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EFFECTS OF CLIMATE CHANGE ON FIRE FOR A DECIDUOUS FOREST LANDSCAPE IN TÉMISCAMINGUE, QUÉBEC

THÈSE
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PAR
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AVANT-PROPOS

Ce mémoire est soumis sous forme de quatre articles scientifiques. Dans tous les cas, j'étais la personne responsable du design d'expérimentation, de la récolte et la compilation de données, des analyses statistiques, et de la rédaction.

Le premier article était : « How is ecological resilience relevant for natural disturbance-based forest management? ». Les autres auteurs étaient : Garry Peterson, Christian Messier, Yves Bergeron, et Mike Flannigan. Cet article a été publié en 2006 dans le Revue Canadienne de Recherche Forestière (volume 36, pages 2285-2299).

Le deuxième article était : « Fire and canopy species composition in the Great Lakes-St. Lawrence forest of Témiscamingue, Québec ». Les autres auteurs étaient : Christian Messier, Yves Bergeron, et Frédérik Doyon. Cet article a été publié en 2006 dans le revue Forest Ecology and Management (volume 231, pages 27-37).

Le troisième article était : « Fire and the relative roles of weather, climate and landscape characteristics in the Great Lakes-St. Lawrence forest of Canada ». Les autres auteurs étaient : Mark C. Drever, Christian Messier, Yves Bergeron, et Mike Flannigan. Cet article a été accepté en mai 2007 par la revue Journal of Vegetation Science.

Le quatrième et dernier article était : « Effects of climate change on fire for a deciduous forest landscape in Témiscamingue, Québec ». Les autres auteurs étaient : Mark C. Drever, Christian Messier, Yves Bergeron, et Mike Flannigan. Cet article a été soumis au journal Applied Vegetation Science.

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RÉSUMÉ

Le paradigme de la gestion durable des forêts demande aux gestionnaires de s'inspirer des perturbations naturelles lors de l'application des coupes forestières tant en fréquence qu'en répartition spatiale dans le paysage. Mais est-ce que ce paradigme peut vraiment maintenir la résilience écologique des forêts sous aménagement? La théorie écologique suppose qu'une gestion basée sur le régime des perturbations naturelles pourrait maintenir la résilience si les stratégies forestières maintiennent les patrons et les processus qui perpétuent les états désirés à l'intérieur de leur fourchette de variabilité naturelle tout en réduisant ceux qui augmentent la résilience des états indésirables. Mettre en application ces idées dans un contexte de gestion exige cependant une articulation soignée des états d'écosystème en considération ainsi que des perturbations et des stress qui affectent la persistance des états alternatifs possibles. De plus, ces états doivent être caractérisés aux échelles spatiales et temporelles pertinentes à leur expression afin de traduire ceux-ci en modalités de gestion adéquatement ajustés à ces échelles.

Dans cette thèse, je pose la question suivante : comment les changements climatiques affecteront-t-ils la fréquence des grands feux dans les forêts feuillues du Témiscamingue dans le Québec méridional ? J'examine cette question en faisant le lien entre la résilience des écosystèmes forestiers et le régime des perturbations naturelles dans un contexte de changements climatiques. Dans ce système, les états alternatifs considérés sont des peuplements de feuillus tolérants, des peuplements dominés par des pins et ceux dominés par des espèces pionnières où la principale perturbation naturelle considérée est le feu. Au tout début, je fais dans le premier chapitre une revue de la littérature concernant la résilience écologique. Après, j'analyse l'histoire du feu au Témiscamingue, avec une emphase sur son influence sur la composition forestière dans le paysage; j'étudie ensuite un gradient régional de feux afin de développer des modèles prédictifs en fonction de variables du climat ; et finalement, j'estime les effets du changement de climat au Témiscamingue sur la végétation forestière dans un contexte dynamique avec le régime de feu en employant les modèles prédictifs du feu utilisant les sorties d'un modèle climatique global de circulation.

Le deuxième chapitre me permet de reconstituer l'historique des grands feux au Témiscamingue à l'aide d'archives provinciales sur les feux, de l'interprétation de photos aériennes anciennes et de données dendrochronologiques. Sur la base de cette information, la fréquence du feu est estimée et l'hypothèse selon laquelle le temps écoulé depuis le dernier feu est une cause déterminante sur la composition en arbres est testée. En dépit de sa proximité à la forêt boréale mixte, ce paysage brûle relativement rarement, avec plus de 60% du paysage n'ayant pas brûlé depuis les dernières 413 années. Le cycle global du feu estimé, une évaluation du temps requis pour brûler une aire de taille équivalente au secteur d'étude, est de 494 ans (IC de 95% : 373-694 ans). Des analyses multivariées ont permis de distinguer des assemblages distincts en espèces d'arbre selon le temps écoulé depuis le dernier feu :

les assemblages de *Populus-Pinus* dominent la canopée lorsque le temps écoulé depuis le dernier feu est court alors que des assemblages dominés par l'érable à sucre, le bouleau jaune et la pruche de l'est dominent lorsque temps écoulé depuis le dernier feu est long. Comparativement aux autres variables écologiques examinées, le temps écoulé depuis le dernier feu est celle qui explique le plus la composition forestière. Ces résultats suggèrent aussi que les plus longs cycles de feu observés récemment devraient favoriser une augmentation de la proportion des espèces évitant le feu dans le paysage, avec des conséquences potentiellement négatives sur la résilience de l'écosystème si un tel phénomène favorise l'exclusion locale d'espèces adaptées au feu.

Pour comprendre le rôle que joue le climat sur le système forêt-feu de la forêt feuillue, les caractéristiques climatiques, humaines et biophysiques du paysage ont été mises en relation avec l'occurrence des grands feux et les superficies brûlées pour tout le territoire Grands Lacs/Fleuve St-Laurent du Canada. Cette évaluation a été faite en (i) caractérisant les grands feux (> 200 ha) récents (1959-1999) dans 26 paysages et (ii) en analysant ces données dans le cadre de la théorie de l'information que pour comparer six hypothèses concernant les rôles des conditions météorologiques propices aux incendies de forêt, impliquant les normales de climat, les densités de population et de route, et les caractéristiques écologiques telles que les dépôts de surface et la présence des coupe-feu. Trois cents quatre-vingt-douze grands feux ont brûlé 833.698 ha pendant la période d'étude, brûlant annuellement en moyenne 0.07% ± 0.42% (± écart-type) de la superficie forestière des paysages. L'activité du feu était fortement saisonnière, avec la plupart des feux se produisant en mai et juin. Une combinaison 1) des précipitations de l'hiver précédent, 2) du déficit ou du surplus en précipitation pendant la saison de feu et 3) du pourcentage de paysage couvert par les dépôts de surface bien drainées explique le mieux l'occurrence des feux et la superficie brûlée. L'occurrence du feu change seulement en fonction des variables du climat, tandis que la superficie brûlée est également expliquée par l'importance du tremble et du pin dans le couvert forestier, la densité de population humaine et deux caractéristiques durables du paysage, soit la superficie occupée par de grands plans d'eau et celle occupée par les dépôts fluvioglaciaires.

Ces résultats peuvent aider à concevoir des stratégies d'adaptation pour les augmentations prévues de l'occurrence des conditions météorologiques propices aux feux sévères, surtout dans l'ouest de la région. Ils permettent aussi de mettre en priorité les paysages selon les caractéristiques durables mentionnées ci-dessus et donnent des indications sur les modalités de gestion à définir dans un contexte de contrôle des effets du feu sur les ressources forestières et le maintien de l'intégrité écologique.

Basé sur le modèle développé au chapitre 3, le dernier chapitre présente comment les changements climatiques pourraient affecter la dynamique forêt-feu au Témiscamingue. Cette évaluation est réalisée en trois étapes. D'abord, j'identifie le rôle relatif de différentes variables des conditions météorologiques propices au feu en expliquant l'occurrence des grands feux et la superficie brûlée à travers la forêt de la

zone de végétation des Grands Lacs/Fleuve St-Laurent de l'est du Canada. En second lieu, j'examine comment ces variables météorologiques ont changé historiquement au Témiscamingue et, troisièmement, comment ils peuvent changer selon différents scénarios de changements climatiques selon le modèle global canadien de circulation. Au Témiscamingue, les moyennes des températures maximales mensuelles pendant la saison de feu (mai à octobre) et des périodes sèches ont expliqué le mieux l'occurrence du feu et la superficie brûlée. Depuis 1910, les températures moyennes mensuelles maximales sont restées stationnaries au Témiscamingue tandis que les périodes sèches sont devenues moins fréquentes. Chacun des trois scénarios de changement climatique montre une augmentation des températures maximales mensuelles moyennes et une diminution des périodes sèches pendant le 21ème siècle, combinaison impliquant une augmentation faible des superficies annuelles brûlées. En dépit de cette augmentation, et étant donné que les coupes forestières affectent des superficies plusieurs fois plus grandes que celles affectées par les feux, les effets du changement de climat sur le feu n'affecteront probablement pas la structure et la composition des forêts autant que la foresterie, la succession ou les perturbations naturelles telles que le chablis. La résilience des peuplements dominés par les pins diminuera probablement tandis que celle des peuplements dominés par les feuillus tolérants augmentera.

Mots-clés : résilience écologique; forêt feuillue; Témiscamingue; feu; changements climatiques.

INTRODUCTION GENERALE

0.1 RÉSUMÉ

Le feu, ainsi que d’autres perturbations naturelles, sont des facteurs critiques dans la création et le maintien de la diversité biologique et structurale des écosystèmes de la forêt feuillue. Les feux sont fonction des facteurs structuraux et compositionnels (par exemple, la continuité verticale et horizontale des carburants, l'abondance et la distribution des conifères), des facteurs topographiques (par exemple, l’abondance et la distribution des coupe- feu telles que les lacs, les marais et les villes) et des facteurs météorologiques (par exemple, les précipitations, l'humidité relative, la température et la vitesse du vent). Cela dit, il est reconnu qu’une variable fortement déterminante du type, de la sévérité et de la fréquence des feux est le climat. Il semble donc probable que des changements futurs du climat provoqueront des modifications importantes dans le régime du feu au cours des prochaines décennies. Ces modifications peuvent être plus importantes pour la composition et la structure des paysages forestiers que les effets directs des précipitations et des changements de température. L'effet des changements climatiques sur le régime du feu sera probablement fortement variable dans le temps et l'espace à travers le Canada et le Québec. Par exemple, la fréquence du feu pourrait augmenter pour une grande partie des forêts de l’ouest du Canada, alors que l’on observera l’inverse pour des régions de l’est du Québec et du Canada. Les changements climatiques affecteront possiblement deux aspects importants de l'écologie du feu : la longueur de la saison de feux et l'abondance de la foudre et, du même coup, la fréquence de l'allumage des feux de forêt. Au coeur de la recherche décrite ci-dessous réside l’objectif suivant : évaluer comment les changements climatiques affecteront la fréquence des grands feux dans les forêts feuillues caduques du Témiscamingue dans le Québec méridional.

0.2 LE FEU ET LA FORET FEUILLUE

Le feu, ainsi que d’autres perturbations naturelles, sont des facteurs critiques dans la création et le maintien de la diversité biologique et structurale des écosystèmes forestiers (Pickett et White 1985). Le type, le patron, la sévérité, la fréquence et l'hétérogénéité interne des feux sont fonction du climat à long terme et à l'échelle de la région (Carcaillet et al. 2001; Bergeron et Archambault 1993; Clark 1988). Plusieurs auteurs ont avancé que les changements climatiques - définis ici comme étant les changements du climat attribuables au moins en partie aux augmentations anthropiques des concentrations atmosphériques des gaz à effet de serre - auront des conséquences importantes sur les régimes des feux de forêt (Weber et Flannigan 1997; Overpeck et al. 1990).

Contrairement à la forêt boréale, les feux vastes et sévères dans les forêts feuillues tempérées sont généralement peu fréquents, avec des cycles qui varient entre 1400 et 2500 ans (Frelich 2002). Ces feux sont aussi de faible intensité (Doyon et Sougavinski 2002). Le feuillage de la forêt feuillue est relativement peu inflammable, ne constituant pas un carburant favorable pour les feux de couronne (Patterson III et al. 1983). Quoique la plupart soient de faible intensité, les feux y jouent tout de même un rôle écologique important. Par exemple, suite au passage d'un feu, l'abondance de végétation de sous-bois telle que les arbustes et les herbacées diminue, de même que la régénération pre-établié (Wright et Bailey 1982). Cette diminution de l'abondance réduit la compétition et mène à l'augmentation soudaine de la disponibilité des ressources dans le sol (Wright et Bailey 1982). Ainsi, ce sont les individus qui survivent au feu ou qui s'établissent rapidement qui sont favorisés.

Beaucoup de facteurs influencent le comportement et les impacts du feu dans une forêt feuillue. Parmi ceux-ci, il y a les facteurs structuraux et compositionnels (par exemple, la continuité verticale et horizontale des carburants, l'abondance et la distribution des conifères (Lesieur et al. 2002 ; Hély et al. 2000)), les facteurs topographiques (par exemple, l'abondance et la distribution des coupe- feu tels que les lacs, les marais et les villes (Larsen 1997; Bergeron 1991)) et les facteurs météorologiques (par exemple, les précipitations, l'humidité relative, la température et la vitesse du vent (Dale et al. 2001; Carcaillet et al. 2001; Weber et Flannigan 1997)).

De tous ces facteurs, ce sont les facteurs météorologiques qui sont les plus importants (Carcaillet et al. 2001), étant donné que la majorité des feux progressent surtout lorsque les conditions météorologiques sont propices (Flannigan et Harrington 1988).

0.3 LE FEU ET LES CHANGEMENTS CLIMATIQUES

Il semble probable que des changements futurs du climat provoqueront des modifications importantes dans le régime des feux au cours des prochaines décennies (Overpeck et al. 1990). Ces modifications peuvent être plus importantes pour la composition et la structure des paysages forestiers que les effets directs des précipitations et des changements de température (Loehle 2000; Weber et Flannigan 1997). D'ailleurs, bien que le feu soit un processus naturel, une augmentation des superficies brûlées peut contribuer à l'augmentation des concentrations atmosphériques de gaz à effet de serre (Amiro et al. 2001; Weber et Stocks 1998; McAlpine 1998).

Plusieurs auteurs ont étudié les effets du réchauffement global et des concentrations accrues en gaz à effet de serre sur le régime du feu au Canada et au Québec. Pour ce faire, certains auteurs ont utilisé des données d'archives et/ou dendrochronologiques pour lier les conditions climatiques des siècles précédents à la fréquence des feux afin de prévoir comment les futurs changements de climat peuvent modifier les régimes des feux (Podur et al. 2002; Bergeron et al. 2001; Johnson et al. 1999). Une autre méthode consiste à utiliser des indices forêt-météo pour comprendre comment le risque d'incendie change en fonction des futures conditions climatiques prévues par un ou plusieurs modèles globaux du climat (MGC) (Flannigan et al. 2001; Weber et stocks 1998). Dans l'ensemble, ces études montrent des rapports climat-feu qui sont beaucoup plus complexes qu'un simple lien direct entre les températures annuelles moyennes plus élevées et l'augmentation de la fréquence des feux, tel que postulé dans les premières études portant sur les changements climatiques et le feu (Flannigan et van Wagner 1991; Overpeck et al. 1990; Clark 1988).

L'effet des changements climatiques sur le régime du feu sera probablement fortement variable dans le temps et l'espace à travers le Canada et le Québec. Par exemple, la fréquence du feu pourrait augmenter pour une grande partie des forêts de l'ouest du Canada, alors que l'on observera l'inverse pour des régions de l'est du Québec et du Canada (Weber et stocks 1998; Bergeron et Archambault 1993). La différence dans ces prévisions est principalement due aux variations des quantités actuelles et prévues des précipitations à travers les différentes régions du pays.

