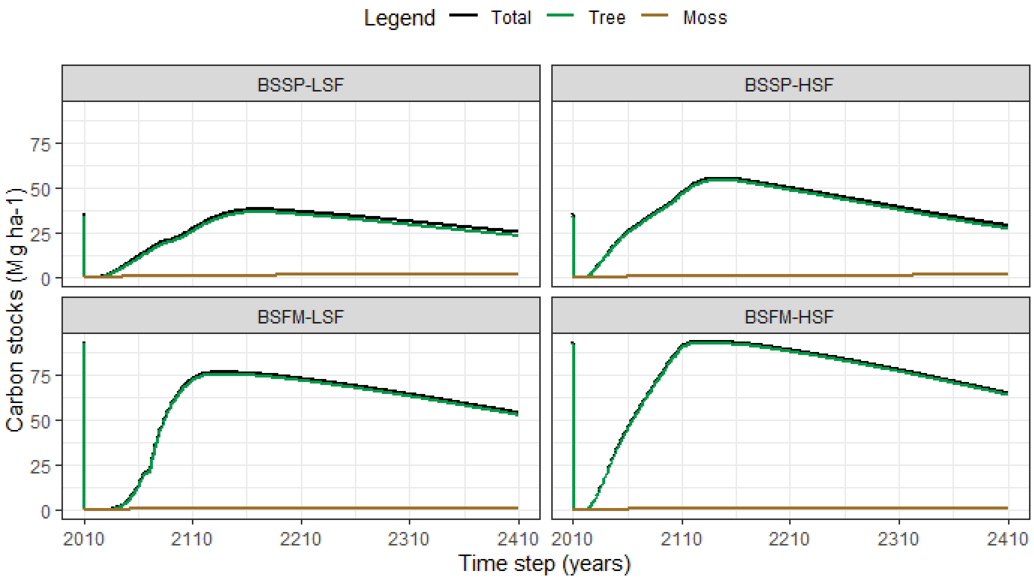


Figure I3: moss-derived organic layer C stocks proportion on total C stocks. LSF in dashed purple line and HSF in orange line.

Mosses contributed 80% to the total soil carbon pool (DOM and organic layer) in BSSP initially, dropping to 60% at mature stand age, and reaching 70-75% by the end of the simulation. In contrast, their contribution in BSFM stands was much lower, ranging from 15% to 25-30%.

- Total Biomass dynamic and organic layer

Carbon stocks in living biomass according to fire severity and stand types



Carbon stocks in organic layer according to fire severity and stand types.

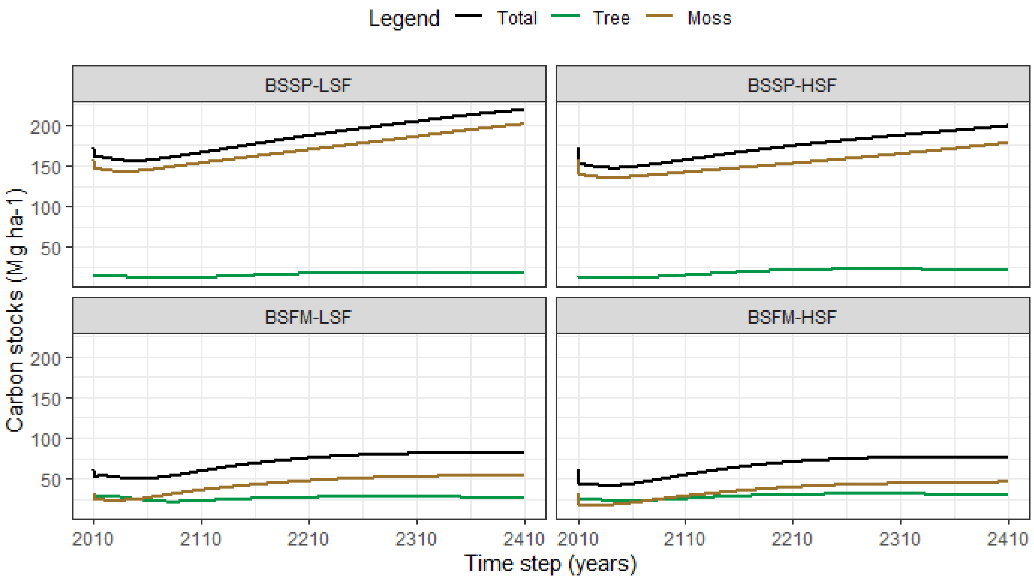
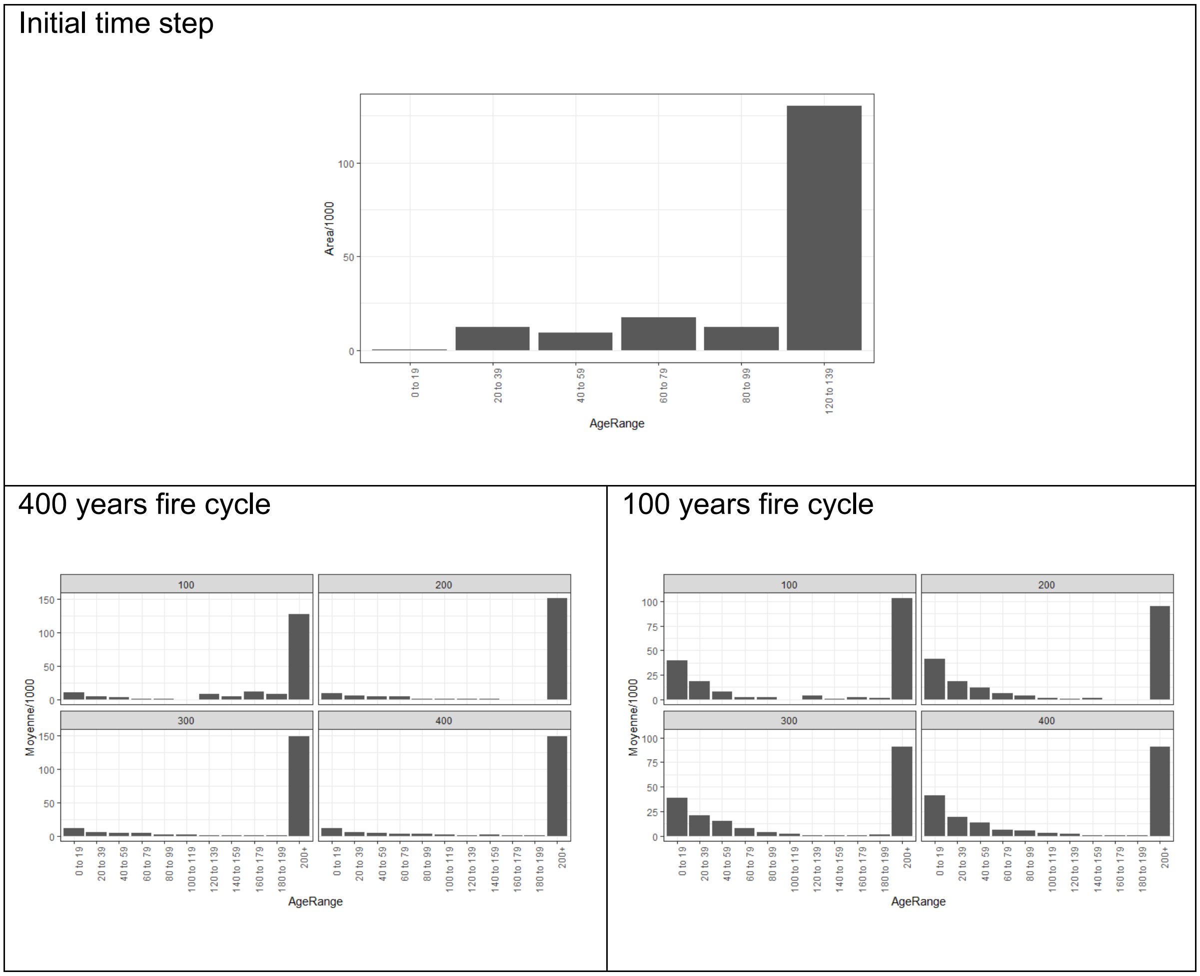


Figure I4: Total Biomass dynamic and organic layer over 400-year simulation period. trees in green, mosses in brown, and total from mosses and trees in black.

Landscape level

Age class distribution



Initial time step

400 years fire cycle

100 years fire cycle

Figure I5: Age-class distributions resulting from 400 years of simulations under 400-year fire cycle and 100-year fire cycle on the same inventory.

Emission

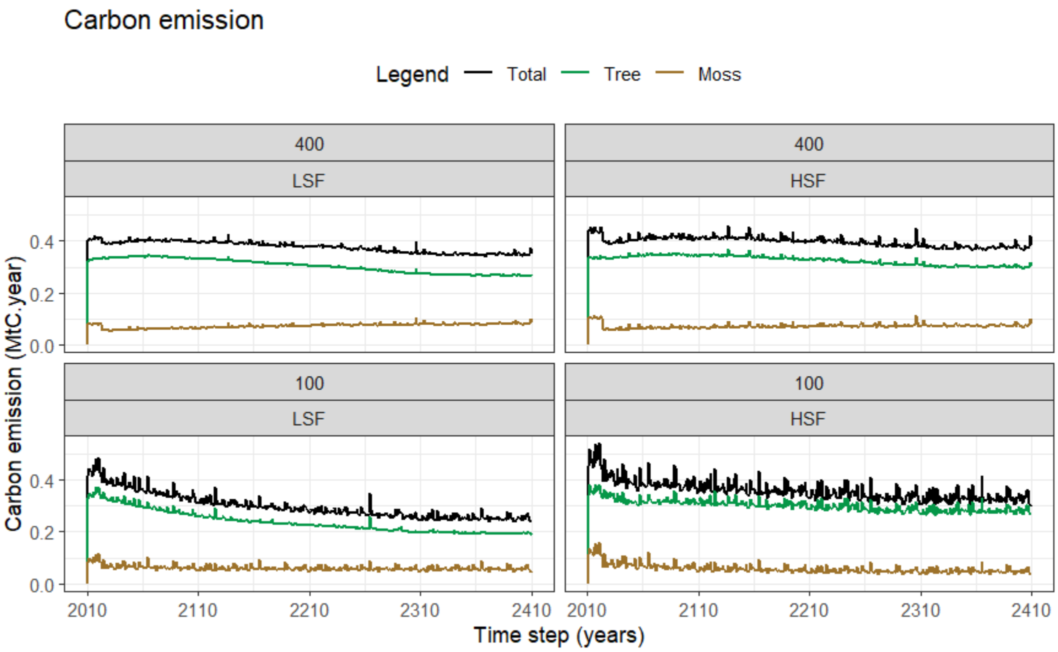


Figure I6: C emission over 400-year simulation period. trees in green, mosses in brown, and total from mosses and trees in black.

Table I2: landscape total carbon stocks (tree and moss, excluding mineral soil): initial and final stocks, average, standard deviation SD and ratio per disturbance scenario. Ratio in percent of the difference between final and initial values relative to the initial value. Stocks were divided to total landscape area (181500,6 ha) to find by ha and converted into tC (1 MtC = 10^6 tC).

Fire cycle	Severity	Initial stock (tC/ha)	Final stock (tC/ha)	Average (tC/ha)	SD (tC/ha)	Ratio (%)
400	LSF	204.04	215.84	215.6	8.1	5.8
400	HSF		210,1	213,5	5,0	2,97
100	LSF		169,14	181,5	2,2	-17.1
100	HSF		165,45	180,6	6,3	-18.9

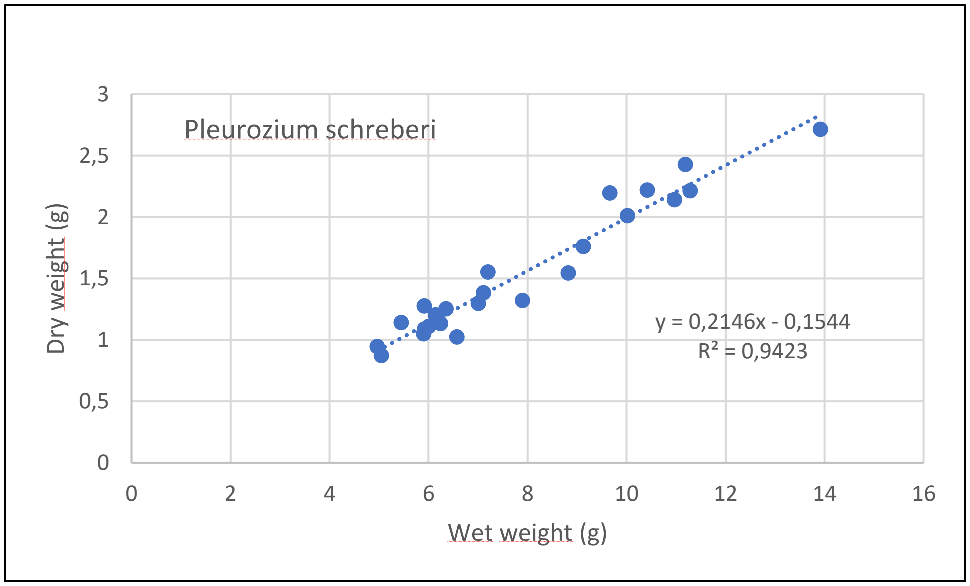
APPENDICE B – APPENDICES DU CHAPITRE 2

Appendix A: linear regression between dry and wet masses from others unincubated samples for each bryophyte type.

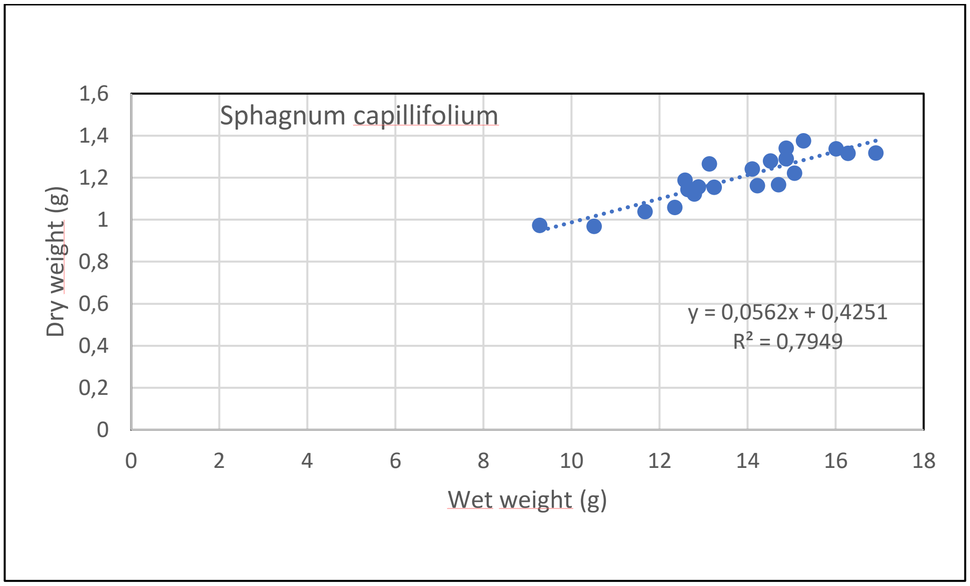
Linear regression between dry and wet weights was established to estimate the initial dry weight of the incubated samples from wet weight (Figure A). The advantage of this method is to avoid damage the incubated samples via drying and to assume the similar dry weight for different stems of the same bryophyte's species (Fenton *et al.*, 2010).

For each bryophyte type, constant wet and dry weights of about 30 unincubated samples were measured. A constant wet weight is the water remaining inside the plant so all water in exterior macropores of bryophytes was remove by centrifugation in salad spinner (see Frego and Carleton (1995), Mulligan and Gignac (2001), Fenton and al, 2010). The spinning cycle was repeated until the weight of sample stops decreasing. The wet unincubated samples are then dried at 30° C to constant weight.

A1. Pleurozium schreberi



A2. Sphagnum capillifolium



A3. Sphagnum fuscum

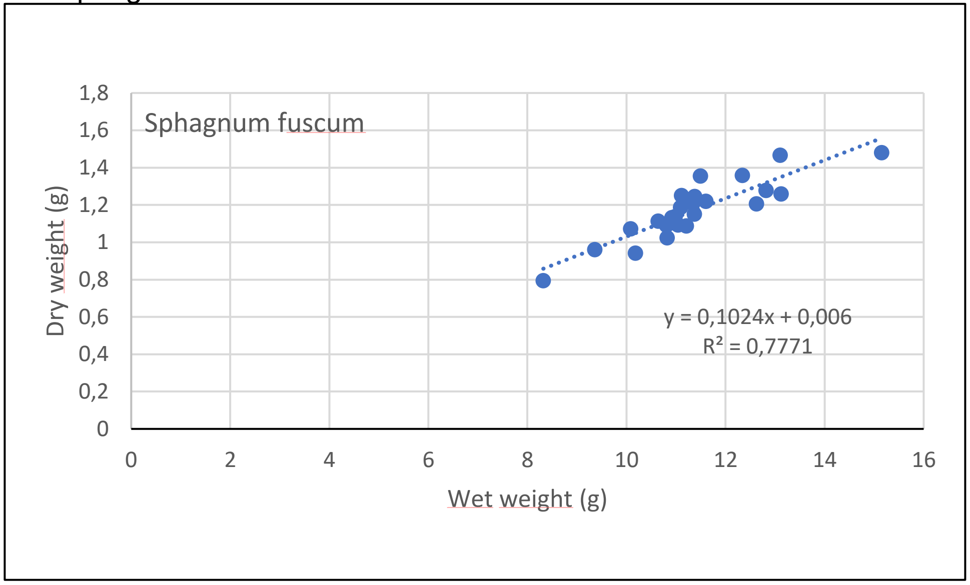


Figure A. Regression line of wet (constant) and dry (oven dried) weights with the regression equation for: A1. Pleurozium schreberi, A2. Sphagnum capillifolium and A3. Sphagnum fuscum

Appendix B: initial peat thickness determination

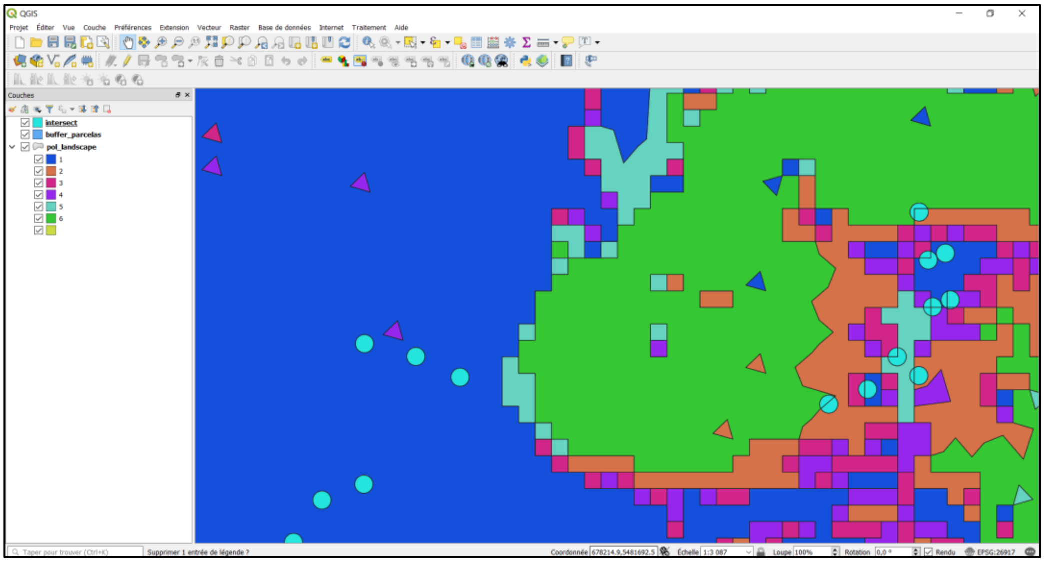
The initial peat thickness data was extracted from the soil organic thickness map of 10 m resolution produced by Laamrani *et al.* (2014).