Au Québec, plusieurs études effectuées dans la région de l'Abitibi montrent que toute augmentation de la fréquence du feu liée à une augmentation de la température pourrait être neutralisée par une augmentation de l'abondance ou un changement dans la distribution temporelle des précipitations (Bergeron et al. 2001; Bergeron 1998; Bergeron et Flannigan 1995; Bergeron et Archambault 1993; Bergeron 1991). Ainsi, les changements climatiques mèneront à une réduction de l'incidence des feux de forêt pour cette région (Bergeron 1998; Stocks et al. 1998). Cela confirme que la quantité de précipitation - en particulier la succession des événements de précipitation et de sécheresse - est un facteur déterminant de la fréquence des feux de forêt au Canada (Flannigan et Harrington 1988). Le secteur du Témiscamingue méridional est transitoire entre les secteurs où une diminution de la fréquence du feu est prévue et ceux où une augmentation est prévue (Flannigan et al. 2001).

0.4 LA SAISON DES FEUX ET L'ABONDANCE DE LA FOUDRE

Les changements climatiques affecteront possiblement deux aspects importants de l'écologie du feu : la longueur de la saison du feu et l'abondance de la foudre. La saison des feux, la période de l'année où le risque de feu est le plus élevé, augmentera probablement en longueur. Dans un scénario où le taux de gaz carbonique doublerait ($2XCO_2$), la saison du feu au Québec pourrait augmenter de près de 27 jours (Wotton et Flannigan 1993) et commencer plus tôt au printemps et se prolonger plus tard à l'automne (Stocks et al. 1998). Cela est important pour la forêt feuillue car c'est à la fin du printemps et à l'automne, quand les arbres n'ont pas leurs feuilles, que cette forêt est la plus susceptible aux feux. Donc, l'effet des changements climatiques dépendra aussi de la modification de la phénologie du débourrage et de la sénescence des feuilles vis-à-vis des changements des conditions météorologiques.

Les changements climatiques peuvent avoir des conséquences importantes sur l'intensité et la fréquence de la foudre et, du même coup, sur la fréquence de l'allumage des feux de forêt (Bergeron et al. 2001; Flannigan et al. 2000). Au Canada, seulement 35 % des feux sont issus de la foudre mais ils représentent 85 % de la surface totale brûlée (Weber et Stocks 1998). Les résultats de MGC indiquent qu'une augmentation des concentrations atmosphériques en CO_2 peut augmenter l'incidence d'orages électriques dans l'hémisphère nordique (Price et Rind 1994).

0.5 OBJECTIFS DE LA THESE

Cette thèse touche à différentes disciplines - écologie forestière, écologie du paysage, sylviculture et météorologie - afin de soulever certaines des lacunes importantes dans nos connaissances sur la façon dont les changements climatiques peuvent affecter la structure et la composition des paysages de la forêt feuillue. Le type de forêt d'intérêt principal est l'érablière à bouleau jaune du Québec et, pour ce qui est des perturbations, ce sont les grands feux. Au coeur de la recherche décrite ci-dessous réside l'objectif suivant : évaluer comment les changements climatiques affecteront la fréquence des grands feux dans les forêts feuillues caduques du Témiscamingue dans le Québec méridional.

Cette étude sera essentielle aux gestionnaires et aux chercheurs forestiers cherchant à mettre de l'avant une gestion durable de la forêt sous un climat changeant. Par exemple, savoir comment les changements climatiques affecteront le feu peut aider à identifier des types de forêts susceptibles de disparition ou qui seront considérablement réduits. De plus, une telle étude peut aider à prioriser les efforts de recherche et conduire à des techniques de gestion qui augmentent la résilience d'écosystème face aux changements climatiques. Cette étude peut également être utile pour identifier les types de forêts où la résilience est plus élevée. En outre, ce travail peut aider à formuler des stratégies de gestion inspirées par des régimes des perturbations naturelles, particulièrement celles qui causent le remplacement des peuplements, et fournir des conseils sur la façon de diversifier les pratiques dans ce paysage feuillu.

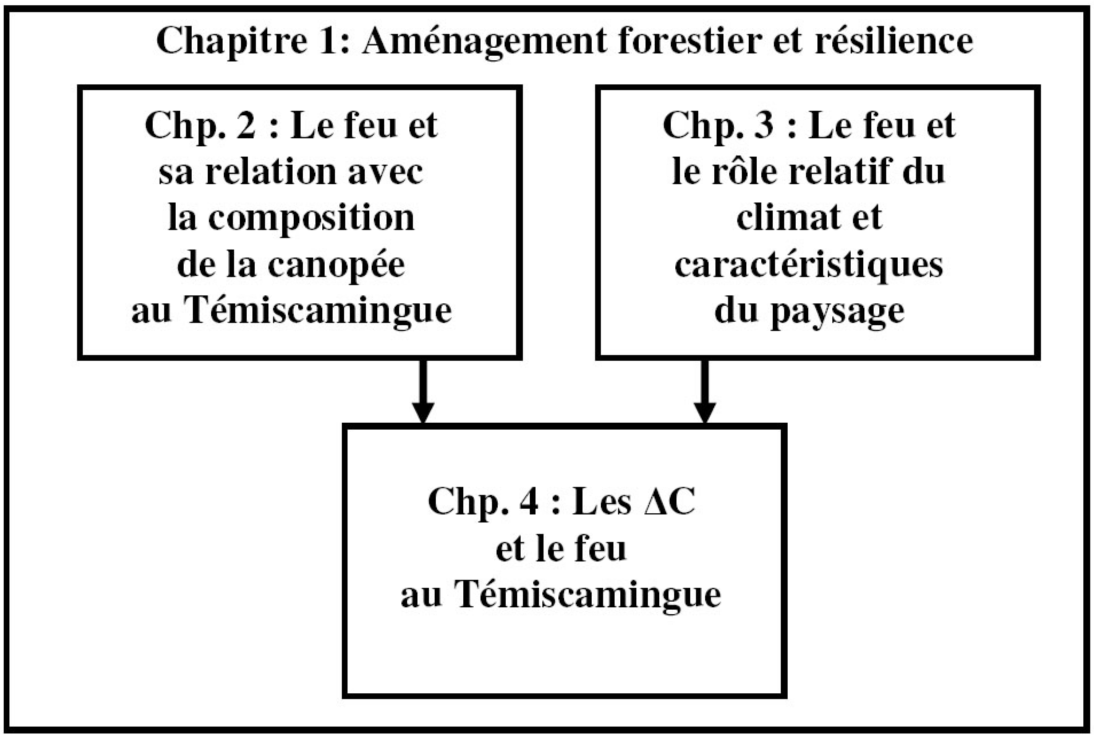
Ce projet mesure les effets possibles des changements climatiques sur le feu en employant une substitution espace-temps, où le climat historique dans différents secteurs le long d'un gradient régional donne de l'information sur les relations climat-feu. Je commencerai par une revue de littérature sur la résilience écologique et sur ce que signifie ce concept pour la gestion durable des forêts. La deuxième partie de ce projet est une caractérisation de l'histoire des feux aussi bien que la structure et la composition de la forêt feuillue au Témiscamingue méridional. Troisièmement, j'examinerai un gradient régional à travers le biome forestier des Grands Lacs/Fleuve St-Laurent des conditions climatiques liées au feu. Finalement, j'intègrerai ce qui précède en employant les régressions déterminées dans la deuxième partie pour prévoir les effets possibles des changements climatiques sur le feu et les superficies brûlés pour le paysage feuillu du Témiscamingue.

0.6 LES LIENS ET LA STRUCTURE DE LA THÈSE

Les quatre chapitres de cette thèse présentent quatre travaux qui sont indépendants mais reliés (Figure 0.1). Le premier chapitre, une revue de la littérature au sujet de la résilience écologique, fournit le cadre conceptuel et théorique dans lequel les résultats et les implications de la recherche dans les trois autres chapitres sont considérés. Le deuxième chapitre, une évaluation de l'histoire du feu au Témiscamingue, fournit des informations nécessaires concernant le cycle du feu, les rapports entre la composition de la canopée et le feu et, de façon plus importante, une évaluation de la superficie brûlée annuellement par des grands feux. Le troisième chapitre fournit l'évidence que le climat est la variable explicative la plus importante derrière l'occurrence et la superficie brûlée dans la région des Grands-Lacs/Fleuve St-Laurent du Canada. Le dernier chapitre synthétise les données et les rapports ci-mentionnés afin d'essayer de prévoir la superficie brûlée annuellement en raison du changement de climat au Témiscamingue.

0.7 FIGURES

Figure 0.1 Liens entre les chapitres de la thèse.



CHAPITRE 1

REVUE CRITIQUE DE LA LITTÉRATURE PERTINENTE

CAN NATURAL DISTURBANCE-BASED FOREST MANAGEMENT MAINTAIN ECOLOGICAL RESILIENCE?

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1.1 ABSTRACT

Given the increasingly global stresses on forests, many ecologists recommend managers sustain ecological resilience, the capacity of ecosystems to absorb disturbances without undergoing fundamental change. In this review we ask: can the emerging paradigm of natural disturbance-based management (NDBM) maintain ecological resilience in managed forests? Applying resilience theory requires careful articulation of the ecosystem state in consideration, the disturbances and stresses that affect persistence of possible alternative states, and the spatial and temporal scales of management relevance. Implementing NDBM while maintaining resilience means recognizing that: (i) biodiversity is important for long-term ecosystem persistence, (ii) natural disturbances play a critical role as a generator of structural and compositional heterogeneity at multiple scales, and (iii) traditional management tends to produce forests more homogeneous than those disturbed naturally and increase the likelihood of unexpected catastrophic change by constraining variation of key environmental processes. NDBM may maintain resilience if silvicultural strategies retain the structures and processes that perpetuate desired states while reducing those that enhance resilience of undesirable states. Such strategies require understanding harvesting impacts on slow ecosystem processes, such as seed bank or nutrient dynamics, which in the long term can result in ecological surprises by altering the forest's capacity to reorganize following disturbance.

Keywords: Ecological resilience; Natural disturbance-based forest management; Alternative stable states; Fire; Biodiversity; Sustainability; Adaptive cycle.

1.2 RESUME

Étant donné l'augmentation globale des stress dans les forêts, plusieurs écologistes croient que les gestionnaires doivent maintenir la résilience écologique, c'est-à-dire la capacité des écosystèmes à absorber les perturbations sans subir de changements importants. Dans cette revue, nous nous demandons : Le nouveau paradigme de l'aménagement basé sur les perturbations naturelles, peut-it maintenir la résilience écologique dans les forêts aménagées? L'application de la théorie de la résilience exige une articulation minutieuse de l'état de l'écosystème considéré, des perturbations et des stress qui affectent la persistance d'états alternatifs potentiels ainsi que les échelles spatiales et temporelles de la pertinence de l'aménagement. L'application de l'aménagement basé sur les perturbations naturelles, tout en maintenant la résilience, oblige à reconnaître que (i) la biodiversité est importante pour la persistance à long terme de l'écosystème, (ii) les perturbations naturelles jouent un rôle crucial dans la genèse de l'hétérogénéité de la structure et de la composition à multiples échelles et (iii) l'aménagement traditionnel tend à rendre les forêts plus homogènes que celles qui sont naturellement perturbées et à augmenter les chances de changements catastrophiques inattendus en réduisant la variation de processus environnementaux clés. L'aménagement basé sur les perturbations naturelles peut maintenir la résilience si les stratégies sylvicoles permettrent de conserver les structures et les processus qui perpétuent les états désirés tout en réduisant ceux qui favorisent la résilience des états indésirables. De telles stratégies exigent de comprendre les impacts de la récolte sur les processus écosystémiques lents, tel que la banque de graines ou la dynamique des nutriments, qui peuvent causer des surprises à long terme en modifiant la capacité de la forêt à se réorganiser après une perturbation.

Mots clés: résilience écologique; aménagement basé sur les perturbations naturelles; états stables alternatifs ; feu; biodiversité; durabilité; cycle adaptif.

1.3 INTRODUCTION

The structure and composition of forests are increasingly influenced by global phenomena such as introduced species, climate change or anthropogenic alterations of biogeochemical cycles (Newman 1995; Vitousek et al. 1997; Simberloff 2000; Dale et al. 2001; Honnay et al. 2002). Major transformations of forest ecosystems have occurred or are underway, in many cases resulting in substantial losses of timber values or commercial extinction of valuable tree species along with a reduction in native biodiversity (Liebhold et al. 1995; Laurance 1999; Simberloff 2000; Hoekstra et al. 2005). In addition to these global influences, forests are also being shaped by regional and local forces. In Canada, for example, the area logged per year doubled between 1960 (~500,000 $ha \cdot yr^{-1}$) and 1995 (~1 M $ha \cdot yr^{-1}$) and, in many years, harvesting has replaced fire as the dominant disturbance in productive forests (World Resources Institute 2000). Moreover, forests often experience the cumulative and often fragmentary impacts of development pressures such as roads, oil and gas development, urban and suburban encroachment, and agriculture (World Resources Institute 2000; Forman 2000; Trombulak and Frissell 2000; Schneider et al. 2003). This combination of disturbance and developmental pressures can reduce biodiversity and diminish the capacity for forests to continue providing the same quantity and quality of ecological goods and services in perpetuity (Toman and Ashton 1996; Costanza et al. 2000).

In response to these concerns, many ecologists recommend that resource managers focus on maintaining ecological resilience, defined as the capacity of natural systems to absorb disturbances without undergoing change to a fundamentally different state (Holling 1973; Holling 1986; Peterson et al. 1998). The recommendation is that rather than setting specific and sustained targets such as allowable annual cuts or a minimum amount of wildlife habitat, managers should seek to enhance the resilience of ecosystem states deemed essential to the provision of ecological goods and services while decreasing the resilience of states that do not provide these or that do so at low levels (Gunderson and Holling 2002). Here we ask:

can natural disturbance-based management maintain or confer ecological resilience in managed forests? We review the concepts inherent in such a question, drawing principally from examples in North American forests, and present ideas on how to anticipate and avoid potential ecological surprises that can arise when goods and services are extracted from forest ecosystems on a sustained basis. We then recommend how managers can integrate resilience into their silvicultural regimes in the context of natural disturbance-based forest management. While our recommendations are most applicable to large tracts of forest managed for traditional forest values such as timber or wildlife habitat, we believe they also apply to other types of tenures such as wood lots or community forests.

1.4 REVIEW OF CONCEPTS

Ecological resilience

Ecological resilience is the capacity of an ecosystem to absorb disturbance and undergo change while still maintaining its essential functions, structures, identity and feedbacks (Holling 1973; Peterson et al. 1998; Walker et al. 2004). Resilience is often synonymous with adaptive capacity i.e. the ability of a system to reconfigure itself in the face of disturbance or stresses without significant decreases in critical aspects such as productivity or composition (Gunderson 2000; Carpenter et al. 2001). Resilience is an emergent property of ecosystems that may be estimated by the magnitude of disturbance a system can absorb without undergoing significant transformational change (Gunderson 2000; Walker et al. 2004). For example, resilience can be estimated by the maximum frequency of intense fires or drought that aspen parkland can experience before conversion to grassland. This definition of resilience therefore assumes the existence of alternative system states (e.g. parkland vs. grassland) and is primarily concerned with (i) how changes in the structure of an ecosystem can alter its behaviour, and (ii) how a given state persists over time (Holling 1973; Gunderson 2000; Walker et al. 2004). Alternatively, resilience may be measured as the probability a given state will persist over the time period of interest (Peterson 2002a). Whether resilience is desirable or not depends both on the values desired from the system and on its current state (Carpenter et al. 2001). For instance, a fragmented patch of forest composed principally of exotic species with low social utility may be an undesirable but highly resilient state, or alternatively, a productive monoculture of trees may be an economically desirable state with low resilience to root disease or other pathogens.

Unfortunately, the multiple meanings given to ecological resilience in the literature have created confusion and hindered application of a concept that is inherently somewhat inexact (Beisner et al. 2003; Walker et al. 2004). To estimate resilience, it is necessary to specify several qualitative characteristics. These include: the state(s) and spatial scale of the system being considered (resilience *of* what), the perturbations of interest that affect the persistence of system states (resilience *to* what), and the temporal scale of interest (Carpenter et al. 2001; Walker et al. 2004). The temporal scale depends on relevant aspects of the system of interest, such as turnover rates of dominant species; for example, it will be considerably shorter in studies of microcosms than of forests (Connell and Sousa 1983). Temporal scale is also important for classifying the system processes as either fast or slow i.e. as either variables or parameters in a model of the system (Ludwig et al. 1978; Beisner et al. 2003).

In a forest context, it is important to clarify the relationship succession holds with respect to the system states of interest. A forest that changes as it ages is not necessarily changing states; rather it may be considered to be undergoing compositional and structural change internal to the system without switching into an alternative state e.g. when a forest changes into a grassland, tundra or sphagnum bog. However, there are some circumstances when succession may give rise to an alternative and less desired state, for example, when a forest undergoes a gradual paludification to a bog-like state (Banner et al.1983; Fenton et al. 2005), or when an shade-intolerant deciduous stand senesces into a permanently shrub-dominated condition (Shields and Bockheim 1981; Nierenberg and Hibbs 2000). In essence, the role succession plays depends on how research questions or management issues are formulated, the time scales involved and their relation to relevant system states (Carpenter et al. 2001).

Alternative stable states

The term "alternative stable states" can have one of two, not necessarily mutually exclusive, meanings (Beisner et al. 2003). In population or community ecology, a stable state refers to a relatively constant configuration or assemblage of individuals or species (Lewontin 1969; Sutherland 1974; Law and Morton 1993). In ecosystem ecology, a stable state refers to a set of self-perpetuating and mutually reinforcing structures and processes (Holling 1973; Peterson et al. 1998) – the definition used in this paper. Within a 'stable' state, ecosystem attributes such as species composition can fluctuate, but these fluctuations occur within certain relatively constant limits maintained by internal ecosystem structures or external constraints (Connell and Sousa 1983; Scheffer et al. 2001). Moreover, stable states always show slow trends, such as paludification (Scheffer and Carpenter 2003). When resilience is diminished and an ecosystem reorganizes, control of ecosystem behaviour shifts from one set of interacting physical and biological structures and processes to another (Peterson et al. 1998).

A great variety of ecosystems can exist in alternative stable states (Walker and Meyers 2004). Alternative stable states have been documented for lakes, coral reefs, marine fisheries, benthic systems, wetlands, forests, savannas, and rangelands (Carpenter et al. 2001; Scheffer et al. 2001). Examples of alternative stable states in forest ecosystems are presented in Table 1.1.

As with resilience, whether a specific stable state is desirable depends on its social or economic utility and the management context. In unmanaged ecosystems, evolution has led to dominant states that have high resilience as a function of how prevailing natural disturbances influence the structures and processes that provide this resilience. Human interventions beyond the historical range of variability can cause novel pressures on ecosystems and alter the natural dynamics between disturbances and stable states, thereby changing the 'stability landscape' (Scheffer et al. 2001) in the managed ecosystem. Often, this change is manifested as an increase in the number of stable states.

Typically, shifts among states can result in one of two ways: (i) from changes in slow processes that trigger responses in a fast process, and (ii) compounding of disturbances. A classic forest example of changes in slow variables triggering rapid change is the control of spruce budworm (*Choristoneura fumiferana* Clemens) by birds and its relation to the density of budworm's primary prey, balsam fir (*Abies balsamea* (L.) Mill.) in spruce-fir forests of eastern Canada (Holling, 1973; Ludwig et al. 1978). Between outbreaks, the density and size of balsam fir and spruce (*Picea* spp.) increase gradually (slow process), reducing the efficiency with which predatory birds forage (fast process). Once foraging efficiency drops below a certain threshold, avian predation can no longer play an important role in maintaining low budworm abundance and, in combination with other controlling factors, leads to budworm outbreaks (Holling 1988).

In addition to slow changes in environmental variables, shifts between ecosystem states depend on the frequency, intensity and compounding of disturbances. Typically, disturbances at the larger end of their range of intensity and size do not cause a lasting change in the fundamental character of a system (Paine et al. 1998). Large, infrequent disturbances usually do not undermine the mechanisms that determine species composition, as composition before and after such disturbances is typically similar (Turner et al. 1997). Exceptions include very severe events, such as lava flows or intense and soil-destroying fire, which eliminate key ecosystem components. Wholesale transition to fundamentally different states – "regime shifts" – most often occur when multiple disturbances occur within the normal recovery time of the system (Paine et al. 1998). When either the spatial extent, or more commonly, the frequency of a given severe disturbance occurs at or beyond the extreme end of its historical range of variability, the regenerative capacity of a ecosystem may be overwhelmed (Paine et al. 1998). One example of such a regime shift is the abrupt change from closed canopy spruce-moss forest to open canopy woodland that can occur following consecutive disturbances in eastern North America (Payette et al. 2000). When insect outbreaks, fire or harvesting occur in rapid succession, their combined impact can dramatically reduce the spruce seed pool, not only immediately post-disturbance but for a prolonged period, thereby engendering a shift from closed forest to sparse trees with thick lichen mats that inhibit further tree establishment in a positive feedback loop (Payette et al. 2000; Payette and Delwaide 2003). Wind and fire can similarly occur as cascading catastrophic perturbations in northern hemlock-hardwood forests, causing a shift in dominance from shade-tolerant trees to sustained aspen-birch dominance (Lorimer 1977; Frelich and Reich 1999). Likewise, in the near-boreal forests of Minnesota, USA, when two severe fires occur within 10 yr, a shift in species dominance from jack pine to aspen can occur as the result of the second fire consuming surviving jack pine propagules (Heinselman 1973).

Panarchy: an integrative theory of change and ecosystem dynamics

A general theory of change in complex adaptive systems, termed Panarchy, integrates resilience and ecosystem dynamics at multiple scales (Gunderson et al. 1995; Holling 2001; Gunderson and Holling 2002). It combines the notion of an adaptive cycle with the hierarchical structure of ecosystem structures and processes. Briefly, Panarchy considers ecosystems as a set of interacting adaptive cycles that occur at different spatial and temporal scales. It is a way of looking at natural systems, rather than a specific testable hypothesis itself (Carpenter et al. 2001; Gunderson and Holling 2002).

The adaptive cycle

Resilience theory states that complex systems such as forests do not simply tend towards an equilibrium condition, but rather cycle dynamically based on the tension between forces that select for efficiency and those that select for novelty (Gunderson and Holling 2002). This adaptive cycle can be divided into four phases: rapid growth and exploitation (**r**) and conservation (**K**) – the front loop – followed by release or collapse (**Ω**) and, subsequently, renewal or reorganization (**α**) – the back loop (Holling, 1986; Fig. 1.1). The growth and conservation phases usually occur slowly as a system organizes. Collapse and renewal are usually rapid as a system reorganizes. The two complementary and sequential loops foster both conservation and change (Holling 1986; Holling 2001). The front loop maximizes production, efficiency and accumulation of capital, while the back loop maximizes invention, variety and re-assortment. The lags between these periods of stability and bouts of strong selection and innovative rearrangement mean that diversity is perpetually created, thereby providing the elements for resilience over time across a large range of environmental conditions (Holling and Gunderson 2002).

The front-loop of the adaptive cycle is essentially ecological succession. The system slowly builds biomass, connectedness and potential for change. Gradual change in the front loop increases a system's vulnerability to disruption and reorganization. For example, as a stand ages and its dominant trees ascend beyond the canopy, windthrow hazard gradually increases for the entire stand (Everham and Brokaw 1996). During the front loop, resilience is initially high during the exploitation phase – the system is capable of absorbing a wide range of disturbances without shifting into an alternative stable state – but gradually decreases through the K-phase as the system reaches its limits of mature growth (Fig. 1.2). Rigidity and internal control of processes (connectedness) increase as do its stored energy and biomass (potential). At the end of the K-phase, the system has low resilience so that even small perturbations can initiate a cascade of rapid structural changes throughout the highly-connected system. In the K-phase, the system may actually be very stable e.g. a forest will rebound quickly from a small disturbance such as localized windthrow. However, this stability is narrow and local (Gunderson 2000), highlighting the possible trade-off between resilience and stability (Holling 1973).

Rapid change and reorganization characterize the back loop. As the system collapses (Ω) and begins to reorganize (α), attributes or components of the ecosystem can be lost or dramatically changed before a new system begins to organize itself. Moreover, external forces or variability can exert strong influence on the system e.g. seeds from veteran trees or trees outside the disturbed area can exert a lasting influence on the composition of a regenerating stand. As the α-phase unfolds, many experimental and innovative combinations of system components are 'tried' locally and most fail, as observed, for example, in the many distinct assemblages of trees and shrubs that established following the eruption of Mount St. Helens (Franklin and MacMahon 2000). It is during these periods of collapse and reorganization that future development of the system can most intimately depend on small, fast processes i.e. as it reorganizes around 'seeds' of order emerging from lower levels of organization (Peterson 2000). During the back loop, connectedness and potential are low and internal controls of system processes are weak; resilience to another similar disturbance is high because change in one component of the system has few consequences for other components (Fig. 1.2). Following the back loop, if sufficient system components remain, the system may then enter back into the r-phase and begin anew to build complexity, connectedness and resilience. Alternatively, the system may re-organize to a new configuration based on the arrival of novel species or components.

Cross-scale structure and connections

The processes and structures that control ecosystem behaviour can be understood as a cross-scale, nested set of adaptive cycles (Holling and Gunderson 2002). For instance, in the boreal forest, Holling (1986) recognizes the following scales and their associated processes (Fig. 1.3): at fine scales, pattern and process are dictated by biophysical forces that control plant physiology and morphology. At the coarser and slower scale of patch dynamics, competition among plants for nutrients, light and water determines local species composition and regeneration. The next coarser and slower scale is set by meso-scale contagious processes such as fire, insect outbreaks and large mammal herbivory that together determine structure and successional dynamics of stands at scales ranging from tens of meters to kilometres, and from years to decades. At the landscape scale, climate along with geomorphological and biogeographical processes alter ecological structure over hundreds of kilometres and millennia.

The hierarchical structure contributes to resilience because it performs two important functions: (i) to conserve and stabilize conditions for the smaller and faster levels i.e. interactions across scales in a hierarchy are asymmetrical so that larger, slower levels constrain behaviour of smaller, faster levels, and (ii) to generate and 'test' innovations such as mutations or new species assemblages by experiments occurring within each level (Simon 1974; Levin 1981; O'Neill et al. 1986). In other words, finer-scale positive feedback creates heterogeneity while coarser-scale negative feedback stabilizes it (Levin 2000).

Two types of cross-scale connections or feedbacks among adaptive cycles affect resilience: "Revolt" and "Remember" (Holling and Gunderson 2002; Fig. 1.4). These connections are important during times of rapid change in adaptive cycles. "Revolt" refers to what occurs when a level in the hierarchy enters the collapse (Ω) phase, and triggers a crisis in the next larger, slower cycle because the larger, slower cycle is itself in its K-phase, a phase of low resilience. For example, in the western boreal forest, this occurs when fuel loads and climatic conditions allow a locally ignited ground fire to spread to a tree crown, then to a patch of trees and eventually an entire stand. If revolt leads to another crisis in higher cycles undergoing a phase of low resilience, then change to an alternative state can be caused by an abrupt positive feedback chain of collapse (Holling 1996; Carpenter et al. 2001).

"Remember" refers to the cross-scale connection that occurs when collapse occurs at one level and its development is shaped by the accumulated biomass or potential of a slower and larger level in its K-phase (Fig. 1.4). A forest "remembers" its pre-disturbance composition and structure by the presence of at least three interacting parts (Nystrom and Folke 2001; Lundberg and Moberg 2003; Folke et al. 2004): biological legacies, mobile links and support areas. Biological legacies are species, patterns or structures that persist within a disturbed area and act as sources of ecosystem recovery, such as large living and dead trees or tree clusters that provide seeds, buried rhizomes or roots and nutrients to the regenerating stand (Franklin and MacMahon 2000). In some cases, these legacies may be biased towards structures or patches more likely to survive the disturbance, such as wet or low-lying sites during forest fires. Mobile links are 'keystone' organisms that move between habitats and ecosystems after a disturbance to provide lacking essential ecosystem processes such as pollination, seed dispersal, or nutrient translocation by connecting areas that may be widely separated spatially or temporally (Lundberg and Moberg 2003). Support areas refer to landscape patches or habitats that maintain viable populations of mobile links (Lundberg and Moberg 2003). Together, these interacting parts play a pivotal role for renewal and reorganization in a disturbed system.

Natural disturbance-based management

Natural disturbance-based management (NDBM) refers to silvicultural practices and strategic planning approaches that emulate natural disturbance regimes (Hunter 1993; Hansen et al. 1991; Attiwill 1994; Bergeron and Harvey 1997; Angelstam 1998). The underlying rationale is that management approaches based on the dominant natural disturbance regime will restore or maintain biodiversity and essential ecosystem functions by restoring or maintaining the full historical range of habitat heterogeneity observed at multiple scales in unmanaged forests (Franklin and Forman 1987; Pickett et al. 1997). It is essentially a coarse-filter approach to biodiversity conservation that recognizes that (i) humans tend to homogenize or constrain the type, frequency, severity and size of disturbed patches to the meso-scale (Kuuluvainen 2002; Bergeron et al. 2002), and (ii) spatial and temporal variability in disturbances, both human and natural, are essential to maintaining the habitat heterogeneity on which all component species of an ecosystem depend (Hunter et al. 1988; Kuuluvainen 2002). Typically, NDBM approaches seek to maintain biodiversity relative to present or reference (pre-industrial or pre-European settlement) forest ecosystems (Bergeron et al. 2002).

NDBM approaches have been articulated for landscape and stand scales. At the landscape or strategic planning scale, NBDM entails creating or maintaining, as closely as possible to that observed in 'natural' forests, the abundance and distribution of different age classes, the diversity of size classes of disturbed areas and the spatial arrangement of different stand types over time (Angelstam 1998; Bergeron et al. 1999). These objectives may be attained by varying rotation length (Seymour and Hunter 1999; Burton et al. 1999) or diversifying silvicultural practices in different cohorts (Bergeron et al. 2002). At the stand scale, NDBM focuses on silvicultural systems, such as variable retention, that preserve elements of forest structure within the historic range of variability retained by dominant disturbances (e.g. large woody debris, trees of varying sizes and species including large canopy dominants, snags of varying sizes and different stages of decay, and gaps in the canopy) (Franklin 1993; Franklin et al. 1997; Mitchell and Beese 2002; Aubry et al. 2004). Retaining structure can maintain associated ecological functions and processes; preserve genetic information of trees, shrubs and associated biota; maintain structural complexity; improve connectivity between cutting units and forested areas; and serve as 'lifeboat' habitat for organisms that might otherwise be lost temporarily or permanently (Franklin et al. 1997; Haeussler and Kneeshaw 2003). At both scales, NDBM schemes include a myriad of other components that are beyond the scope of this article (e.g. frequency of landscape-scale disturbances, strategies for managing the abundance and distribution of various structural types in time and space, process restoration such as by prescribed fire).