The map represents the organic layer thickness distribution in 2010 at the landscape scale in the same our study area (49° 28' N, 78° 30' W) in the Clay Belt region of northwestern Quebec (Canada). It is based on a model of the relationship between organic layer thickness (OLT) and topographic variables (slope, aspect, and mean curvature). Six different landscape units (A, B, C, D, E, and F, see table B) were delimited using regression tree analysis. Topographic variables were extracted from a LiDAR-derived digital terrain model. The data and the extraction of the landscape units were used to produce a map with a cell resolution (cell size) of 10 m using ArcGIS 10.0 (Environmental Systems Research Institute [ESRI] 2011. The map showed a good matching (71%) with areas characteristics (Laamrani *et al.*, 2014).

Table B: Summary statistics for OLT by landscape unit for the study area. The landscape units correspond to those obtained by regression tree analysis. n is the number of sites and SD is the standard deviation (Laamrani *et al.*, 2014).

Landscape units	n	Mean (cm)	SD (cm)	Median (cm)
A	543	62	25	60
B	122	56	25	55
C	140	48	21	46
D	117	29	14	25
E	158	43	16	41
F	520	27	11	25

Our plots were overlaid over the landscape units map and retrieved the corresponding thicknesses in 2010 (figure B).



Supprimer 1 entrée de légende ?

Coordonnée 678214.9,5481692.5

Échelle 1:3 087

Loupe 100%

Rotation 0,0 °

☑ Rendu

EPSG:26917

Figure B1: Example to the overlaying of our plots on the organic layer thickness map of Laaramni in QGIS 3. Cycles represent the point layer with a buffer of 10 m and the other geographic elements represent the landscape unit (1, 2, 3, 4, 5 and 6 is A, B, C, D, E and F) with different colors.

However, because controls sites presented the very high degree of paludification, some plots with similar characteristics than harvested sites were selected (filter (Treatment=="Control" & Canopy openness <80)).

We assumed that the harvested sites were not more open like the image in the left.

	
B2a) Example of plot excluded due to canopy openness >80%	B2b) Example of plot retained in the selection

Figure B2: Images of 2 different plots in the control sites.

APPENDICE C – APPENDICES DU CHAPITRE 3

Appendix A: disturbance matrices

Disturbance matrice in the tree component

Fire matrices result in the modeling of fuel consumption for each pool using national forest inventory, fire weather information, and the Boreal Fire Effects model (BORFIRE)(de Groot, 2006; Kurz et al., 2009).

Harvesting matrice was developed by Ministère des Ressources naturelles et de la faune (Québec) and Canadian Forest Service Carbon Accounting Team. In CBM, CPRS matrice represent 97% of the merchantable volume removing. The default fire matrice was used to reflect HSF while the proportion was halved to represent LSF. For CPRS, the default CBM matrice remained unchanged. However, the CPRS matrice was modified for MSP, especially on soil pool, and the default partial cut matrice of 50% of the merchantable biomass harvesting was adjusted to 40%, based on findings from prior studies.

Table A1 : disturbance matrices (fire and sylvicultural practices) about tree component

FIRE MATRICE

Rownames	Columnnames	Default Proportion	Proportion _ HSF	Proportion _LSF
Softwood Merchantable	Softwood Stem Snag	1	1	1
Softwood Foliage	CO2	0,9	0,9	
Softwood Foliage	CH4	0,01	0,01	
Softwood Foliage	CO	0,09	0,09	
Softwood Other	Softwood Branch Snag	0,750002	0,750002	0,79250166
Softwood Other	CO2	0,224998	0,224998	0,18674834
Softwood Other	CH4	0,0025	0,0025	0,002075
Softwood Other	CO	0,0225	0,0225	0,018675
Softwood Submerchantable	Softwood Branch Snag	0,75	0,75	0,7925

Softwood Submerchantable	CO2	0,225	0,225	0,18675
Softwood Submerchantable	CH4	0,0025	0,0025	0,002075
Softwood Submerchantable	CO	0,0225	0,0225	0,018675
Softwood Coarse Roots	Aboveground Fast DOM	0,5	0,5	
Softwood Coarse Roots	Belowground Fast DOM	0,5	0,5	
Softwood Fine Roots	Aboveground Very Fast DOM	0,454948	0,454948	0,45859676
Softwood Fine Roots	Belowground Very Fast DOM	0,5	0,5	0,50401008
Softwood Fine Roots	CO2	0,040546	0,040546	0,03365318
Softwood Fine Roots	CH4	0,000451	0,000451	0,00037433
Softwood Fine Roots	CO	0,004055	0,004055	0,00336565
Aboveground Very Fast DOM	Aboveground Very Fast DOM	0,031467	0,031467	0,56416015
Aboveground Very Fast DOM	CO2	0,87168	0,87168	0,392256
Aboveground Very Fast DOM	CH4	0,009685	0,009685	0,00435825
Aboveground Very Fast DOM	CO	0,087168	0,087168	0,0392256
Belowground Very Fast DOM	Belowground Very Fast DOM	1	1	
Aboveground Fast DOM	Aboveground Fast DOM	0,358407	0,358407	0,71128315
Aboveground Fast DOM	CO2	0,577434	0,577434	0,2598453
Aboveground Fast DOM	CH4	0,006416	0,006416	0,0028872
Aboveground Fast DOM	CO	0,057743	0,057743	0,02598435
Belowground Fast DOM	Belowground Fast DOM	1	1	
Medium DOM	Medium DOM	0,60786	0,60786	0,823537
Medium DOM	CO2	0,352926	0,352926	0,1588167
Medium DOM	CH4	0,003921	0,003921	0,00176445
Medium DOM	CO	0,035293	0,035293	0,01588185
Aboveground Slow DOM	Aboveground Slow DOM	0,909897	0,819794	0,95945365

Aboveground Slow DOM	CO2	0,081093	0,162186	0,03649185
Aboveground Slow DOM	CH4	0,000901	0,001802	0,00040545
Aboveground Slow DOM	CO	0,008109	0,016218	0,00364905
Belowground Slow DOM	Belowground Slow DOM	1	1	

Empty cells signify unchanged values, thus the same as the default values.

CPRS MATRICE

Rownames	Columnnames	Proportion
Softwood Merchantable	Medium DOM	0,03
Softwood Merchantable	Products	0,97
Softwood Foliage	Aboveground Very Fast DOM	1
Softwood Other	Aboveground Fast DOM	1
Softwood Submerchantable	Aboveground Fast DOM	1
Softwood Coarse Roots	Aboveground Fast DOM	0,5
Softwood Coarse Roots	Belowground Fast DOM	0,5
Softwood Fine Roots	Aboveground Very Fast DOM	0,5
Softwood Fine Roots	Belowground Very Fast DOM	0,5
Aboveground Very Fast DOM	Aboveground Very Fast DOM	1
Belowground Very Fast DOM	Belowground Very Fast DOM	1
Aboveground Fast DOM	Aboveground Fast DOM	1
Belowground Fast DOM	Belowground Fast DOM	1
Medium DOM	Medium DOM	1
Aboveground Slow DOM	Aboveground Slow DOM	1
Belowground Slow DOM	Belowground Slow DOM	1
Softwood Stem Snag	Medium DOM	1
Softwood Branch Snag	Aboveground Fast DOM	1
Black Carbon	Black Carbon	1

MSP

Rownames	Columnnames	Proportion
Softwood Merchantable	Medium DOM	0,03
Softwood Merchantable	Products	0,97
Softwood Foliage	Aboveground Very Fast DOM	1
Softwood Other	Aboveground Fast DOM	1
Softwood Submerchantable	Aboveground Fast DOM	1
Softwood Coarse Roots	Aboveground Fast DOM	0,5
Softwood Coarse Roots	Belowground Fast DOM	0,5
Softwood Fine Roots	Aboveground Very Fast DOM	0,5
Softwood Fine Roots	Belowground Very Fast DOM	0,5
Aboveground Very Fast DOM	Aboveground Very Fast DOM	0,27
Aboveground Very Fast DOM	Aboveground Slow DOM	0,73
Belowground Very Fast DOM	Belowground Very Fast DOM	1
Aboveground Fast DOM	Aboveground Fast DOM	0,27
Aboveground Fast DOM	Aboveground Slow DOM	0,73
Belowground Fast DOM	Belowground Fast DOM	1
Medium DOM	Medium DOM	0,27
Medium DOM	Aboveground Slow DOM	0,73
Aboveground Slow DOM	Aboveground Very Fast DOM	0,05
Aboveground Slow DOM	Belowground Very Fast DOM	0,15
Aboveground Slow DOM	Aboveground Slow DOM	0,8
Belowground Slow DOM	Belowground Slow DOM	1
Softwood Stem Snag	Medium DOM	1
Softwood Branch Snag	Aboveground Fast DOM	1

PB

Rownames	Columnnames	Proportion
Softwood Merchantable	Medium DOM	0,078
Softwood Merchantable	CO2	0,1098
Softwood Merchantable	CH4	0,00122
Softwood Merchantable	CO	0,01098
Softwood Merchantable	Products	0,8
Softwood Foliage	Aboveground Very Fast DOM	0,55
Softwood Foliage	CO2	0,405
Softwood Foliage	CH4	0,0045

Softwood Foliage	CO	0,0405
Softwood Other	Aboveground Fast DOM	0,55
Softwood Other	CO2	0,405
Softwood Other	CH4	0,0045
Softwood Other	CO	0,0405
Softwood Submerchantable	Aboveground Fast DOM	0,714
Softwood Submerchantable	CO2	0,2574
Softwood Submerchantable	CH4	0,00286
Softwood Submerchantable	CO	0,02574
Softwood Coarse Roots	Aboveground Fast DOM	0,5
Softwood Coarse Roots	Belowground Fast DOM	0,5
Softwood Fine Roots	Aboveground Very Fast DOM	0,284
Softwood Fine Roots	Belowground Very Fast DOM	0,5
Softwood Fine Roots	CO2	0,1944
Softwood Fine Roots	CH4	0,00216
Softwood Fine Roots	CO	0,01944
Aboveground Very Fast DOM	Aboveground Very Fast DOM	0,568
Aboveground Very Fast DOM	CO2	0,389
Aboveground Very Fast DOM	CH4	0,039
Aboveground Very Fast DOM	CO	0,004
Belowground Very Fast DOM	Belowground Very Fast DOM	1
Aboveground Fast DOM	Aboveground Fast DOM	0,568
Aboveground Fast DOM	CO2	0,3888
Aboveground Fast DOM	CH4	0,00432
Aboveground Fast DOM	CO	0,03888
Belowground Fast DOM	Belowground Fast DOM	1
Medium DOM	Medium DOM	0,785
Medium DOM	CO2	0,1935
Medium DOM	CH4	0,00215
Medium DOM	CO	0,01935
Aboveground Slow DOM	Aboveground Slow DOM	1
Belowground Slow DOM	Belowground Slow DOM	1
Softwood Stem Snag	Medium DOM	0,785
Softwood Stem Snag	CO2	0,1935
Softwood Stem Snag	CH4	0,00215
Softwood Stem Snag	CO	0,01935
Softwood Branch Snag	Aboveground Fast DOM	0,568

Softwood Branch Snag	CO2	0,3888
Softwood Branch Snag	CH4	0,00432
Softwood Branch Snag	CO	0,03888

PC

Rownames	Columnnames	Default Proportion	Proportion_CP40
Softwood Merchantable	Softwood Merchantable	0.5	0,6
Softwood Merchantable	Medium DOM	0.1	0,1
Softwood Merchantable	Products	0.4	0,3
Softwood Foliage	Softwood Foliage	0.5	0,6
Softwood Foliage	Aboveground Very Fast DOM	0.5	0,4
Softwood Other	Softwood Other	0.7	0,6
Softwood Other	Aboveground Fast DOM	0.3	0,4
Softwood Submerchantable	Softwood Submerchantable	0,7	0,56
Softwood Submerchantable	Aboveground Fast DOM	0,3	0,44
Softwood Coarse Roots	Softwood Coarse Roots	0.5	0,6
Softwood Coarse Roots	Aboveground Fast DOM	0.25	0,2
Softwood Coarse Roots	Belowground Fast DOM	0.25	0,2
Softwood Fine Roots	Softwood Fine Roots	0.5	0,6
Softwood Fine Roots	Aboveground Very Fast DOM	0.25	0,2
Softwood Fine Roots	Belowground Very Fast DOM	0.25	0,2
Aboveground Very Fast DOM	Aboveground Very Fast DOM	1	1
Belowground Very Fast DOM	Belowground Very Fast DOM	1	1
Aboveground Fast DOM	Aboveground Fast DOM	1	1
Belowground Fast DOM	Belowground Fast DOM	1	1
Medium DOM	Medium DOM	1	1
Aboveground Slow DOM	Aboveground Slow DOM	1	1
Belowground Slow DOM	Belowground Slow DOM	1	1
Softwood Stem Snag	Medium DOM	0.5	0,4
Softwood Stem Snag	Softwood Stem Snag	0.5	0,6
Softwood Branch Snag	Aboveground Fast DOM	0.5	0,4
Softwood Branch Snag	Softwood Branch Snag	0.5	0,6

Table A2 : Disturbance matrice of silvicultural practice effects about moss component

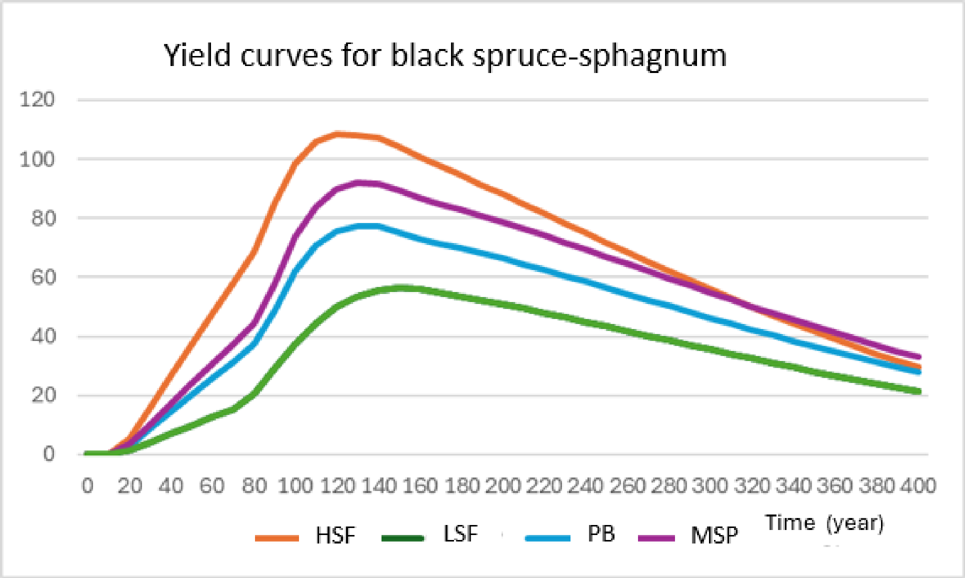
We used previous empirical studies to reflect effect of silvicultural practice on moss component: Kpodo et al,2012 ; Lafleur et al , 2016 , Mayer et al, 2018 ; Fenton and Bergeron, 2007, ;Nave 2010; Botroh et al, 2023 ; Renard et al, 2016.

Parameters	Description	CPRS	PC	MSP	PB
Prop_Cburned_in_fire_Fmlive	proportion of living Feather moss carbon burned to atmosphere	0	0	0	1
Prop_Cburned_in_fire_Splive	proportion of living Sphagnum moss carbon burned to atmosphere	0	0	0	0,716
Prop_Cburned_in_fire_FMfast	proportion of Feather moss DOM fast carbon burned to atmosphere	0	0	0	0,46
Prop_Cburned_in_fire_SPfast	proportion of sphagnum moss DOM fast carbon burned to atmosphere	0	0	0	0,46
Prop_residuel_livefeather	proportion of Feather moss live biomass non impacted by disturbance, remaining after disturbance	0,446	0,93	0,317	0
Prop_residuel_livesphag	proportion of Sphagnum moss live biomass non impacted by disturbance, remaining after disturbance	0,379	0,93	0,46	0,284
Prop_unburned_FM_DOM_FAST	proportion of Feather moss DOM fast carbon unburned (1-prop_Cburned)	0	0	0	0,46
Prop_unburned_SP_DOM_FAST	proportion of Sphagnum moss DOM fast carbon unburned (1-prop_Cburned)	0	0	0	0,46
Prop_residuel_FM_DOM_FAST	proportion of Feather moss live biomass non impacted by disturbance, early remaining in the pool, at year disturbance	0,8	0,93	0,705	0,272
Prop_residuel_SP_DOM_FAST	proportion of Sphagnum moss live biomass non	0,8	0,93	0,705	0,415

	impacted by disturbance , really remaining in the pool, at year disturbance				
Decay_multiplier for uppermost organic layer	proportion to multiplier decay after fire before steady state of forest, short time here 15 years	1,22	1,22	2,33	1,64
Decay_multiplier for deeper organic layer	proportion to multiplier decay after fire before steady state of forest, short time here 15 years	1	1	2,33	1

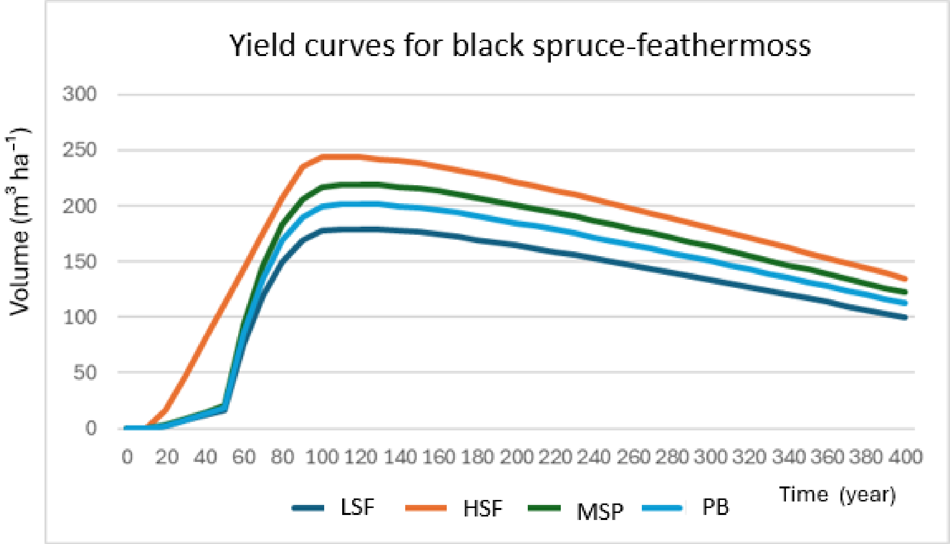
Appendix B : Yield curves

A)



LSF = CPRS = PC

B)



LSF = CPRS = PC

Figure B1 : Yield curves according disturbance in paludified stand (Black spruce-sphagnum dominated stand) and non paludified stand (Black spruce-feathermoss dominated stand). LSF: low severity fire, HSF: high severity fire, CPRS: cut with protection of regeneration and soils, MSP: mechanical site preparation, PB: prescribed burning, PC: partial cut. LSF, CPRS and PC have the same yield curve.