1.5 TRADITIONAL FOREST MANAGEMENT, BIODIVERSITY AND RESILIENCE

Typically, at the onset of a given management regime, managers do well at achieving a narrow set of well-defined objectives by controlling a target variable such an allowable annual cut or a given rotation period (Holling and Meffe 1996; Gunderson 2000). This achievement is typically the result of a "command and control" strategy that seeks to standardize or minimize the natural variability of key ecosystem processes such as fire or regeneration (Pastor et al. 1998). Unfortunately, initial management successes have the simultaneous consequences of (i) encouraging people to grow dependent on the continuation of the management regime and (ii) eroding the ecological resilience of the desired ecosystem state by slowly changing other parts of the ecosystem and reducing ecosystem variability in time and space (Holling 1986). This leads to a condition in which ecological change becomes increasingly undesirable yet more difficult to avoid. When dramatic change does arrive, it is typically a surprise and a policy crisis ensues (Gunderson et al. 1995). Such is the "pathology of resource management" – any system of practices, institutions and regulations that manages an ecosystem for the consistent and predictable flow of goods or services results in reduced resilience, dependent societies and inflexible management agencies (Holling and Meffe 1996). For example, elevated harvesting rates and intensive management for sustained timber yields from the 1950s to 1980s resulted in greatly reduced compositional, structural and age-class diversity across the spruce-fir forests of New Brunswick, Canada. This shift had several unexpected consequences, including a long-term decline in diversity (though not volume) of wood products, elevated harvesting costs, widespread failures of advance regeneration, commercial extinction of white pine, a long period of chronic budworm infestations and sustained shortages of high-quality timber (Baskerville 1985; 1988; Regier and Baskerville 1986).

The resilience theory summarized above suggests that homogenization of landscape structure and composition through reductions in the variability of disturbance regimes can entrain faster major cycles of adaptive change because normalized forests are more connected and less diverse, both across time and space. This suggestion is consistent with general concerns that increased forest homogenization can increase overall landscape susceptibility to fires, insects, and diseases (Franklin and Forman 1987; Turner and Romme 1994; Bergeron et al. 1998) or that disturbances initiated in one part of the landscape are more easily transmitted to other parts in a well-connected landscape (Peterson 2002b). In the long term, reducing the diversity in type, frequency or size of disturbance means fewer selective forces are at work due to the harmonisation of disturbance cycles – this harmonization would lead to less renewal and, eventually, less diversity. Evidence of this process at work comes from documentation of extensive losses of biodiversity in intensively-managed Scandinavian forests from the combined effects of fire suppression and long-term, intensive forest management (Berg et al. 1994; Berg et al. 1995; Hansson 1997; Angelstam 1998).

Given the homogenizing potential of traditional management on forest diversity, ecologists and managers are increasingly emphasizing biodiversity should be maintained to sustain desirable system states as environmental conditions change over time (Peterson et al. 1998; Kuuluvainen 2002; Bengtsson et al. 2003). It is established in the literature that (i) heterogeneous natural systems like forests are more stable than simpler artificial systems like crops and laboratory populations (Murdoch 1975; McCann 2000), (ii) adequate performance of ecosystems depends on having diverse species in all necessary functional groups (e.g. pollinators, primary producers, herbivores, etc.) (Schindler 1990), and (iii) the persistence of ecological function over time depends on the diversity of response by different species within a functional group to changing environmental conditions (i.e. response diversity) (Walker et al. 1999; Naeem et al. 1994; Elmqvist et al. 2003). In other words, resilient ecosystems depend on functional diversity and response diversity at multiple scales (Peterson et al. 1998).

In forests, evidence that diversity and complexity afford increased resilience comes from studies of both natural and managed areas. These studies show that forests with higher structural and compositional diversity more easily resist disturbance, more quickly regain pre-disturbance composition and, in some cases, are more productive than less diverse forests. At the landscape scale, increasing complexity may decrease the spread rate and extent of diseases and disturbances (Turner et al. 1989; Rodriguez and Torres-Sorando 2001), decrease survival and reproductive rates of introduced species (Simberloff 1988; Cantrell and Cosner 1991; Saunders et al. 1991; Andren 1994; Bender et al. 1998; Hiebeler 2000), and globally stabilize locally unstable population dynamics (Hastings 1977; May 1978; Reeve 1990; Taylor 1990). At the stand scale, species composition in sites with high diversity of understory plants has been shown to be more stable and recover more quickly from disturbance than in less species-rich sites (De Grandpre and Bergeron 1997; Turner et al. 1999). Plantations are typically less resilient to disturbances such as fire and often experience more pest outbreaks than natural forests (Coyle et al. 2005; Perry 1998; Schowalter 1988). In terms of productivity, Caspersen and Pacala (2001) analyzed a large set of US forest inventory data and found that rates of stand growth increase monotonically with canopy species richness, irrespective of successional status. In other words, stands with high tree diversity, either of early or late successional status, had higher annual growth than respective low diversity early or late successional stands (Caspersen and Pacala 2001).

In practice, whether enhanced connectedness as a result of homogenizing management activity is beneficial or detrimental depends on the exchange process, patch types, stresses or disturbances, desired states, and management objectives for the forest in question. For example, if maintaining viable grizzly bear populations in managed areas is an objective, retaining connectedness among habitat types required by grizzlies for foraging and reproduction is critical (Weaver et al. 1996). Conversely, if management objectives include the maintenance of viable populations of endemic herbs, the enhanced connectedness provided by forest roads that augment the dispersal of introduced plant species is clearly detrimental (Trombulak and

Frissell 2000).

1.6 CAN FOREST MANAGEMENT BASED ON NATURAL DISTURBANCES MAINTAIN ECOLOGICAL RESILIENCE?

In a forestry context, resilience-based management refers to a set of planning and silvicultural practices aimed at building the ecosystem's capacity to persist in a desired state and to reorganize in the face of severe disturbances (Walker et al. 2002). In cases where the ecosystem is already in an undesirable state or configuration, resilience-based management means reducing the resilience of the present state and enhancing that of more desirable states. Any such practices or planning approaches must consider the present system state in relation to its adaptive cycle, its likely trajectories given ongoing stresses or other likely influences on system change (i.e. climate change, introduced species, development pressures from urbanization, etc.) and how the present state compares to desired system states.

The principal foci for a NDBM approach that can maintain ecological resilience are, clearly, disturbances and their retained biological legacies. Both NDBM and a resilience perspective recognize the importance of maintaining structural and compositional heterogeneity at multiple scales as well as the role that disturbances play in generating heterogeneity. Integrating the two perspectives requires understanding not only the dynamics of disturbance to allow emulation of disturbance at relevant scales, but also understanding the impacts of such an approach on the processes and structures that perpetuate desired system states. Moreover, it requires understanding how management can affect slow system processes, such as long-term seed bank or nutrient dynamics, that may in the long term result in ecological surprises by altering the capacity of a forest to reorganize and re-establish following a disturbance. For instance, tree removal in northern forests can alter the competitive balance on the soil towards cryptogram mats that inhibit the germination and establishment of tree regeneration (Dussart and Payette 2002).

At a strategic or landscape level, implementing NDBM while maintaining resilience means forest managers must make several decisions regarding the role of disturbances within their jurisdiction, in particular whether harvesting will substitute or be additive to the frequency of intense natural disturbances. If the former, the role of large-scale disturbance suppression and its impacts on the slow variables that can, in the long term, diminish resilience of certain states needs to be addressed. If the latter, a decision then needs to made about the implications of compounded impacts on the resilience of any desired states put at risk by the additive disturbances, especially if deemed that present harvesting rates are outside the historic range of variability or if harvesting has become the dominant disturbance across the landscape. A further landscape-level consideration is that the potential for homogenization makes it important to not implement the same silvicultural system everywhere. Given the infeasibility of truly replicating natural disturbance by harvesting, preventing homogenization also depends on a network of reserves and protected zones to maintain heterogeneity throughout the managed area as well as support areas for important mobile link species (Bengtsson et al. 2003).

At the stand scale, the retention of ecological structures during harvest as prescribed by a NDBM approach fits well with managing for resilience. Retaining structure allows for the harvested stand to 'remember' its pre-harvest condition: genetically, compositionally, structurally and otherwise. In this respect, a NDBM approach that maintains resilience involves paying close attention to management impacts on 'neighbourhood effects' – any feedback processes mediated by canopy trees that alter the probability of replacement by the same or other species following the death of canopy trees (Frelich and Reich 1995; 1999). Neighbourhood effects may act as positive feedback (self-replacement) or negative feedback (replacement by another species). Examples of positive neighbourhood effects include stump sprouting, seed rain, or the deep shade cast by shade-tolerant species (Wilson and Agnew 1992; Frelich and Reich 1999). Examples of negative neighbourhood effects include alterations of soil physical and chemical properties or changes in mycorrhizal characteristics that permit the favour the growth of alternative species than what are presently on site (Frelich and Reich 1999; Frelich 2002). Tailoring prescriptions of stand-level retention to the species and structures that have positive neighbourhood effects can help maintain the resilience of desired states. In doing this, it is important to keep in mind that although management may seek to emulate disturbance by retaining structure, fundamental differences exist between harvesting and fire, windthrow, insect outbreaks or pathogens (McRae et al. 2001; Quine et al. 1999). These differences mean that maintaining diversity through harvesting may be insufficient to maintain resilience unless the process and not only the pattern of disturbance is emulated (Haeussler and Kneeshaw 2003). For example, in the boreal forest of Quebec, a form of clearcutting that protects regeneration and soils may produce stands with the same even-aged structure as severely burned stands, but these stands may be less productive that post-fire stands because of enhanced biomass partitioning into the forest floor rather than living trees (Fenton et al. 2005; Lecomte et al. 2005).

An important consideration is the compounding of severe disturbances resulting from salvage operations. Salvage logging of burned, insect-attacked or blowdown areas can reduce biological legacies during a critical phase of system recovery, diminish capacity for re-establishment of pre-disturbance communities and alter biogeochemical and hydrological processes (Beschta et al. 2004; Lindenmayer et al. 2004). For example, large-scale salvage operations following the 1938 hurricane in central New England, USA entrained a regional-scale shift in hydrology and, compared to non-salvaged areas, delayed or prevented the recovery of pre-disturbance composition and ecosystem processes (Foster et al. 1997). Moreover, removal of burned logs and other woody structure can have long-term consequences on forest diversity and productivity. In the eastern Canadian boreal, for example, thin, post-fire soils are particularly sensitive to compaction and alteration by harvesting equipment (Brais et al. 2000) and salvage logging simplifies the biodiversity in ground vegetation (Purdon et al. 2004). Other important salvage impacts include the recruitment dynamics of trees that require nurse logs for regeneration and habitat for cavity-nesting birds and other keystone taxa (Harmon et al. 1986; Imbeau et al. 2001; Nappi et al. 2004; Fisher and Wilkinson 2005). In many cases, it may not be economically or socially desirable to forgo entirely timber supply losses due to fire or other disturbances, making tradeoffs necessary in terms of biological legacies and salvaged timber. It is these cases, resilience theory can be especially helpful in identifying key structural legacies such as seed trees or CWD that are paramount to the maintenance of ecosystem functions and processes.

To return to our title question, can NDBM maintain resilience? The answer is probably yes, if implemented silvicultural strategies based on natural disturbances retain the patterns, structures and processes that perpetuate desired states while reducing those that enhance the resilience of undesirable states. Essentially, forest managers should view harvesting and regeneration treatments as the back loop of the managed forest and seek to emulate the 'creative destruction' character of natural disturbances. In this context, we propose several recommendations for managers who want to implement NDBM while seeking to maintain resilience of desired ecosystem states (Table 1.2). The recommendations assume that: (1) resilience under changing environmental conditions depends on maintaining both adequate membership in all functional groups as well as diversity and redundancy within and across scales for all functional groups (Peterson et al. 1998; Walker et al. 1999; Elmqvist et al. 2003), (2) patterns of retention at stand and landscape scales should maintain elements of ecological 'memory' of desired system states such as support areas, and (3) retained biological legacies at stand and landscape scales should maintain specific ecological functions or habitats of mobile link taxa. Examples of structural features that allow stands to reorganize post-disturbance include seed trees, advance regeneration, CWD, snags, and inocula sources for mycorrhizae.

To illustrate the above recommendations, we provide an example of how they may be applied in the context of western North America's pine forests (Table 1.3). Presently, the largest recorded outbreak of a native insect, the mountain pine beetle (*Dendroctonus ponderosae* Hopkins), is currently underway. These insects caused significant mortality of pines across over four million hectares in British Columbia, Canada, in 2003 alone (Taylor and Carroll 2004). These forests are connected at multiple scales – the last seven winters of above-normal temperatures have greatly increased the amount of climatically-suitable habitat for mountain pine beetles while freeing these insects from the over-wintering mortality that typically maintain them at low abundances (Carroll et al. 2004). At the same time, central interior British Columbia has experienced a tripling in recent decades of susceptible mature pine forest as a result of reduced fire frequency, providing an unprecedented connectedness of the beetles' preferred habitat type – mature pine forest (Logan and Powell 2001; Taylor and Carroll 2004). Unless cold winters return, this outbreak could continue to spread until the supply of susceptible pine trees is exhausted. Such a scenario entails significant and sustained long-term decreases in timber supply (BC Ministry of Forests 2003) and widespread ecosystem changes. The combination of climatic conditions and landscape age-class structure suggests this forest landscape is 'resetting' itself and that a collapse (α) phase is unfolding. Understanding this historical development and its possible future consequences (e.g. the potential for catastrophic fires resulting from a large 'pulse' of dead forests across the landscape combined with forecasted increases in the severity of fire weather (Flannigan et al. 2001)) provides essential context for designing a strategic management approach that may sustain resilience during times of rapid change (Table 1.3). Moreover, since different parts of the landscape are at different stages of outbreak, different practices will be simultaneously useful for enhancing resilience.

1.7 CONCLUSION

Understanding resilience may hold the key to overcoming the challenges inherent to applying NDBM for the sustainable use of forest resources. Moreover, this understanding can provide resolution to the fundamental mismatch between the dynamics of unmanaged systems and human-dominated systems (Levin 2000) e.g. the mismatch between disturbance cycles that create a diversity of forest types *vis-à-*

vis economic forest rotations that reduce much of this diversity. The mismatch makes it difficult (i) to see the signs of possible irreversible change until that change is underway (Scheffer et al. 2003), (ii) to understand the effects of individual management decisions on coarse scales, and (iii) for individuals to feel their actions can influence sustainability since it is the actions of billions that dictate the dynamics of the global commons (Levin 2000). Resolution of this mismatch in the context of forest management may come from developing policies and practices of resource use that recognize:

- Traditional forms of harvesting are fundamentally different disturbances than fire, windthrow or other disturbances in our forests. The relevant question, from a scientific perspective, is not whether traditional logging or other severe silvicultural interventions fit within the range of historic natural variability, but rather how far management can deviate from this range before compromising ecosystem integrity and resilience (Perry 1998). That said, NDBM should not only emulate structure and pattern but also process, as ecological processes and functions are a fundamental component of resilience.
- Ecosystems are prone to catastrophic change when managed by command and control strategies that minimize variation of key environmental variables such as disturbance regime or when subjected to long-term anthropogenic pressures that simplify the forest landscape (Holling and Meffe 1996; Carpenter et al. 2001). This potential means management approaches and policies based on maximum sustained yield are appropriate only in the rare situations of high certainty and controllability of the managed system (Peterson et al. 2003).
- The globalization and intensification of anthropogenic stresses such as introduced species and climate change are expected to increase the frequency and intensity of compounded perturbations and thereby the frequency of both adaptive and catastrophic shifts within ecosystems, including forests (Paine et al. 1998; Regier and Kay 2002). This increase implies ‘Revolt’ may become

more common and puts an onus on managers to build or maintain resilience. In the case of forests, implementing these elements in a management regime that maintains resilience may be easier than for other resources such as fisheries or grazing grounds because forest inventories are easily tractable, ample knowledge exists about ecosystem function and dynamics, and there is an increasing social willingness to sustain their ecological goods and services for future generations (Ostrom et al. 1999; Scheffer et al. 2000). At the very least, understanding resilience can better contextualize NBDM within its broader goal of sustainability, focus management attention on process rather than principally on pattern, and help managers escape their preoccupation with the likely unattainable goal of unfaltering stability (Carpenter et al. 2001; Johnson et al. 2003).

1.8 ACKNOWLEDGMENTS

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1.9 TABLES

Table 1.1 Alternative stables states in forest ecosystems.

Location	Alternative states	Reference(s)
Boreal forest of New Brunswick, Canada	outbreak vs. endemic abundance of the spruce budworm	Jones, 1975; Ludwig et al. 1978
Deciduous old-growth stands in Minnesota, USA	high vs. low abundance of a forest forb in relation to high abundance of an ungulate herbivore	Augustine et al. 1998
Serengeti-Mara woodlands, Tanzania	wooded vs. grassland states resulting from fire and elephant disturbance	Dublin et al. 1990
Northern boreal forest, Sweden	birch-spruce succession vs. pine dominance as a function of ungulate browsing	Danell et al. 2003
Southern boreal forest of Québec, Canada	dense spruce-moss stands vs. lichen woodlands as a function of compounded disturbance	Payette et al. 2000; Dussart and Payette 2002
Pinelands in New Jersey, USA	open, sparsely vegetated areas dominated by lichen mats vs. moss-dominated areas that foster pitch pine (*Pinus rigida* P. Mill.) forests	Sedia and Ehrenfeld 2003
Sandhill forests Florida, USA	hardwood vs. longleaf pine-(*Pinus palustris* P. Mill) as a function of fire frequency	Peterson 2002a
Tropical rainforests	Mesic, fire-inhibiting vs. xeric fire-susceptible forests mediated by opening of roads and agriculture	Cochrane 2003

Table 1.2 Management recommendations at ecosystem, landscape and stand scales to maintain ecological resilience of desired forest states while implementing natural disturbance-based management

Scale	Recommendations	Example	Reference(s)
Ecosystem	• **know your ecosystem –** determine its possible alternative system states, understand how disturbance affects transitions among these states, and investigate how slow processes such as seed pool or nutrient dynamics affect long-term persistence of desired states.	• closed canopy spruce-moss forest or open canopy lichen woodland in eastern boreal of North America	• Payette et al. 2000
	• contextualize management area within relevant hierarchy of scales and understand how disturbances maintain habitat heterogeneity across time and space	• time and space scales of the boreal forest	• Holling 1986
	understand how type,	• deleterious	• Payette and

	frequency and intensity of silvicultural interventions affect the persistence of structures and processes that sustain desired states → **ensure compounded management actions do not diminish or fundamentally alter these structures and processes**	effects of compounded logging and fire on spruce regeneration	Delwaide 2003
Landscape	• understand how dominant disturbances create or maintain heterogeneity in landscape age-class structure and composition; seek to emulate these *processes* with harvesting, rather than an arbitrary characteristic such as disturbance size • characterize, map and model how different disturbance regimes affect age-class composition and structure across the landscape	• role of fire in eastern Canadian boreal and its impacts on the abundance and distribution of old-growth forest or bird species assemblages • delineation of different management zones according to historical fire	• Bergeron et al. 2001; Drapeau et al. 2000 • Cissel et al. 1998; Cissel et al. 1999

		frequency in an Oregon, USA	
	• diversify forest practices – do not implement the same silvicultural system everywhere	• 'Triad' zoning approach for intensive, extensive, and biodiversity-emphasis management	• Seymour and Hunter 1999; D'Eon et al. 2004
	• maintain landscape heterogeneity by allocation of permanent and temporary reserve networks; these networks should (i) provide support areas and allow connectivity for mobile links important for desired states and (ii) act as reference areas to understand slow ecosystem changes and characterize range of variability of key processes such as disturbance.	• riparian reserves to maintain mobile link role of salmon-bear interactions for the transfer of marine nutrients into temperate rainforests	• Hilderbrand et al. 1999; Wilkinson et al. 2005

	• understand and minimize the impacts of salvage harvesting on biodiversity	• reduction of woodpecker habitat and subsequent impacts on secondary cavity nesters	• Nappi et al. 2004
Stand	• understand how harvesting intensity and frequency affects diversity and abundance of component species and structural heterogeneity	• impacts on plant and structural diversity of harvesting, in combination with other disturbances, in eastern boreal forest	• De Grandpre et al. 1993; De Grandpre and Bergeron 1997; Reich et al. 2001; Haeussler et al. 2002
	• retain biological legacies in harvested areas that enhance 'neighbourhood effects' for the desired states	• retain patches of large veteran pine trees as seed sources in fire-dependent ecosystems	• Palik and Pregitzer 1994; Weyenberg et al. 2004
	• tailor the amount, type and pattern of retention to	• retain white birch in	• Simard et al. 1997

	maintain representation of species in all functional groups such as pollination or nutrient cycling	Douglas-fir stands to maintain ectomycorhizzal linkages	
	• maintain habitat of mobile link species	• retain snags as habitat to maintain mobile link role of woodpeckers and secondary cavity nesters for endemic pest control	• Parry et al. 1997; Fayt et al. 2005

Table 1.3 Examples of management actions at each phase of the adaptive cycle aimed at maintaining ecological resilience of pine-dominated forests to a catastrophic outbreak of the mountain pine beetle.

Scale	Phase of adaptive cycle: exploitation (r): 'enhance'	conservation (K): 'resist'	collapse (Ω): 'minimize damage'	renewal (α): 'direct'
Tree-level responses	• invest in below ground resources	• develop thick bark • enhance cone production	• 'pitch out'	• seed dispersal to mineral soil
Stand-level treatments	• mixed species planting • promote or maintain stand complexity	• variable density thinning • snag creation or protection	• 'green-tree' group selection harvesting • prescribed fire following harvesting • maintain stand complexity during salvage	• retain non-host and unaffected trees during salvage • retain habitat for beetle predators and pathogens

Landscape-level tactics	• maintain heterogeneous age-class structure • retain resistant stands	• reduce beetle dispersal (e.g. pheromone baiting, habitat mngt.)	• ‘leading edge’ sanitation harvesting	• retain entire killed stands • maintain dead fuel continuity

1.10 FIGURES

Figure 1.1 The adaptive cycle of a complex adaptive system. The arrows indicate the speed of change in the cycle; short, narrowly spaced arrows indicate slow change whereas long arrows indicate rapid change. The exit from the adaptive cycle on the lower left suggests where in the cycle the potential of a system can leak away and where a switch into a system with lower productivity and organization is most likely to occur.

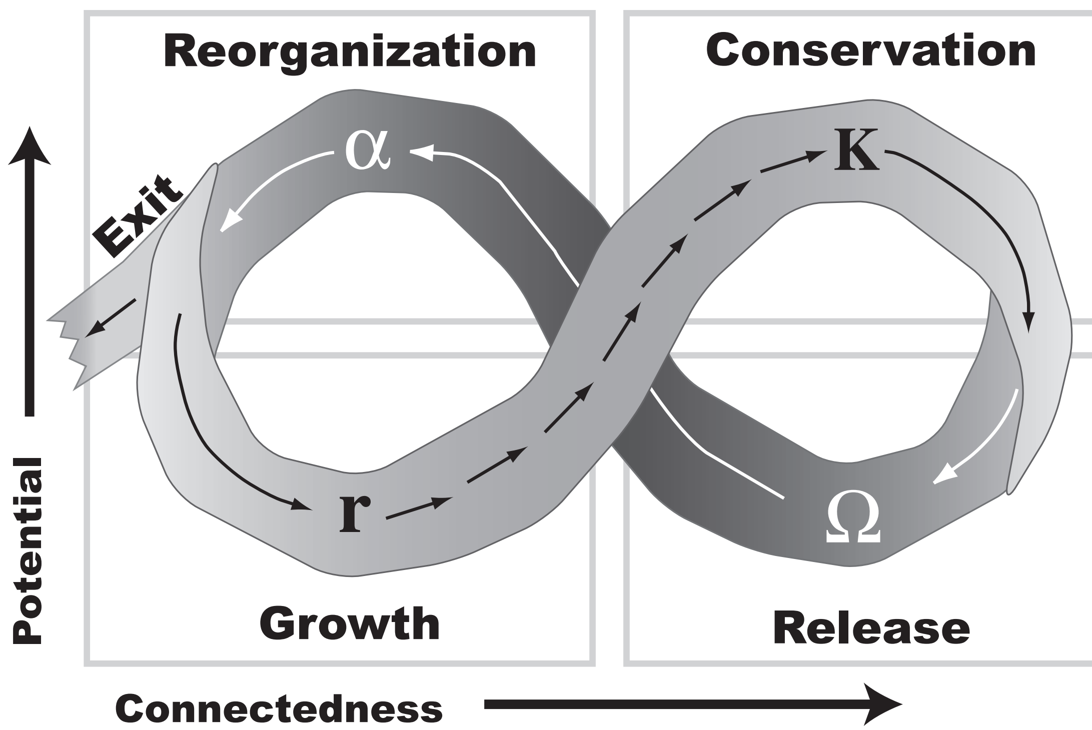


Figure 1.2 Resilience as a third dimension in the adaptive cycle. Resilience changes throughout the cycle, gradually decreasing as the cycle moves from r- to K-phase and increasing as the system rapidly shifts in the back loop.

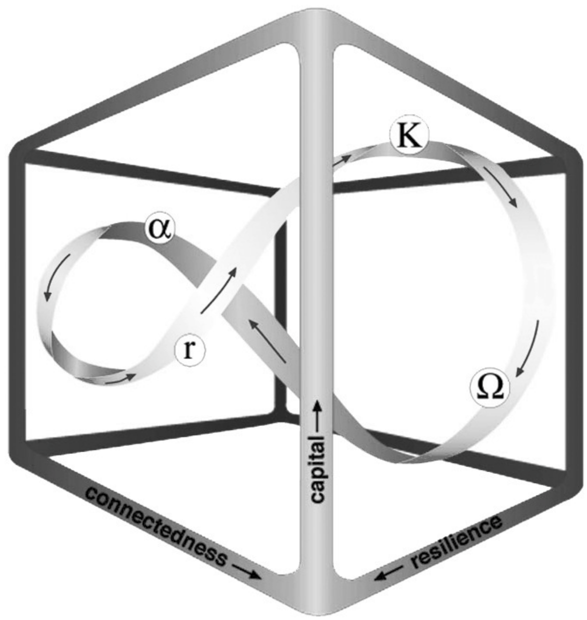


Figure 1.3 Time and space scales in the boreal forest (Holling, 1986) and the atmosphere (Clark, 1985) illustrating relationships among processes that structure the forest. The interactions among the slow vegetation processes and faster atmospheric processes are mediated by meso-scale contagious processes like fire or outbreaks of the spruce budworm.

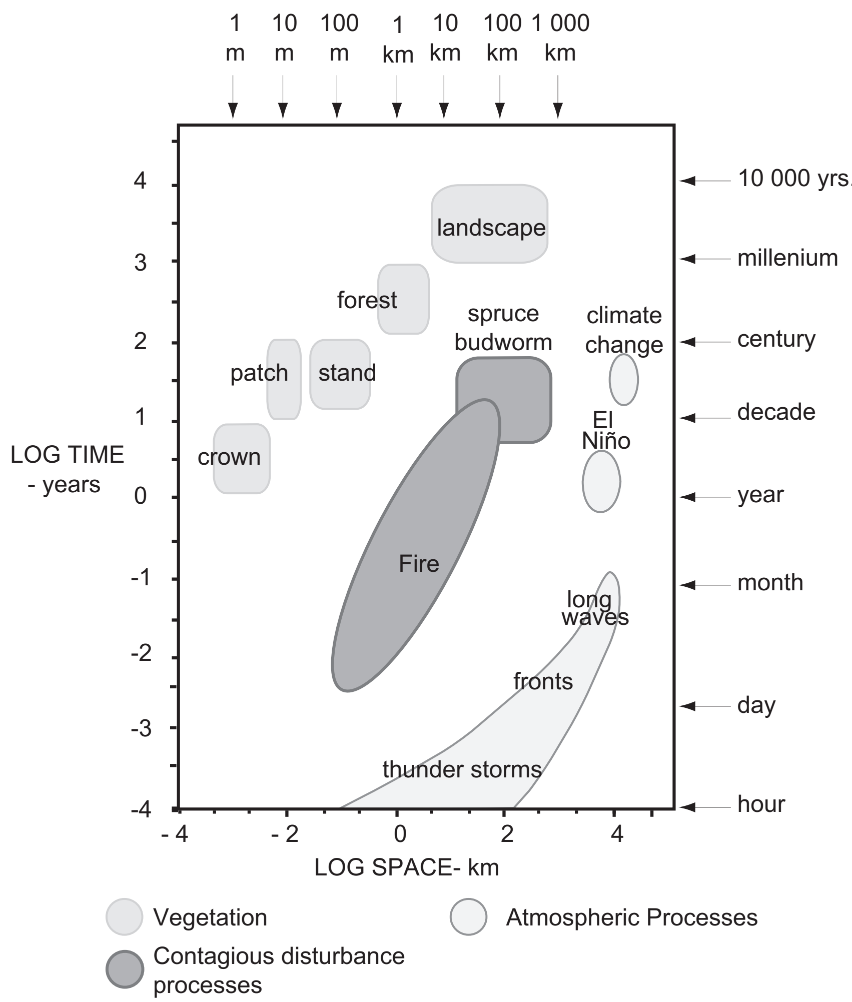
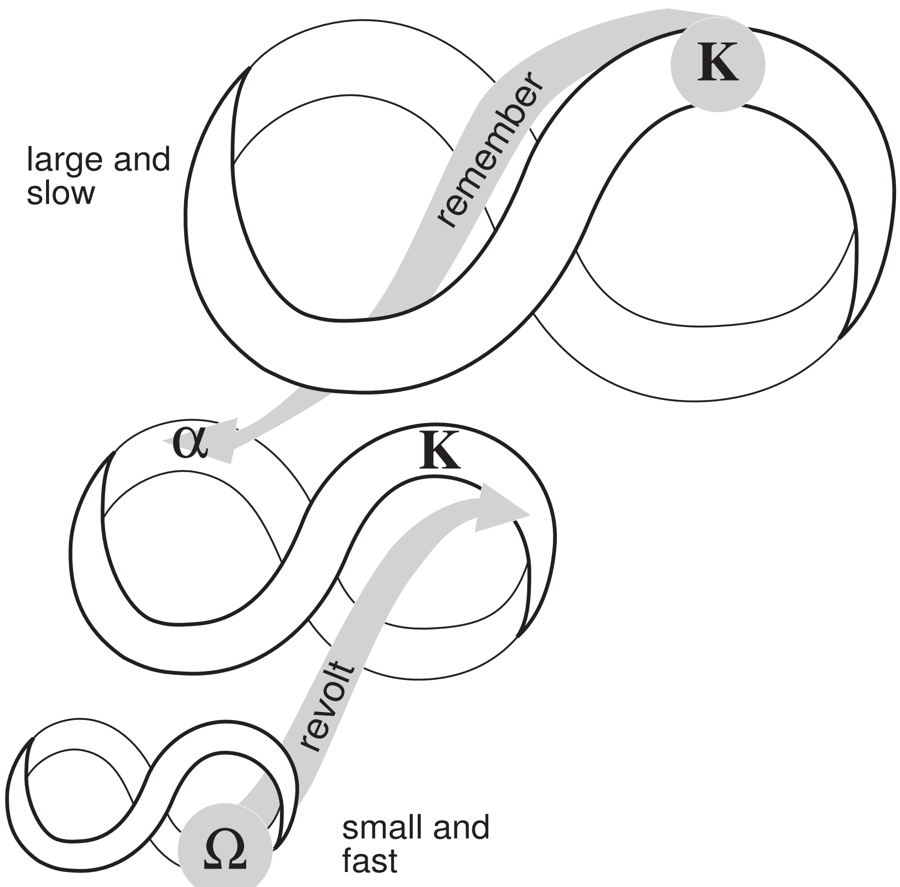


Figure 1.4 Connections among adaptive cycles. “Revolt” refers to how a critical change in one adaptive cycle can cascade upwards to a vulnerable stage in a larger, slower cycle, such as when forest conditions allow a locally ignited ground fire to spread to a tree crown, then to a patch of trees and eventually to the entire stand. “Remember” refers to how renewal is facilitated at a given scale by drawing on accumulated potential in a larger, slower cycle; for example, by the presence of veteran trees providing seeds and nutrients to a regenerating stand.