Appendix C : Other results

Stand level

-Total ecosystem stocks tree and moss

Table C1: Summary of total C stocks (Mg C ha⁻¹ ,final and initial C stocks, mean and standard deviation, percent reduction final and initial), according to tree and moss total ecosystem stocks per disturbance scenario

Forest_type	**Disturbance**	**val_initiale**	**val_finale**	**mean**	**SD**	**reduction_percent**
BSSFM	init HSF current LSF	194.3402	180.0354	156.7114	30.067540	-7.36
BSSFM	init HSF current HSF	194.3402	195.3555	171.5858	32.740882	0.52
BSSFM	init HSF current CPRS_UP	194.3402	173.0619	149.5321	32.976737	-10.95
BSSFM	init HSF current MSP_UP	194.3402	186.5520	154.3196	43.362531	-4.008
BSSFM	init HSF current PB_UP	194.3402	179.2139	147.6704	41.614784	-7.783
BSSFM	init HSF current PC_UP	194.3402	125.1075	142.9094	10.959974	-35.624
BSSP	init LSF current LSF	224.0991	247.5538	215.2805	24.955573	10.466
BSSP	init LSF current HSF	224.0991	249.4286	222.6627	28.398565	11.303
BSSP	init LSF current CPRS_P	224.0991	252.7188	223.9808	23.412910	12.771

BSSP	init LSF current MSP_P	224.0991	247.5397	224.3969	23.710757	10.46
BSSP	init LSF current PB_P	224.0991	238.3559	213.6217	23.377952	6.362
BSSP	init LSF current PC_P	224.0991	231.4614	222.9815	5.535132	3.285

Table C2 : Summary of accumulation rates(Mg C ha⁻¹ year-1) , based on final and initial C stocks, according to tree and moss total ecosystem stocks per disturbance scenario

Disturbance	Accumulation rate BSSP	Accumulation rate BSFM
LSF	0.0938187	-0.0572192
HSF	0.1013176	0.0040609
CPRS	0.1144785	-0.0851134
MSP	0.0937623	-0.0311530
PB	0.0570271	-0.0605055
PC	0.0294490	-0.2769311

LIVING BIOMASS - Reduction percent % (final-initial)/initial

Table C3 : Reduction percent % (final-initial)/initial in living biomass

Disturbance	Reduction percent BSSP	Reduction percent BSFM
LSF	-4.278908	-26.252974
HSF	24.704388	-10.522429

CPRS	-5.946689	-27.464240
MSP	17.004138	-17.341650
PB	6.561034	-21.707563
PC	-64.075183	-61.394381

High severity disturbance led to higher C accumulation in biomass. HSF>MSP>PB>LSF>=CPRS>PC OK for all forest type. Here, positive or negative value is due to the initial value of tree. in BSSP the gain is due to the lower productivity of previous stand have while in BSFM previous stand have an initial higher biomass.

ORGANIC - Reduction percent % (final-initial)/initial

Table C4 : Reduction percent % (final-initial)/initial in organic soil layer

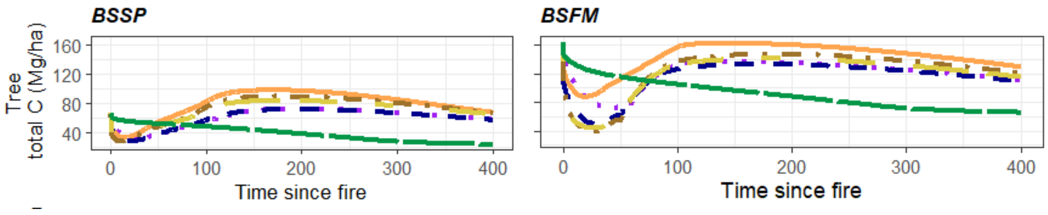
Disturbance	Reduction percent BSSP	Reduction percent BSFM
LSF	13.502649	28.093641
HSF	5.024306	20.431132
CPRS	16.866375	19.350940
MSP	6.769284	21.000849
PB	4.317465	18.118929
PC	21.510899	9.299129

In BSSP, PC, CPRS and LSF have the greater C accumulation in organic layer and PB, HSF and MSP the lowest respectively (PC>CPRS>LSF>MSP>HSF>PB). PC the highest and PB the lowest

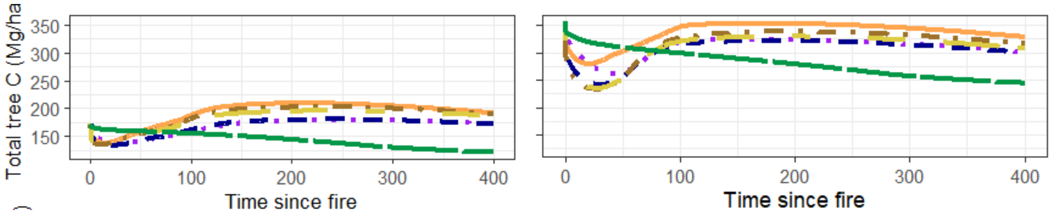
In BSFM, LSF>HSF>CPRS>PB>PC. LSF the highest due to moss component and PC the lowest.

Details of C stocks in tree and moss

Total tree without mineral soi



with mineral soil



Total moss

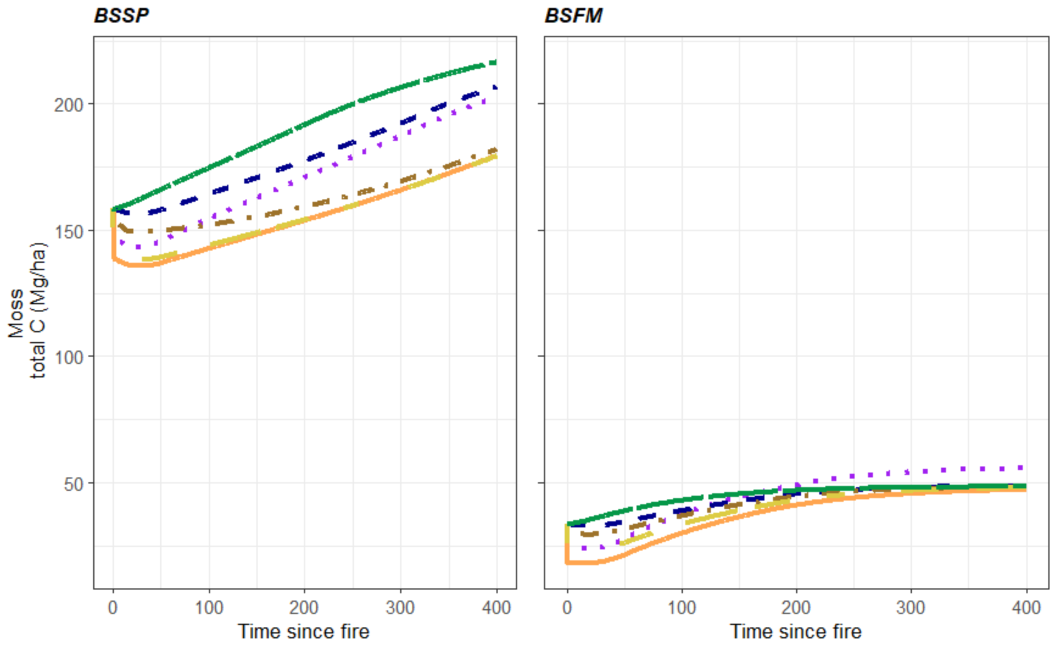
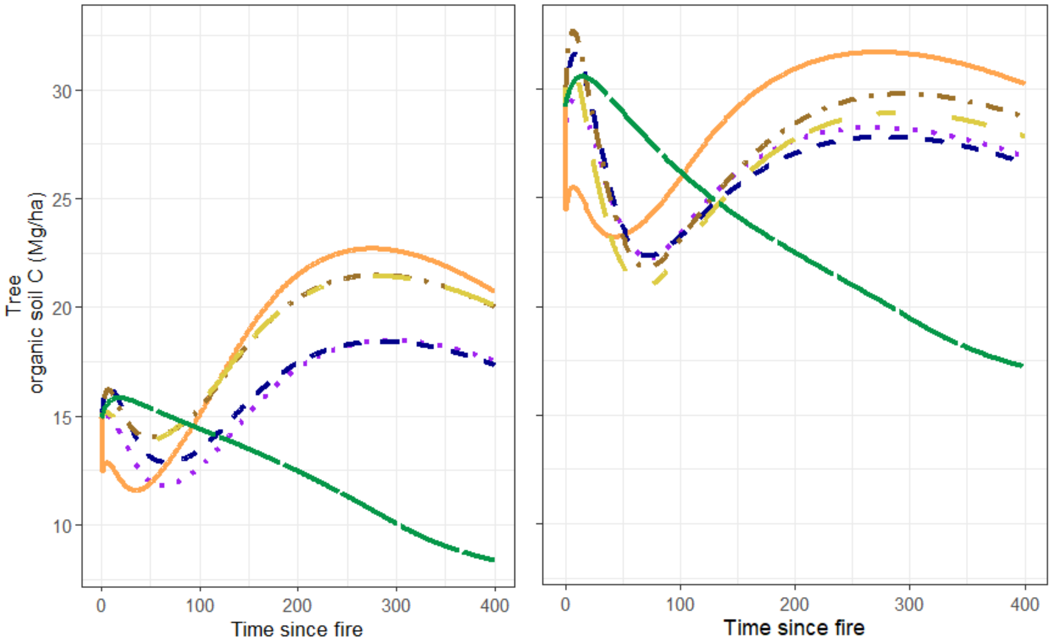


Figure C1 : Stand level total ecosystem carbon stocks (Mg C ha^{-1}, in tree and moss separately) with time since disturbance (400 years) for the two stand types (BSSP and BSFM) and the five disturbance scenario (LSF, HSF, CPRS, MSP, PC). HSF in orange, LSF in purple, CPRS in blue, MSP +plantation in brown, PB +plantation in yellow, PC in green.

Organic soil

Tree organic soil



Moss organic

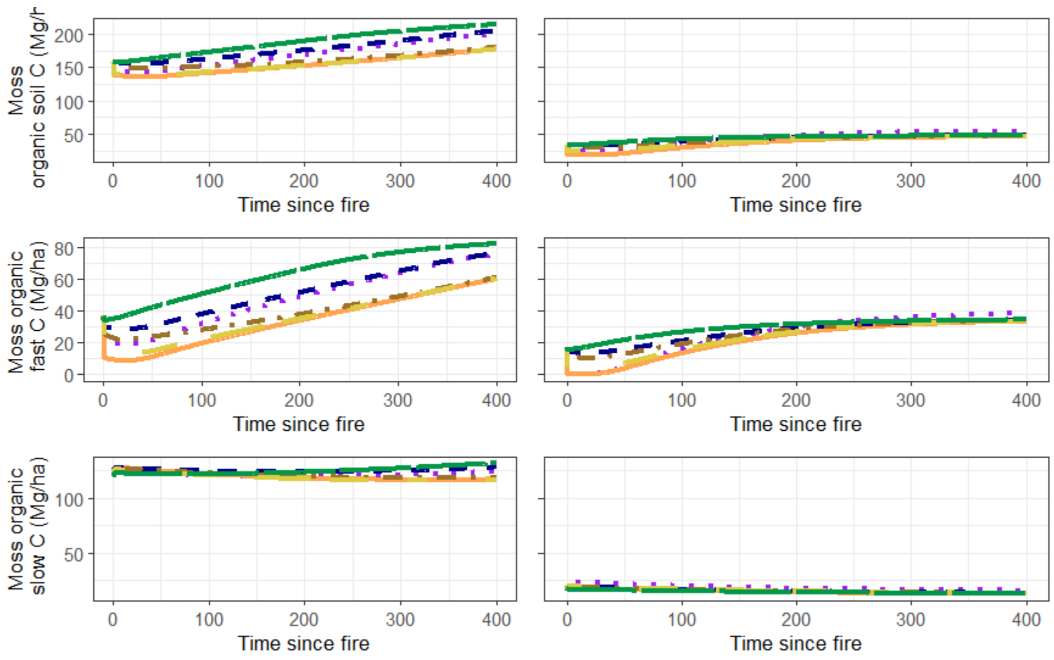


Figure C2 : Stand level organic soil layer carbon stocks (Mg C ha-1 ,in tree and moss separately) with time since disturbance (400 years) for the two stand types (BSSP and BSFM) and the five disturbance scenarios (LSF, HSF, CPRS, MSP, PC). HSF in orange, LSF in purple, CPRS in blue, MSP +plantation in brown, PB +plantation in yellow , PC in green.

Other presentation of C stocks - stand level results

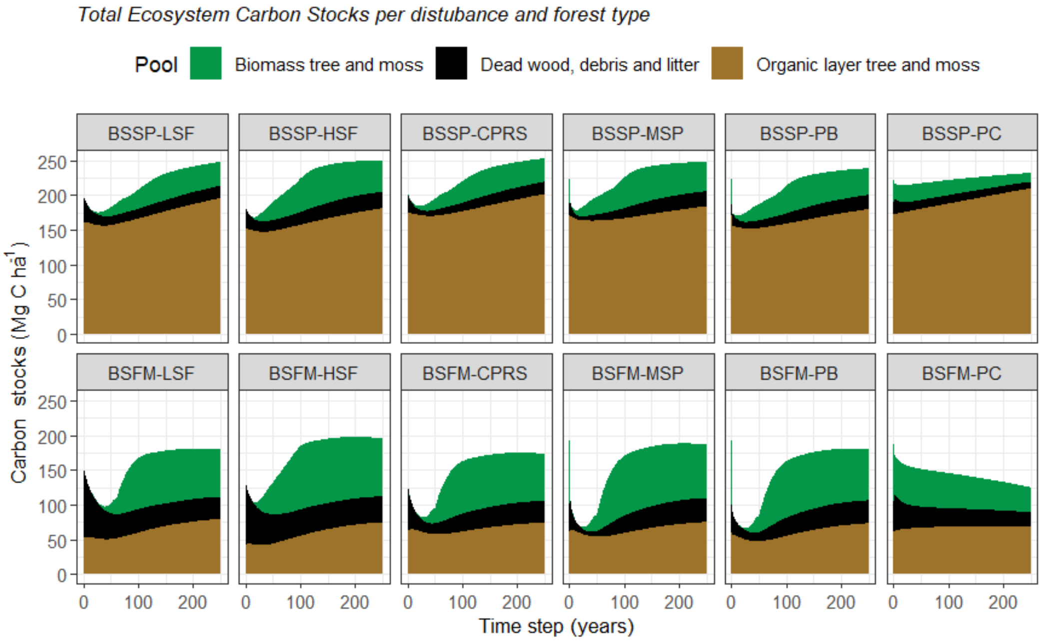


Figure C3 : Stand level total ecosystem carbon stocks (Mg C ha-1 , live biomass in green moss and tree, + litter + debris+ dead wood in black, organic soil layer moss and tree in brown) ; with time since disturbance (250 years) for the two stand types (BSSP and BSFM) and the five disturbance scenario (LSF, HSF, CPRS, MSP, PC).

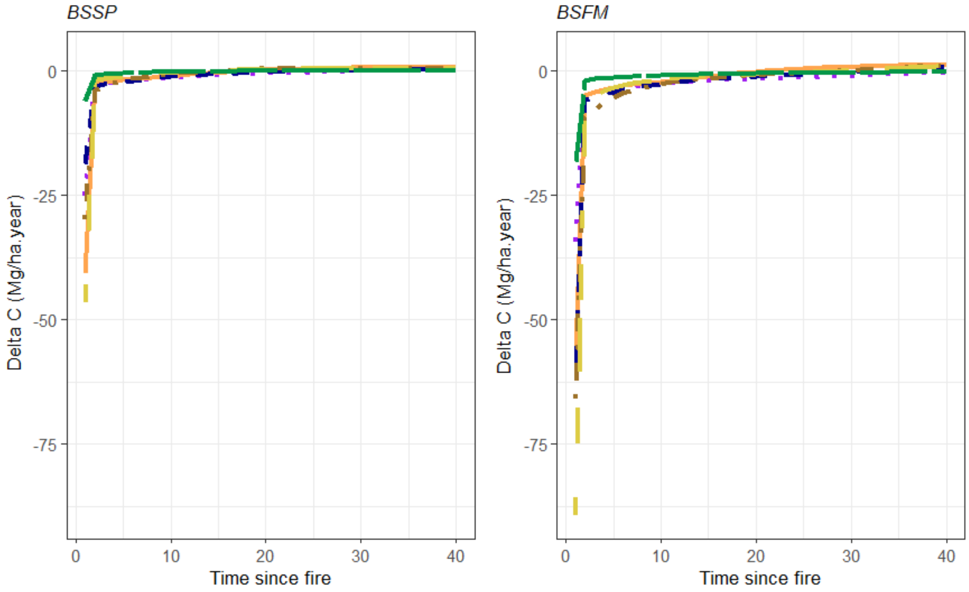
-DELTA ECOSYSTEM

Table C1: Summary of total carbon balance (Mg C ha-1 year-1 = 100g C m^{-2} y^{-1}) per disturbance scenario: carbon balance at disturbance year - min_value, maximum value, time f maximum value the first positive value and the delay to switch to a sink for all tree compartment and moss compartment (live biomass and moss-derived C) contributions.

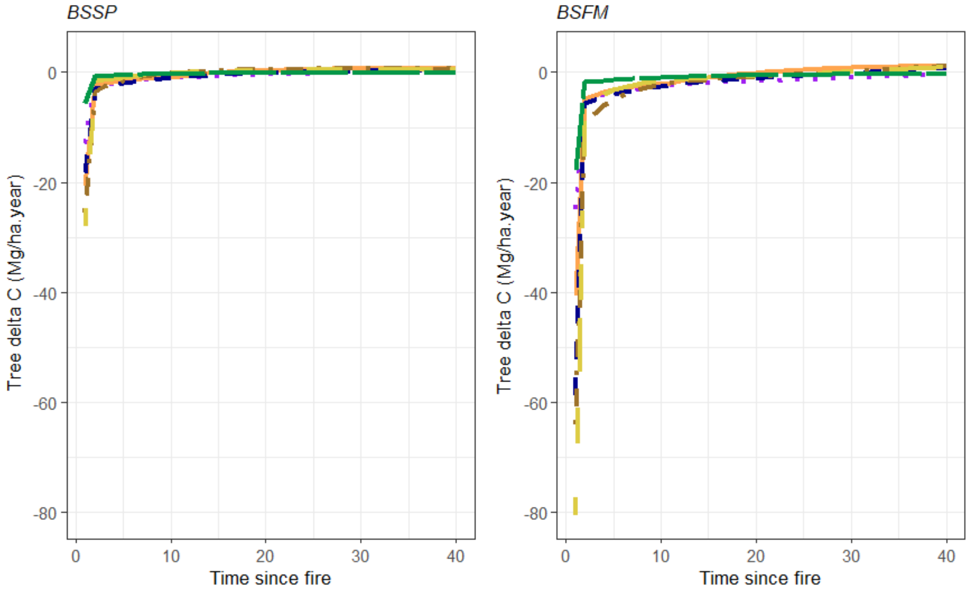
Disturbance	**Forest_type**	**min_value**	**max_time_step**	**max_value**	**first_positive_time_step**	**positive_val**
init LSF current LSF	BSSP	-25.199	100	0.626	30	0.011
init LSF current HSF	BSSP	-40.7	100	0.9	17	0.042
init LSF current CPRS	BSSP	-18.6	90	0.613	24	0.029
init LSF current MSP	BSSP	-27.5	100	0.85	15	0.048
init LSF current PB	BSSP	-46.5	100	0.781	14	0.022
init LSF current PC	BSSP	-5.9	41	0.106	17	0.033
init HSF current LSF	BSSFM	-34.3	61	2.562	42	0.03
init HSF current HSF	BSSFM	-55.9	40	1.243	21	0.041
init HSF current CPRS	BSSFM	-58.97	51	2.686	31	0.001
init HSF current MSP	BSSFM	-64.995	51	3.265	28	0.004
init HSF current PB	BSSFM	-89.396	51	3.0360	29	0.05323
init HSF current PC	BSSFM	-18.039	0	0.0000000	NA	NA