CHAPITRE 2

FIRE AND CANOPY SPECIES COMPOSITION IN THE GREAT LAKES-ST. LAWRENCE FOREST OF TÉMISCAMINGUE, QUÉBEC

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2.1 ABSTRACT

Large severe fires are typically very rare events in the northern hardwood forests of eastern North America, with estimated return intervals as high as 1400 to 4500 yr. We investigated the history of large severe fires in western Québec, Canada, where sugar maple (*Acer saccharum* Marsh.), yellow birch (*Betula alleghaniensis* Britt.), and eastern hemlock ((*Tsuga canadensis* (L.) Carr.) dominate a landscape at northern limit of the Great Lakes-St. Lawrence forest region. Using a combination of provincial archives of fire data, interpretation of historic air photos and dendrochronological sampling, we estimated fire frequency in a 179,300-ha landscape and tested the hypothesis that time-since-fire is a principal determinant of canopy species composition. Our results indicated that despite its proximity to the boreal mixedwood forest, fires have been relatively infrequent in this landscape, with over 60% of the landscape remaining unburned during the 413 yr period of study. The overall fire cycle, an estimate of the time required to burn an area of equivalent size to the study area, was 494 yr (95% CI: 373-694). This estimate showed strong temporal partitioning, with an elevated frequency that coincided with the period of pioneering settlement beginning in 1880, and spatial partitioning according to surficial deposits, pine dominance and ecological site type. Multivariate analyses elucidated both (i) distinct assemblages of canopy species that differed in their association with time-since-fire and (ii) the relative importance of environmental variables affecting canopy species composition, where time-since-fire explained the largest amount of variation. This time-since-fire effect suggested an important role for secondary disturbances in influencing composition of the canopy. Given the current long fire cycles, it is likely this landscape will experience an increasing dominance of fire-avoiding species such as sugar maple and eastern hemlock, with consequences for fire-adapted tree species and fire-dependent communities.

2.2 RESUME

Les grands feux sont des événements en général très rares dans les forêts feuillues de

l’est de l'Amérique du Nord, avec des intervalles de retour estimés aussi hauts que 1400 à 4500 ans. Nous avons étudié l'histoire des grands feux dans le Québec méridional, où l'érable à sucre (*Acer saccharum*), le bouleau jaune (*Betula alleghaniensis*), et la pruche de l’est (*Tsuga canadensis*) dominent un paysage à la limite nordique de la région des Grands-Lacs/Fleuve St-Laurent du Canada. En utilisant une combinaison des archives provinciales des données de feu, l'interprétation des photos aériennes anciennes et l’échantillonnage dendrochronologique, nous avons estimé la fréquence du feu dans un paysage de 179 300 ha et avons évalué l'hypothèse selon laquelle le temps écoulé depuis le dernier feu est une cause déterminante de la composition d'espèces de la canopée. Nos résultats ont indiqué qu'en dépit de sa proximité à la forêt boréale mixte, les feux ont été relativement peu fréquents dans ce paysage, avec plus de 60 % du paysage restant non brûlé pendant la période d'étude de 413 ans. Le cycle global du feu, une évaluation du temps requis pour brûler une aire de taille équivalente au secteur d'étude, était de 494 ans (IC de 95 % : 373-694). Cette évaluation a montré une division temporelle forte, avec une fréquence élevée qui a coïncidé avec la période de colonisation commençant en 1880, et avec la répartition spatiale des dépôts de surface, de la dominance du genre *Pinus* et de l'emplacement des types écologiques. Les analyses multivariées ont élucidé (i) les assemblages distincts des espèces de la canopée qui ont différé dans leur association avec le temps écoulé depuis le dernier feu et (ii) l'importance relative des variables environnementales affectant la composition, où le temps écoulé depuis le dernier feu a expliqué la plus grande quantité de variation. Cet effet du temps écoulé depuis le dernier feu a suggéré un rôle important pour les perturbations secondaires en influençant la composition de la canopée. Etant donné le long cycle de feu courant, il est probable que ce paysage éprouvera une dominance croissante des espèces évitant le feu avec des conséquences pour les espèces et communautés adaptées au feu.

2.3 INTRODUCTION

Large severe fires have important consequences for the spatial patterning of different stand types, composition and age-class structure of temperate forests (Turner & Romme, 1994). However, large severe fires, i.e. fires that destroy most or all of the overstory and understory, are typically rare events in the northern hardwood forests of eastern North America. For example, in the Great Lakes forests of Michigan, estimates of the return interval for large severe fires range between 1400 and 4500 yr (Whitney, 1986; Frelich & Lorimer, 1991). Several factors provide forests dominated by deciduous trees such as sugar maple (*Acer saccharum* Marsh.) and yellow birch (*Betula alleghaniensis* Britt.) with structural attributes and fuel loadings of low flammability as compared to conifer-dominated forests; these include their high foliar moisture content, low amounts of flammable resins and oils, fuel discontinuity between tree crowns and the forest floor, high decomposition rates of coarse woody debris, and relatively fire-retardant fine leaf litter (Mutch 1970; Philpot 1970; Bessie & Johnson, 1995; Hély *et al.*, 2001; Frelich, 2002). However, at the northerly limit of their distribution in western Québec, Canada, northern hardwood forests are adjacent to the boreal mixedwoods, where fire is a common and extensive influence on forest composition and structure (Grenier *et al.*, 2005; Bergeron *et al.*, 2001). In other 'near boreal' forests, such as the red and white pine forests of the Boundary Waters Canoe Area in Minnesota or pine-aspen forests of Algonquin Park in Ontario, large severe fires are relatively more frequent – between 175 to 300 yr – than in the Michigan landscapes (Heinselman, 1973; Cwynar, 1977).

We conducted a landscape-scale assessment of the history of large severe fires for a forested landscape in Témiscamingue, Québec. We calculated the fire cycle, an estimate of the time required to burn an area of equivalent size to the study area (Johnson & van Wagner, 1985), examined its spatial and temporal variability, and tested hypotheses regarding the role of time elapsed since large severe fire on species composition in the canopy. We tested the hypotheses that (i) European settlement increased the frequency of large severe fire and (ii) spatial partitioning of fire frequency exists in this landscape, where fire frequency increases in coarse surficial deposits and pine-dominated stands and decreases in deciduous-dominated ecological site types.

In addition, we addressed some fundamental questions regarding the role of fire in determining canopy species composition. We asked: are there distinct stand types comprised of contrasting assemblages of canopy tree species related to time-since-fire (TSF)? If so, what are these assemblages? We also examined the extent to which variation in the presence or absence of canopy species can be explained by TSF relative to environmental and site history variables such as aspect, slope or frequency of partial harvesting. We tested the hypothesis that TSF is the principal determinant explaining variation in canopy species composition in this landscape.

2.4 METHODS

Study area

The study area (~ 46º40'N; 78º45'W) covers 179,300 ha (Fig. 2.1). The Ottawa River is its western limit; the remaining boundaries correspond to the limits of 'ecological districts' i.e. relatively homogeneous units based on topography, elevation, surficial deposits, parent materials, and hydrology (Robitaille & Saucier, 1998). The climate is subpolar continental with cold winters and warm summers. Based on the nearest meteorological station, Barrage Témiscamingue (46°42'N, 79°6'W, 181 m above sea level), the mean annual temperature is 4.4^{o}C (Environment Canada, 2005). Mean annual precipitation is 963 mm, with 226 mm (23%) falling as snow. Average snow depth is 13 cm. Growing degree days above 5^{o}C range between 1350 and 1550 while the growing season is typically 180 days long (Robitaille & Saucier, 1998; Richard, 2003).

The topography is characterized by gentle to moderately steep hills, with rounded and sometimes rocky summits. The study area is on the Grenville geological province of the Canadian Shield and its surficial deposits are principally undifferentiated or rocky glacial tills. Parent materials are metamorphic or sedimentary, almost exclusively gneissic or paragneissic and acidic (Brown, 1981). Soils are principally humo-ferric podzols (Brown, 1981). Mean elevation is 311 m above sea level. Mean slope is 9%. Forests cover almost the entire land surface while lakes, rivers and other water bodies cover about 20% of the total area.

The landscape is within the Sugar Maple–Yellow Birch bioclimatic domain for western Québec (Robitaille & Saucier, 1998) and, more generally, the Great Lakes–St. Lawrence forest region (Rowe, 1972). Sugar maple and yellow birch, often with a subdominant component of white spruce (*Picea glauca* (Moench) Voss) and balsam fir (*Abies balsamea* (L.) Mill.), typically dominate the canopy. Such stands comprise 70,141 ha (40% of study area). Eighty-three per cent of the landscape is considered mesic, with the potential tree vegetation on midslope mesic sites being sugar maple and yellow birch (Robitaille & Saucier, 1998).

Other common tree species include eastern hemlock (*Tsuga canadensis* (L.) Carr.), white birch (*Betula papyrifera* Marsh.), trembling aspen (*Populus tremuloides* Michx.), large tooth aspen (*Populus grandidentata* Michx.), red pine (*Pinus resinosa* Ait.) and white pine (*Pinus strobus* L.). Stands of red and white pine occur throughout the study area while stands dominated principally by red pine are abundant on dry sites, shores and rocky outcrops, especially adjacent to the Ottawa River. Pines are the dominant or codominant canopy species across 17,372 ha (10% of the study area).

Forest management by Europeans in Témiscamingue began about 1800 with the targeted extraction of large-diameter white pine, red pine and white spruce (Vincent, 1995). However, it was not until after 1874 when forestry concessions were assigned and saw mills built that industrial logging began on a large scale (Vincent, 1995). Settlement followed in the 1880s, greatly enhanced by the completion of a railway in 1890 and the 1916 completion of a pulp and paper mill in the town of Témiskaming. The role of aboriginal burning for berry or ungulate management is unknown.

Fire history reconstruction

The interpretation of historic air photos, dendrochronological analyses and archival records from the provincial forest agency were used to determine the cumulative TSF distribution for the study area. A preliminary TSF map was created in ARCVIEW 3.2a (ESRI, 1996) by combining georeferenced information of large fires from provincial fire archives (1914-2002) with an assessment of 1930 air photos (B&W; 1:30,000) (Fig. 2.1). The boundaries of recent past fires (i.e. since ~1900) were identified and digitized from the air photos based on tonal, shape, pattern and textural attributes characteristic of post-fire patches. Air photos from 1950 were also examined, but no large fires were detected.

We implemented a stratified random sampling design to estimate the TSF distribution (n = 129) (Johnson & Gutsell, 1994). We first overlaid a rectangular grid of approximately 1800-ha cells over the study area and located a potential sampling site within each cell using the random point utility in ARCVIEW 3.2a. During the summers of 2003 and 2004, we sampled 106 transects, each 20-m wide by 250-m long, and assessed percent slope, aspect, elevation, evidence of past fires (i.e. charcoal or burnt seeds in the mineral soil, burned stumps, charred logs or snags, etc.), tree species composition (presence or absence) in the canopy stratum, and evidence of harvesting (i.e. presence and condition of cut stumps, machine ruts, skid roads, etc.). Evidence of windthrow (toppled, snapped or bent trees, abundant pit-mounds, aligned woody debris) was also noted when present. Our study focused on productive sites, so extreme conditions such as hydric conifer forest or rocky, xeric pine ridges were not sampled.

For dendrochronological sampling (n = 106), we obtained disks or increment cores at approximately 30-50 cm above the ground from five to 15 large living trees at each site, preferably of post-fire pioneer species. From first to last, selection priority was: white birch > aspen spp. > red pine > white pine > yellow birch. For sites where a cohort of these species was not present and there was only weak evidence of recent past fires, we sampled for the oldest living tree to ascertain an estimate of minimum TSF.

The increment cores and disks were mounted, dried and sanded with progressively finer sandpaper. Sample age was then determined by counting annual rings using a dissection stereomicroscope (on two radii for the disks). For cores where the pith was not traversed, we used pith locators to estimate the missing years based on ring curvature and growth rate (Applequist, 1958). Visual cross-dating using diagnostic rings (i.e., narrow, light, and frost rings) was performed across cores and disks of the same species to confirm age determinations (Yamaguchi, 1991).

For dendrochronological sites, a TSF estimate was determined using a sliding temporal window to detect a cohort containing at least 60% of the sampled trees that established within the same 30-yr period. In these cases, the TSF estimate corresponded to the age of the oldest tree in the cohort. When such a cohort was not present, a minimum TSF was assigned to the site according to the establishment date of the oldest sampled tree (Bergeron & Dubuc, 1989; Johnson & Gutsell, 1994). Minimum TSF estimates consider that the last fire burned some time before the establishment date of the present stand; they are deemed censored in further survival analyses (see Statistical analyses for details) (Allison, 1995).

Several sites (n = 14) that fell within the fire boundaries of the initial TSF map were assigned the same TSF estimated from other examined sites within the same fire. Nine other sites located outside the fires mapped from 1930 air photos that proved inaccessible in the field were assigned a censored, minimum TSF corresponding to 1900; this was done to complete aerial coverage of the study area. An estimate of fire date was possible for 49 (38%) sites, whereas the remaining 80 (62%) sites provided censored, minimum TSF estimates. Due to the lack of precision in estimating fire date, data in the stand-age distribution are presented in bi-decadal age classes. This method does not have the resolution necessary to detect small or low-severity fires that did not initiate succession.

Statistical analyses

Fire cycle

To estimate the fire cycle, we used established fire frequency methods in combination with survival statistics (Johnson & Gutsell, 1994; Bergeron *et al.*, 2004b). This involved fitting the following negative exponential model of survivorship to TSF estimates (n = 129):

[1] $A(t) = \exp[-(t/b)^c]$

where A(t) is the cumulative proportion of the landscape surviving longer than time t; b, the scale parameter, is an estimate of the fire cycle; and, c, the shape parameter, is equal to one. The key assumption of the negative exponential model is that the hazard of burning is constant with time, i.e. forest age does not influence the instantaneous probability of burning (Johnson & Gutsell, 1994). This assumption is supported by research showing most large and severe fires burn when fire weather is extreme (Stocks, 1991; Johnson *et al.*, 1998); during such periods, fuel loads and stand age become less important than weather characteristics such as wind speed and direction in determining the size and severity of fires (Bessie & Johnson, 1995; Hély *et al.*, 2001).

Survival statistics

We estimated the fire cycle using survival analysis by fitting our TSF estimates to the exponential model. We used the PROC LIFEREG procedure in SAS 8.2 (SAS Institute, 1999) to compare cumulative TSF distributions and assess whether the fire cycle varied as a function of spatial and temporal variables. For each model, a Lagrange multiplier χ^2 test determined whether its hazard function was constant over time. Another χ^2 test evaluated whether the fire cycle estimates of different groups (pine-dominated stands, surficial deposits, etc.) were significantly different (Allison, 1995).

To examine for temporal partitioning in fire cycle, we split the data set into different periods and calculated the fire cycle for each period. To study settlement effects, we partitioned the data set into three periods: pre-settlement (before 1880), settlement (1880-1924) and post-settlement (after 1924). A different analysis involved splitting the entire data set into two periods, before and after the end of the Little Ice Age (1850), to examine whether the fire cycle changed when a regional climatic shift impacted fire regimes throughout eastern Canada (Girardin *et al.*, 2004; Bergeron *et al.*, 2004b). For the post-settlement period, the portion of the study area that had burned before 1924 was considered censured at 79 years; for the settlement period, the portion that had burned before 1880 or after 1924 were censured; and for the pre-settlement period, the portion that had burned before 1880 was censured. A similar process was implemented relative to 1850. This process allowed inter-period comparisons by treating each period distinctly but simultaneously (cf. Bergeron *et al.*, 2004b).

To understand whether spatial partitioning exists in how fire burns the Témiscamingue forest, we estimated the fire cycle for sites that differed in surficial deposits (glacial, fluvioglacial or other), canopy species dominance (pine species or other), and ecological type. Ecological type is a site-scale category of ecosystem classification determined by a combination of potential vegetation and local physical characteristics such as topographic position, drainage and surface deposit (Gosselin, 2002). We examined the following ecological types:

- Sugar maple-yellow birch on shallow to deep deposits of moderate texture and mesic drainage (FE32, n = 32)
- Yellow birch-balsam fir-sugar maple on very shallow to deep deposits of varied texture and xeric to hydric drainage (MJ1, n = 52)
- Balsam fir-yellow birch on shallow to deep deposits, of moderate to coarse texture and xeric to subhydric drainage (MJ2, n = 25)
- Pine or balsam fir on very shallow to deep deposits of varied texture and xeric to subhydric drainage (RP1, n = 20)

Canopy species composition

Two analyses were performed to examine variation of species composition in the canopy: K-means partitioning, a non-hierarchical, iterative clustering method based on a least-squares partitioning technique (MacQueen, 1967; Legendre & Legendre, 1998), followed by Correspondence Analysis (CA), an indirect, unimodal ordination method that preserves the χ^2 distance between objects (Legendre & Legendre, 1998). We first performed the K-means technique (1000 permutations; 2-10 groups) to examine how the sites we field sampled cluster according to canopy species composition (n = 106). With a CA, we then characterized how variation in species composition is partitioned at the different sites. To circumvent the double-zero problem inherent to species abundance or presence/absence data, i.e. nonsensical interpretation of the absence of species from two different sites (Legendre & Legendre, 1998), we first calculated an asymmetrical similarity coefficient among the sites. We then calculated a distance coefficient, ran a Principal Coordinate Analysis and finally the K-means procedure on the table of principal coordinates (Legendre & Legendre, 1998). These analyses were performed using Progiciel R (Casgrain & Legendre, 2001).

Canopy species composition, TSF, and environmental and site history variables

We examined the relative role of TSF in explaining variation in canopy species composition with a ReDundancy Analysis (RDA). RDA is a multivariate ordination technique that allows the simultaneous comparison of a matrix of response variables (**Y**) and of a matrix of explanatory variables (**X**) while structuring of information by separating systematic variation from variation that is not important (ter Braak & Verdonschot, 1995). CANOCO – the program used to perform the canonical analyses – allows the user to determine the importance of individual environmental variables in explaining the variance in **Y** through the use of forward selection of variables (ter Braak & Smilauer, 1998). During the selection process, we used CANOCO to test statistical significance by 1000 Monte Carlo random permutations using the full model with all variables. Since rare species can greatly influence the results of RDA (Legendre & Gallagher, 2001), we excluded species present at five sample sites or less.

The matrix of explanatory variables contained six environmental and two site history variables. The environmental variables were: percent slope, aspect, elevation, surficial deposit type, latitude, and longitude. We coded aspect as a semi-quantitative variable by using two variables that calculate the cosine and sine values for each of eight cardinal directions, thereby presenting aspect as a circular rather than linear variable (P. Legendre, Université de Montréal, *pers. comm.*). Elevation (meters above sea level), surficial deposit type (coarse fluvioglacial, fine fluvioglacial, glacial till, rocky glacial till, glaciolacustrine, rocky), and latitude and longitude (meters from origin of MTM 83 Zone 10 mapping projection) were obtained from the 1996 provincial digital forest inventory (Ministère des ressources naturelles et de la faune du Québec, 2004).

The examined site history variables were TSF and harvesting history. TSF is a quantitative variable measured, as described above, in years before 2004. Harvesting history is a semi-quantitative variable reflecting the frequency of partial cuttings, determined from provincial harvesting records corroborated with field observations. Since 1970, harvesting in the deciduous stands of Témiscamingue has been principally single-tree and group selection or diameter-limit cut. Previous to 1970, logging consisted principally of selective cutting of yellow birch and dominant conifers such as white pine, red pine and white spruce (Vincent, 1995).

We conducted a Hellinger transformation on the species data to deal with the double-zero problem mentioned above (Legendre & Legendre, 1998; Legendre & Gallagher, 2001). The environmental variables were standardized automatically by CANOCO (ter Braak & Smilauer, 1998) to remove the scale effects of the physical dimensions of variables and thereby allow comparisons of the regression coefficients with each other (Legendre & Legendre, 1998). Significance was set at p-value $\leq$ 0.05.

2.5 RESULTS

Fire history reconstruction

The TSF map shows that, within the recent past, large severe fires represent only a fraction of the total area, roughly 20% (Fig. 2.1). No large fires (> 200 ha) were detected since the mid-1920s, with the exception of a 384-ha fire in 1972 that burned in the northwest section of the landscape (Fig. 2.1). Most of the fires (18,504 ha in total) occurred between 1870 and 1890. The largest fire (11,035 ha), in the northeast of the study area, was contiguous with a much larger fire dated to 1921 (Grenier *et al.*, 2005). This pattern was reflected in the forest age distribution, which shows a peak of stand initiation beginning about 1880 (Fig. 2.2a).

This paucity of large severe fire was corroborated by the cumulative TSF distribution (Fig. 2.2b). The survival probability was very close to 100% for the first 76 yr and only decreased to 58% by the end of the 413-yr period of study, meaning the majority of the landscape remained unburned by large severe fires. The rate of burning, as indicated by the slope of the fitted line, was low up to 150 yr ago, increased at the turn of the century, and decreased to its very low level at present from approximately 1928 to 1950. Overall, the fire cycle estimated by the negative exponential distribution was 494 yr (95% CI: 373-694) (Table 2.1). The mean TSF, an estimate biased downwards by censored data, was 213 ± 8 yr (± SE).

Most of the fires detected and dated coincided with the settlement period in Témiscamingue. Two sharp breaks evident in the cumulative TSF distribution indicated a sudden drop in survivorship, one beginning in 1922 (TSF = 82 yr before 2004) and the other in 1884 (TSF = 120 yr) (Fig. 2.2b). Partitioning the temporal record into three periods (pre-settlement, settlement, and post-settlement) indicated a long pre-settlement fire cycle, an elevated fire cycle during settlement, and then a return to a long fire cycle since the mid-1920s (Table 2.1). A similar partitioning relative to the end of the Little Ice Age indicated a longer fire cycle before 1850 than afterwards (Table 2.1).

In terms of spatial partitioning, the fire cycle was shorter on pine-dominated sites than sites dominated by other species (Table 2.2). Moreover, the fire cycle was shorter on sites with surficial deposits of fluvioglacial origin than on sites with glacial deposits. In terms of ecological type, sugar maple-dominated stands (FE32) showed the longest fire cycle, pine and balsam fir-dominated sites (RP2) showed the shortest fire cycle while yellow birch-balsam fir stands (MJ1, MJ2) were intermediate (Table 2.2).

TSF and canopy species composition

At least two distinct assemblages of canopy tree species are present in this landscape. The best partition detected by K-means clustering, in the least squares sense, was for two types of sites. This partitioning had the highest Calinski-Harabasz pseudo-F-statistic (Calinski & Harabasz, 1974; Casgrain & Legendre, 2001). As indicated in the CA ordination by the position of various species along the abscissa (canonical axis 1), canopy species composition varies primarily between sites composed of late-seral, shade-tolerant, thin-barked and fire-avoiding species (i.e. sugar maple, yellow birch, American beech (*Fagus grandifolia* Ehrh.), eastern hemlock, eastern white cedar) versus sites composed of early-seral, fire-adapted species (i.e. red pine, white birch, trembling aspen, large tooth aspen) on the other (Fig. 2.3).

The environmental and site history variables significantly explained about 35% of the variation in canopy species composition (Trace = .347, F-ratio = 3.810, p-value = 0.001). The first two canonical axes (the abscissa and ordinate on Fig. 2.4) captured 79% of the cumulative percent variance of the species-environment relationship. In terms of specific environmental and site history variables, TSF explained the largest amount of the variation in species composition of the canopy (Table 2.3). Several other environmental variables also significantly explained smaller amounts of the variation in canopy species composition: elevation, longitude, latitude, logging history, presence of coarse fluvioglacial deposits (dep_cofl) and presence of undifferentiated rocky glacial tills (dep_tilr) (Table 2.3). Sugar maple, yellow birch and eastern hemlock are positively associated with sites of longer TSF and higher elevation, and, to a lesser extent, with finer textured, unsorted tills put in place without major intervention by glacial meltwaters (Fig. 2.4). In contrast, red and white pine, white birch and aspen species are positively associated short TSF, lower elevation and sandy to gravelly, coarse-textured, sorted, well- to excessively drained soils originating from glacial meltwaters (Fig. 2.4).

2.6 DISCUSSION

Temporal trends in fire frequency

Our results provide evidence that in southern Témiscamingue, as in other areas of eastern Canada, European settlement during the latter half of the 19th century profoundly accelerated fire frequency. Fires originating from burning of logging slash, railroad sparks, land clearing and mineral prospecting created an era of elevated disturbance that affected hundreds of square kilometres in this landscape. It is, however, not possible to discern from our data whether the pronounced shortening of fire cycle was due to heightened human activity alone or some combination of human activity and climate, since settlement coincides with a period of frequent extreme fire weather and severe drought described for the boreal forest of western Québec (Bergeron & Archambault, 1993; Bergeron *et al.*, 2004a; Girardin *et al.*, 2004).

We found the fire cycle was shorter after the end of the Little Ice Age than before. This finding is inconsistent with other studies in western Québec indicating a lengthening of fire cycles since the onset of the wetter and warmer period following the Little Ice Age (Bergeron & Archambault, 1993; Bergeron *et al.*, 2004a). This inconsistency may be due to settlement fires overwhelming the fire record and obscuring a climate effect. It is also plausible that a longer lag exists in deciduous-dominated landscapes, relative to the boreal, for the fire regime to reflect climatic changes. Since deciduous-dominated landscapes are less likely to burn (Hély *et al.*, 2001, Cleland *et al.* 2004), the regime of large severe fire may depend on extreme weather conditions that occur relatively less frequently than those conducive to such fires in the boreal forest. If this is the case, then our temporal record may not be long enough to detect climate-induced changes in fire regime.

Since European settlement ended, the landscape has experienced a longer fire cycle. This elongation is possibly due to an improvement in the efficacy of fire suppression through improved road access, detection networks, and fire-fighting technology as well as a consequence of climate change since the end of the Little Ice Age, during which a gradual increase in average annual temperatures has meant an increase in summertime precipitation and lowered risk of catastrophic fire in the region (Bergeron & Archambault, 1993; Bergeron *et al.*, 2001)

Hazard of burning

Our use of the negative exponential model to estimate fire cycle assumes a constant hazard of burning and that large severe fires burn irrespective of fuel conditions or stand age (Van Wagner, 1978). This assumption is not without debate in the fire ecology literature, where at least two hypotheses have been proposed to characterize how fires burn forests: the *weather hypothesis* (large severe fires are a function of extreme fire weather and burn irrespective of fuel conditions) and the *fuels hypothesis* (variation in fuel types affects fire spread or severity) (Cumming, 2001).

Evidence for the weather hypothesis includes work showing surface fire intensity and crown fire initiation are strongly related to fire weather and weakly related to fuel conditions (Bessie & Johnson, 1995; Johnson *et al.*, 1998). Alternatively, several researchers report that the hazard of burning– as determined by ignition potential or fire spread – increases with stand age, distance to natural firebreaks such as lakes, and conifer dominance of stand composition in the boreal forest of Scandinavia and western North America (Schimmel & Granstrom, 1997; Cumming, 2001; Hellberg *et al.*, 2004; Tanskanen *et al.*, 2005); in other words, as assumed by the use of the Weibull function when estimating fire cycle (Johnson & Van Wagner, 1985). These Scandinavian and western North American forests experience surface fire dynamics that are strongly linked to changes in understory fuels – a phenomenon of reduced importance in crown fire-regulated boreal forests where the hazard of burning is less dependent on understory fuel dynamics (Johnson *et al.*, 2001). In the northern hardwood forests of Témiscamingue, the hazard of burning on some sites may actually decrease with time as a shift occurs in the absence of fire from pine-dominated stands towards dominance of shade-tolerant, late-seral, and relatively less flammable hardwoods.

Canopy species composition, TSF, and environmental variables

Our results support the hypothesis that fire absence, as measured by TSF, is the most significant influence on the composition of dominant tree species in the mesic forests of Témiscamingue. TSF significantly explained the variation in canopy species composition that occurs as stands age and their early successional component changes to late seral species. TSF essentially tracks succession, with early seral species in sites with short TSF and late seral species in sites with long TSF (Fig. 2.3). This finding, in combination with the important part of the variation in composition remaining unexplained by redundancy analysis, alludes to the role of local, secondary natural disturbances in determining canopy species composition. Windthrow, low intensity surface fire, spruce budworm (*Choristoneura fumiferana* Clemens), white pine blister rust (*Cronartium ribicola* J.C. Fisch.), forest tent caterpillar (*Malacosoma disstria* Hubner) and other disturbances, acting singly or in combination, enhance compositional heterogeneity by providing opportunities for gap-phase establishment, growth and release of early seral species in old stands and vice versa (Runkle, 1982; Sousa, 1984; Canham, 1989).

Several studies (e.g. Frelich & Reich, 1995; Davis *et al.*, 1998; Carleton, 2003) describe how the distribution of tree species in northern hardwood forest depends on time since catastrophic disturbance. For instance, red pine and white birch, in combination with shade intolerant hardwoods such as aspen are associated with short TSF, whereas eastern hemlock along with shade-tolerant hardwoods such as yellow birch and sugar maple are associated with long TSF. These assemblages, elucidated by the ordination and clustering procedure (Fig. 2.3), may represent different relationships between fire and vegetation. Fire both perpetuates and is perpetuated by species such as red pine and white birch. Alternatively, species associated with long TSF may persist because the probability of fire decreases dramatically once the transition to shade-tolerant species, in particular tolerant hardwoods, has occurred (Frelich, 2002).

Elevation also explained an important component of the variation in canopy species composition, as evidenced by its position along the primary axis in Fig. 2.4. Cold air ponding in the low-elevation valley bottoms of Témiscamingue can create relatively unfavourable conditions for hardwood species because of the impacts of more frequent freeze-thaw events on their growth (Burke *et al.*, 1976; Brown, 1981). Therefore, coniferous species such as white spruce, balsam fir, and eastern white cedar that are better cold-adapted tend to occur at lower elevations while hardwoods typically dominate upper slopes and mesic ridges (Brown, 1981).