Delta figure

Tree and moss 1 to 40 years, initial step was excluded



In tree 1 to 40 years, initial step was excluded



In moss on 1 to 20 years, initial step was excluded

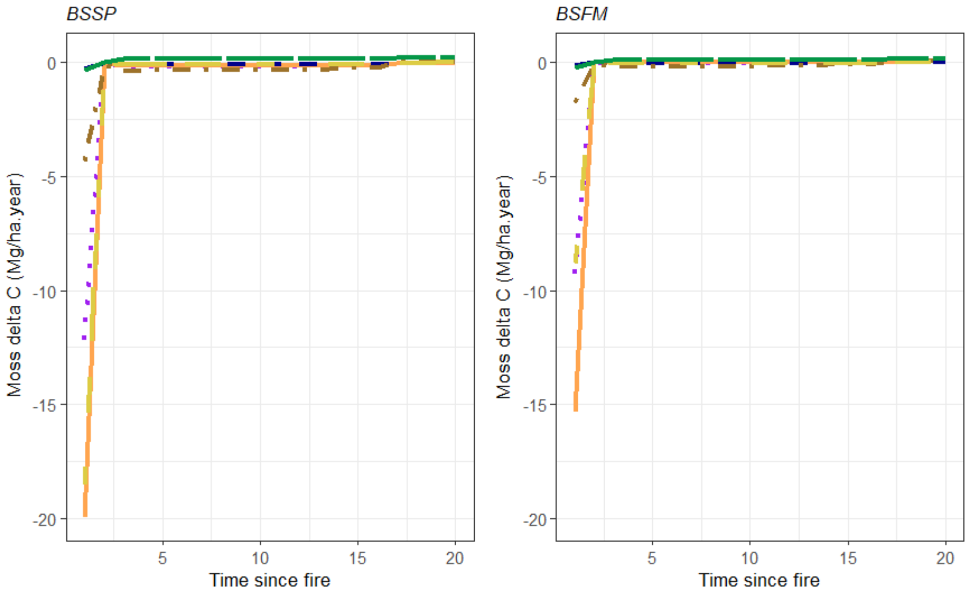
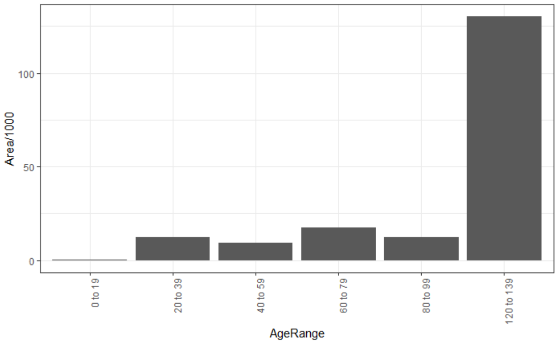


Figure C4: C balance (Mg C ha-1 year-1 = 100g C m⁻² y⁻¹) over the first 40 years, initial step was excluded, for the two stand types (BSSP and BSFM) and the five disturbance scenarios (LSF, HSF, CPRS, MSP, PC). HSF in orange, LSF in purple, CPRS in blue, MSP +plantation in brown, PB +plantation in yellow , PC in green.

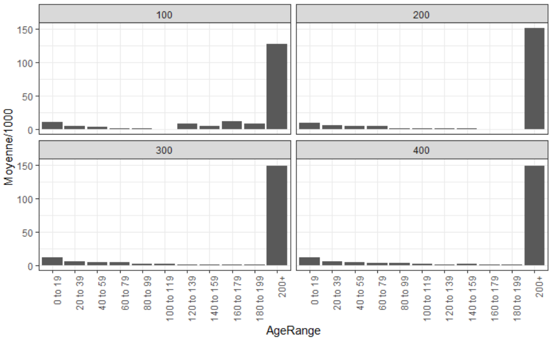
Landscape level

Age classes distribution

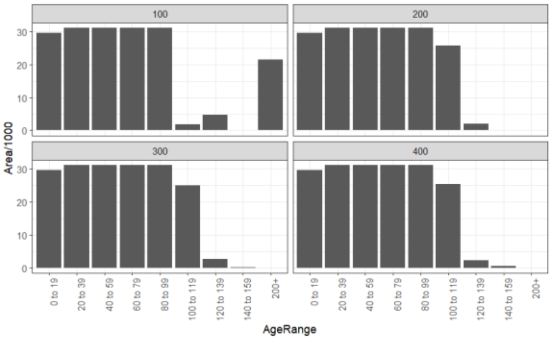
Initial time step



400 years fire cycle baseline



current even-age management or business as usual (BAU)



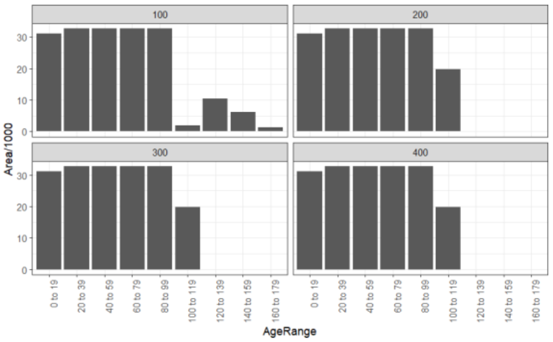
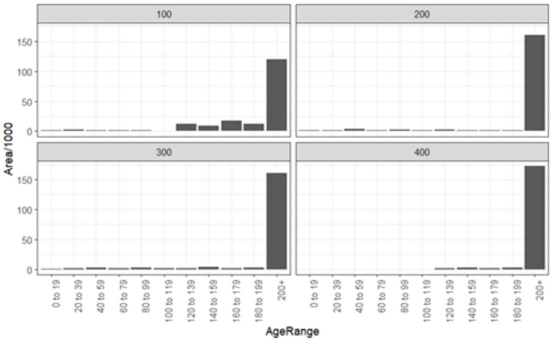
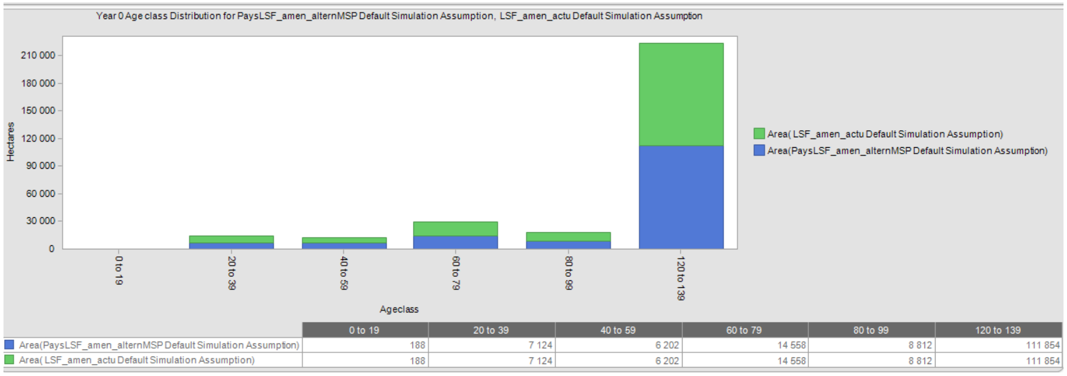
MSP Alternative	CP Alternative
	

Figure C5: Age-class distributions resulting from 300 years of simulations under 400-year fire cycle and three management scenarios (usual or current , MSP-alternative and PC-alternative) on the same inventory.

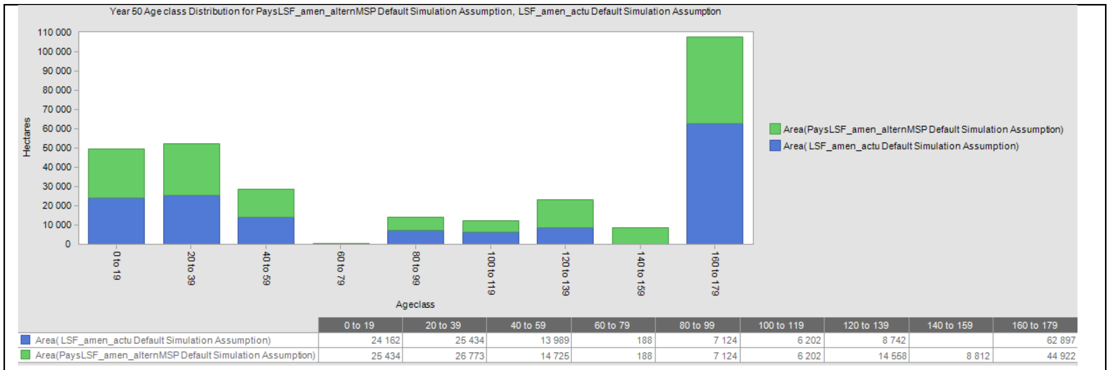
Example of age classes distribution for the highest part of landscape for MSP and current even-age management or business as usual (BAU)

Time step 0



	0 to 19	20 to 39	40 to 59	60 to 79	80 to 99	120 to 139
🟦 Area(PaysLSF_amen_alternMSP Default Simulation Assumption)	188	7 124	6 202	14 558	8 812	111 854
🟩 Area(LSF_amen_actu Default Simulation Assumption)	188	7 124	6 202	14 558	8 812	111 854

Time step 50

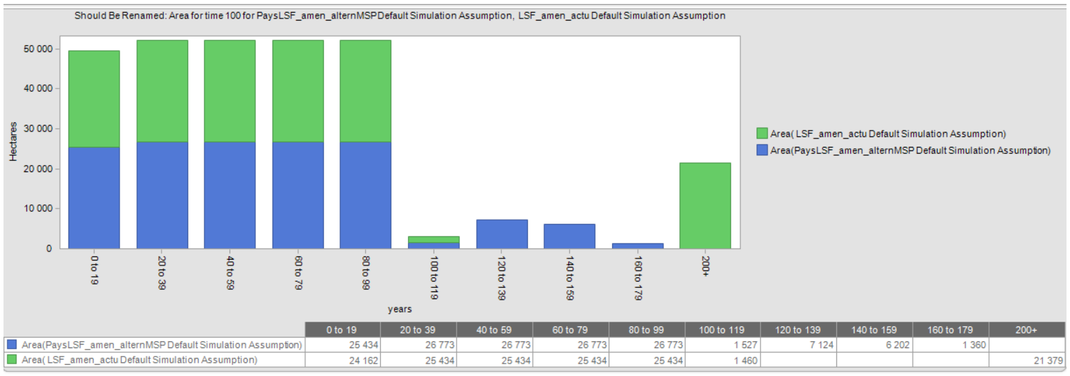


Year 50 Age class Distribution for PaysLSF_amen_alternMSP Default Simulation Assumption, LSF_amen_actu Default Simulation Assumption

	0 to 19	20 to 39	40 to 59	60 to 79	80 to 99	100 to 119	120 to 139	140 to 159	160 to 179
🟦 Area(LSF_amen_actu Default Simulation Assumption)	24 162	25 434	13 989	188	7 124	6 202	8 742		62 897
🟩 Area(PaysLSF_amen_alternMSP Default Simulation Assumption)	25 434	26 773	14 725	188	7 124	6 202	14 558	8 812	44 922

More productive mature stand 120-150 in MSP than current even-age management or business as usual (BAU)

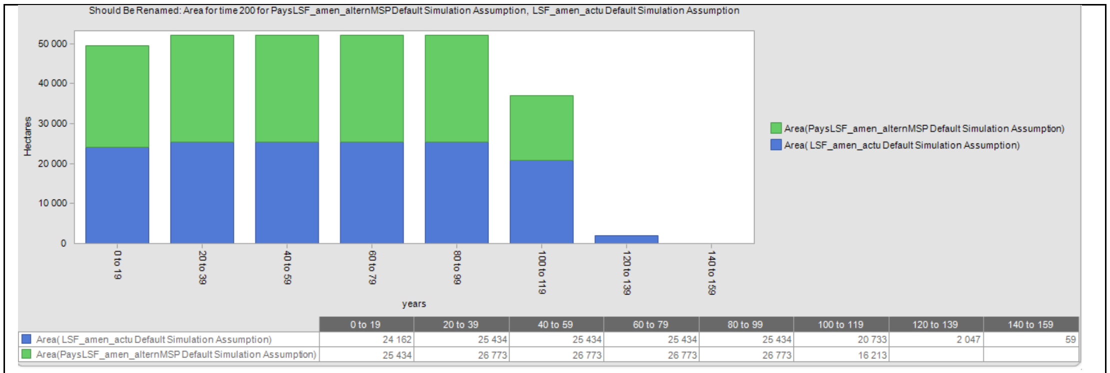
Time step 100



Should Be Renamed: Area for time 100 for PaysLSF_amen_alternMSP Default Simulation Assumption, LSF_amen_actu Default Simulation Assumption

	0 to 19	20 to 39	40 to 59	60 to 79	80 to 99	100 to 119	120 to 139	140 to 159	160 to 179	200+
🟦 Area(PaysLSF_amen_alternMSP Default Simulation Assumption)	25 434	26 773	26 773	26 773	26 773	1 527	7 124	6 202	1 360	
🟩 Area(LSF_amen_actu Default Simulation Assumption)	24 162	25 434	25 434	25 434	25 434	1 460				21 379

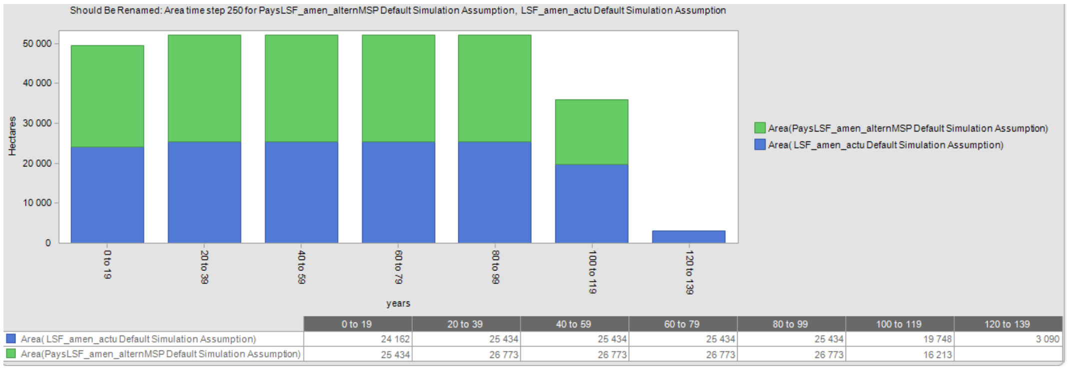
Tiem step 200



	0 to 19	20 to 39	40 to 59	60 to 79	80 to 99	100 to 119	120 to 139	140 to 159
🟦 Area(LSF_amen_actu Default Simulation Assumption)	24 162	25 434	25 434	25 434	25 434	20 733	2 047	59
🟩 Area(PaysLSF_amen_alternMSP Default Simulation Assumption)	25 434	26 773	26 773	26 773	26 773	16 213		

More mature 100-140 year in amen BAU than amen alterna MSP

Time step 250



	0 to 19	20 to 39	40 to 59	60 to 79	80 to 99	100 to 119	120 to 139
🟦 Area(LSF_amen_actu Default Simulation Assumption)	24 162	25 434	25 434	25 434	25 434	19 748	3 090
🟩 Area(PaysLSF_amen_alternMSP Default Simulation Assumption)	25 434	26 773	26 773	26 773	26 773	16 213	

Here, more than 10-140 additional mature-age sites were present in the current management scenario compared to the MSP scenario, which may explain the higher carbon balance observed at the end of the simulation period in the current management scenario relative to the MSP-alternative.

Figure B6: Age-class distributions at different time steps (initial, 50, 100, 200 and 250) under even- management scenarios: usual or current and MSP-alternative on the same inventory (82% of landscape).

Emission

Table C2: Landscape total C emission (MtC; 10^{12} gC or 10^{6} tC) on 250 year-simulation period : initial, final and average and SD standard deviation per disturbance scenario (400-year fire cycle and three management scenarios (usual or current , MSP-alternative and PC-alternative)).

Scenario	val_initial	val_final	average	SD
amen_BAU	0	0.0651966	0.0624507	0.0084596
amen_alternatif_MSP	0	0.0471620	0.0554252	0.0159271
amen_alternatif_CP	0	0.0653322	0.0612031	0.0065798
LSF	0	0.0848294	0.0700201	0.0103441
HSF	0	0.0841765	0.0715355	0.0114608

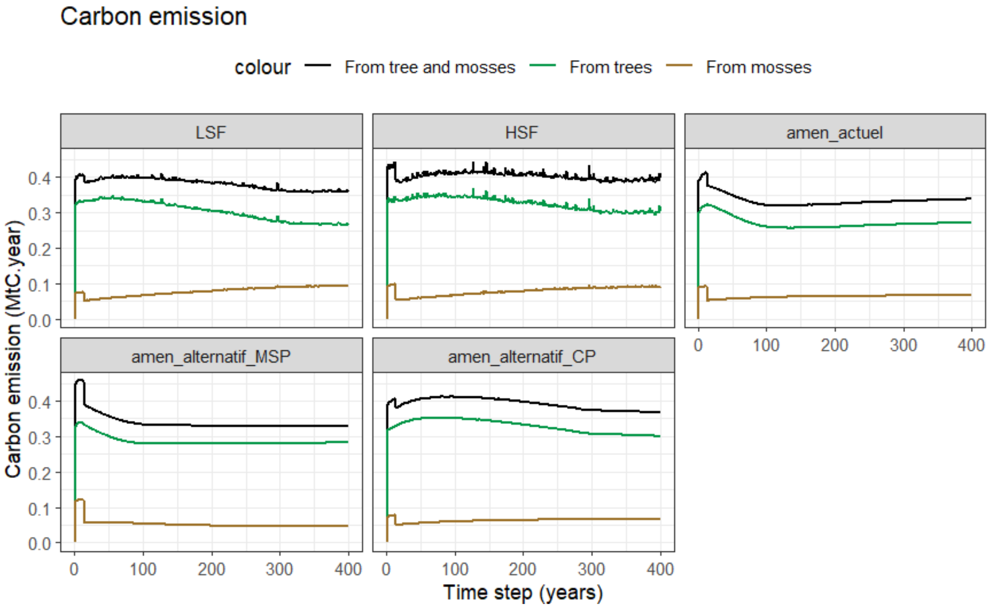


Figure C7: *The landscape carbon emission over 250-year simulation period (MtC year^{-1}; 10^{12} gC or 10^{6} tC by year) in 400-year fire cycle scenario (occurrence of LSF and occurrence of HSF) and in three management scenarios (usual or current, MSP-alternative and PC-alternative)*: *tree contribution in green, moss in brown, and combined moss and tree in black*.

GLOSSAIRE OU LEXIQUE

Terme	**Définition**
Stocks	La quantité absolue de carbone contenue dans un réservoir à un moment donné (GIEC, 2000).
Flux	Les transferts de carbone d'un réservoir à un autre constituent des flux de carbone (GIEC, 2000).
Séquestration de C	La séquestration naturelle de carbone consiste à « absorber » ou à « emmagasiner » le carbone dans un puits ou un réservoir de carbone à long terme.https://savoirs.usherbrooke.ca/bitstream/handle/11143/9637/Meas_Michel_Vararoath_MEnv_2016.pdf?sequence=1
Zones mésiques	zones bien ou modérément bien drainées, selon le Système canadien de classification des sols (Comité d'experts de la prospection pédologique d'Agriculture Canada, 1983)), le sol est un podzol ferro-humique à humique recouvert d'une couche organique d'une profondeur moyenne de 26 cm
Zones humides	Drainage moyen à imparfait, le sol est organique avec une couche organique moyenne de 125 cm. (Giasson et al, 2006)
Cycle de feu	Encore appelé intervalle de retour de feu, le temps requis pour le feu pour brûlé toute une surface, ou une zone.
Fréquence de feu	Surface brûlée par an
Aménagement équienne	vise à maintenir des peuplements composés d’arbres qui ont sensiblement le même âge. Dans ce type de peuplements, les traitements sylvicoles sont appliqués en même temps et uniformément à tous les arbres.
Aménagement inéquienne	vise à maintenir des peuplements composés d’arbres d’âges différents et souvent d’espèces différentes. Dans ce type de peuplements, les traitements sylvicoles ne sont pas appliqués en même temps et de la même façon à tous les arbres.

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