Latitude and longitude significantly explain a minor component of the variation in canopy species composition. In this transitional landscape, a north-south gradient exists from northerly coniferous boreal species adapted to cold temperatures, a short growing season, and frequent fire to southern deciduous species such as American beech, ironwood (*Ostrya virginiana* (Mill.) K. Koch), and American basswood (*Tilia americana* L.) that are favoured by longer growing seasons (Rowe, 1972; Currie & Paquin, 1987). The longitudinal gradient may reflect distributional variation of surficial deposits; fluvial scouring by the paleo-Ottawa River exposed large areas of bedrock and left in place expanses of glacial deposits with boulders in the west of the study area while relatively finer tills and organic deposits are more common in the east (Brown, 1981).

Fire frequency and the role of site variables and ecological types

We present evidence for spatial partitioning of fire frequency in this landscape according to surficial deposits, pine-dominance and ecological site type. Surficial deposits influenced both fire cycle and canopy species composition. These deposits affect the drainage and texture of the growing substrate and influence species composition by the total and seasonal availability of soil moisture and nutrients (Hack & Goodlett, 1960; Whitney, 1986). For example, coarse fluvioglacial deposits are positively associated with pines, white birch and aspen species (Fig. 2.4). Therefore, it seems pines, white birch and aspen species are associated with short TSF and fire cycle not only because these species are present on coarse deposits but also because they are abundant after fire on more intermediate sites where sugar maple will eventually dominate the old-growth stages.

Sites on fluvioglacial deposits, of the pine-fir ecological type or dominated by pine showed a shorter fire cycle than sites on undifferentiated glacial tills, of the maple-birch ecological type or dominated by species other than pine. Together, these findings illustrate the mutually-reinforcing and interdependent relationships between soils, tree cover and fire (Schulte *et al.*, 2005). For example, finer-textured soils less prone to seasonal moisture deficits favor the development of fire-avoiding tolerant hardwoods whereas coarser, well-drained soils favor the development of pines and early seral hardwoods that depend on and propagate fire. These inter-relationships suggest that although TSF is an important determinant of stand composition, its role depends on the influence of physiographic setting on fire. In other words, it appears that at least two different fire frequencies exist as a function of physiography and that TSF has a different effect in each of two distinct successional pathways. Generally, a short fire cycle on coarse-deposit sites maintains pines and other fire-dependent species whereas fine-deposit sites more easily allow a long TSF and succession to shade-tolerant species of low flammability.

Logging history

The frequency of partial cutting is positively associated with the maple-birch-hemlock assemblage and negatively associated with relatively less merchantable, boreal conifers such as white spruce and balsam fir (Fig. 2.4). These results are consistent with several studies that indicate selective and partial cutting treatments in hardwood forests favour a preponderance of sugar maple and other shade-tolerant species (Schuler, 2004; Brisson *et al.*, 1988; Bouchard *et al.*, 1989; Smith & Miller, 1987). At the landscape scale, Jackson *et al.* (2000) documented an increase in abundance, as compared to pre-settlement composition, of sugar maple and its associates in the Great Lakes-St. Lawrence hardwood forests of central Ontario, attributing this trend to the historical dominance of partial cutting since the onset of commercial harvesting in this region. We therefore suggest partial logging advances succession and acts to 'fire proof' the landscape by promoting the development of deciduous assemblages at the expense of fire-adapted assemblages and, possibly, by removing structural features such as supra-overstory white pines that may attract lightning strikes. Moreover, pine removal by logging may entrain a shift in fire regime by altering the structural and compositional aspects of stands that are conducive to the frequent, low-intensity ground fires that maintain pine-dominated stands (Engstrom & Mann, 1991; Ryan, 2002).

Selective logging of pines and spruce may have removed evidence of fire on some sites. If these species previously had a wider distribution in this area, the fire cycle of southern Témiscamingue may have been shorter than our pre-settlement estimate of 909 yr (Table 2.1). For example, Whitney (1986; 1987) reports that pure and mixed stands of pine covered 45% of the area in a pre-industrial northern Michigan landscape and had a cycle of 129-258 yr for large severe fire, while the entire landscape had a fire cycle of 205-411 yr. It seems unlikely that partial cutting affected the TSF estimates on sites where we relied on the oldest living tree; these silvicultural treatments, as implemented in Témiscamingue, often retain the full range of tree ages found before harvest.

2.7 CONCLUSION

Currently the landscape is experiencing a longer fire cycle relative to historical rates. The lack of fire may be diminishing the competitive advantage that pines and other fire-adapted species have over dominant deciduous trees; this diminution may be especially relevant in intermediate sites where the physiographic factors or microclimatic conditions do not overwhelmingly favour one suite of species over another and allow a wide variety of species assemblages. In other words, the fire-dependence of coarse-deposit sites dominated by pines, in combination with their high timber value, makes them more vulnerable to permanent change than sugar maple-yellow birch sites. Therefore, our prognosis for landscape development is for increasing dominance of fire-avoiding species and decreasing abundance of fire-maintained pine stands and their associated floral and faunal communities. Such a decrease in the abundance of white and red pine communities has already been documented throughout their range (Noss *et al.*, 1995; Radeloff et al., 1999; Zhang *et al.*, 1999; Thompson *et al.* 2006) and suggests these sites should be a conservation focus in maintaining the historical diversity of stand types.

The above prognosis has several management implications. Current harvesting regulations mandate the exclusive use of single-tree selection in the hardwood forests of Québec. Under a natural disturbance-based approach to forest management (Hunter, 1993; Bergeron *et al.*, 1999), this silvicultural system is appropriate over most of the landscape given the dominance of late-seral stands and gap-phase dynamics in this forest type. However, large and severe fires were previously important disturbances in this landscape. We therefore recommend a diversification of forest practices to include some low to mid-retention silvicultural systems that resemble the size, pattern, frequency, and within-burn legacies of large severe fires, thereby more closely emulating the full range of natural disturbances. Moreover, these silvicultural systems are suitable for the regeneration of drought-tolerant species like red and white pine, red maple, and red oak and may enhance landscape resilience to possible increases in summertime drought resulting from climate change (Boer *et al.*, 2000).

2.8 ACKNOWLEDGEMENTS

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2.9 TABLES

Table 2.1 Temporal partitioning of fire cycle in relation to European settlement and the Little Ice Age in Témiscamingue, QC

Period	***n***	**Number of detected fires**	**Fire cycle[1] (yr)**	**Survival analysis ($p > \chi^2$)**	**Lagrange statistic ($p > \chi^2$)**
Overall (1591-2004)	129	49	494 (373-694)	< 0.001	< 0.001
Pre-settlement (<1880)	83	13	909 (489-1690)	< 0.001	0.020
Settlement (1880-1924)	127	33	153 (108-218)	< 0.001	0.624
Post-settlement (>1924)	129	2	5081 (3593-20318)	< 0.001	n\a
Little Ice Age (< 1850)	71	3	2322 (748-7199)	0.013	0.9491
Post-Little Ice Age (> 1850)	129	46	375 (281-501)	0.013	n\a

[1]Lower and upper 95% CI in brackets

Table 2.2 Spatial partitioning of fire cycle in relation to pine canopy dominance, surficial deposits and ecological type in Témiscamingue, QC. The ecological types are: FE32: Sugar maple-yellow birch on shallow to deep deposits of moderate texture and mesic drainage; MJ1: Yellow birch-balsam fir-sugar maple on very shallow to deep deposits of varied texture and xeric to hydric drainage; MJ2: Balsam fir-yellow birch on shallow to deep deposits, of moderate to coarse texture and xeric to subhydric drainage; and RP1: Pine or balsam fir on very shallow to deep deposits of varied texture and xeric to subhydric drainage.

Strata	***n***	**Number of detected fires**	**Strata effects ($p > \chi^2$)**	**Fire cycle[1] (yr)**	**Survival analysis ($p > \chi^2$)**	**Lagrange statistic ($p > \chi^2$)**
Canopy dominance			< 0.001			< 0.001
Pine spp.	18	15		165 (100-274)	< 0.001	
Other spp.	111	34		640 (210-1948)	< 0.001	
Surficial deposits			0.018			< 0.001
Fluvioglacial	14	4		216 (48-965)	0.114	
Glacial	90	29		610 (160-815)	0.368	
Other	25	16		438 (236-815)	< 0.001	
Ecological type			0.004			<.0001
FE32	32	4		918 (292-2887)	<0.001	
MJ1	52	20		362 (156-836)	0.066	
MJ2	25	13		259 (108-621)	0.683	
RP1	20	12		233 (161-336)	<0.001	

[1]Lower and upper 95% CI in parenthesis

Table 2.3 Results for Redundancy analyses (with forward selection of variables) for significant environmental variables influencing canopy species composition in southern Témiscamingue, QC.

Environmental variable	**Variance explained**	**F-ratio**	**P-value**
Time-since-fire (TSF)	.17	20.95	.0010
Elevation (Elev)	.04	5.83	.0010
Longitude (Long)	.02	2.84	.0080
Latitude (Lat)	.02	2.29	.0120
Coarse fluvioglacial deposits (dep_cofl)	.02	3.36	.0020
Logging history (logghist)	.02	3.22	.0030
Undifferentiated rocky glacial tills (dep_tilr)	.01	1.85	.0320

2.10 FIGURES

Figure 2.1 Time-since-fire map for study area.

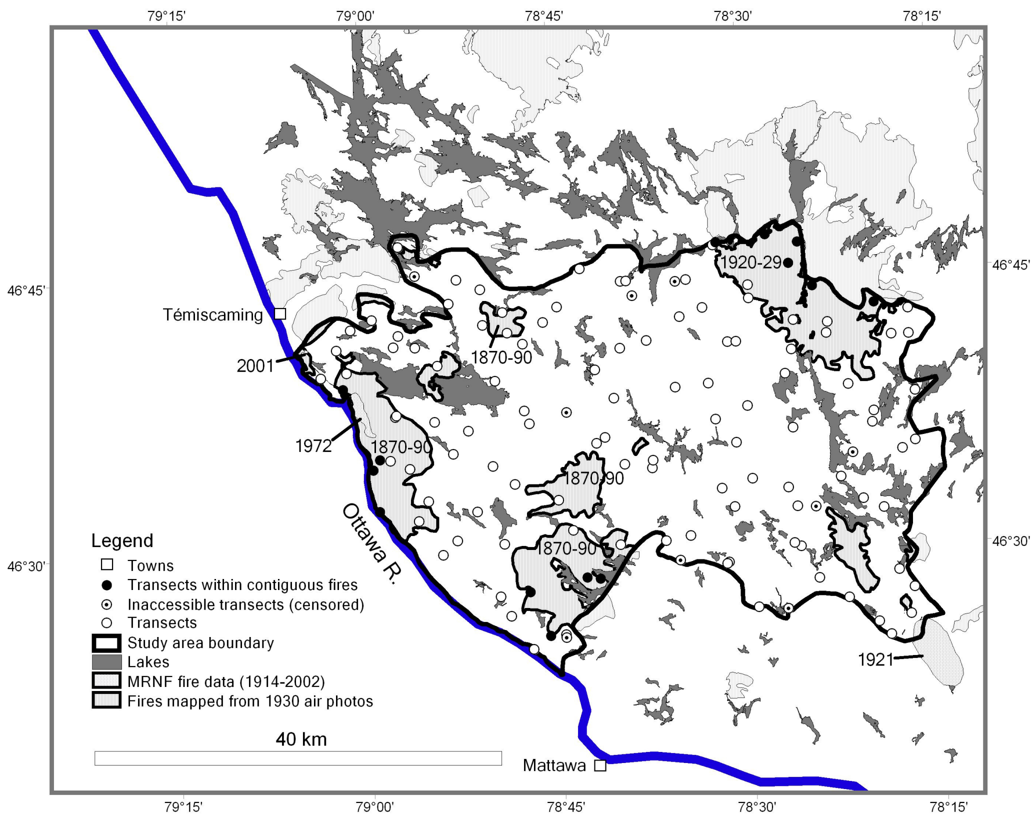


Figure 2.2 (a) Forest age distribution and (b) cumulative survivorship distribution for southern Témiscamingue, QC.

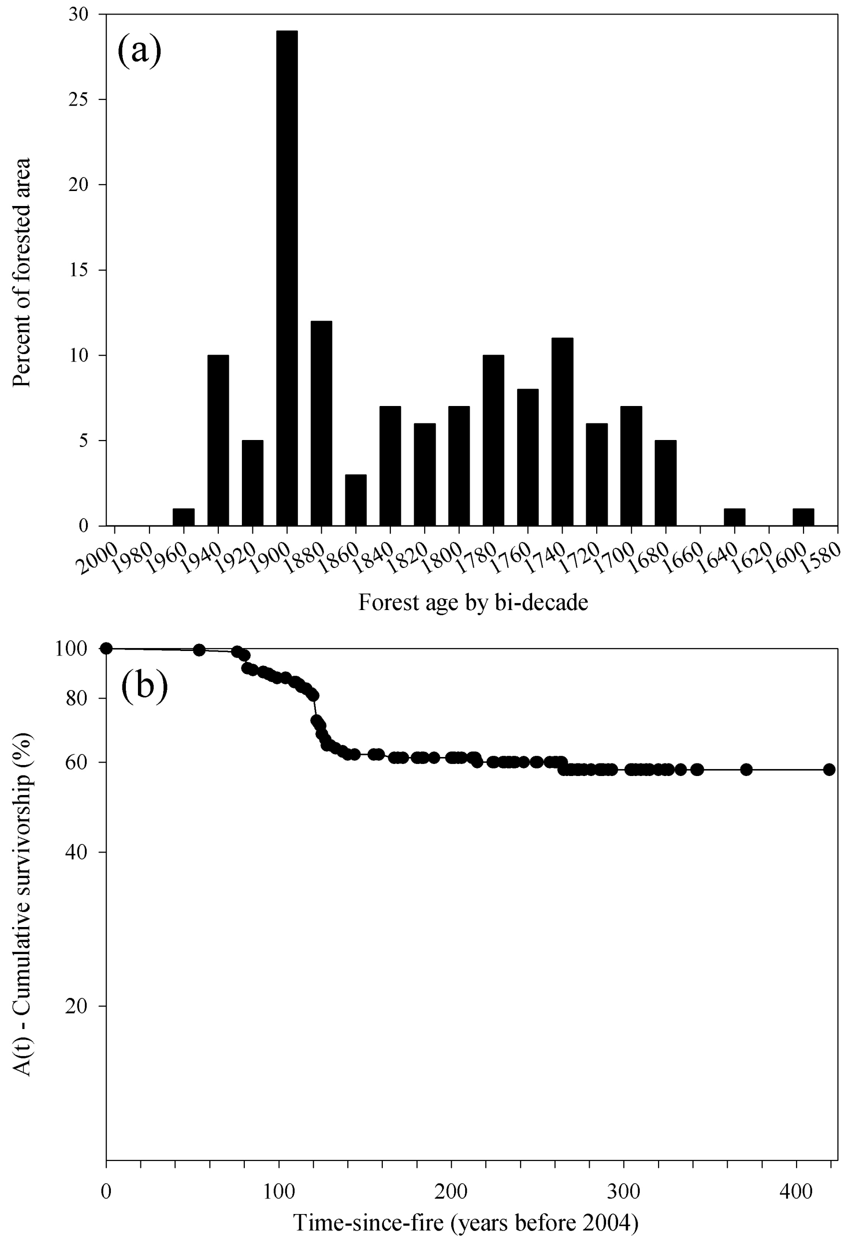
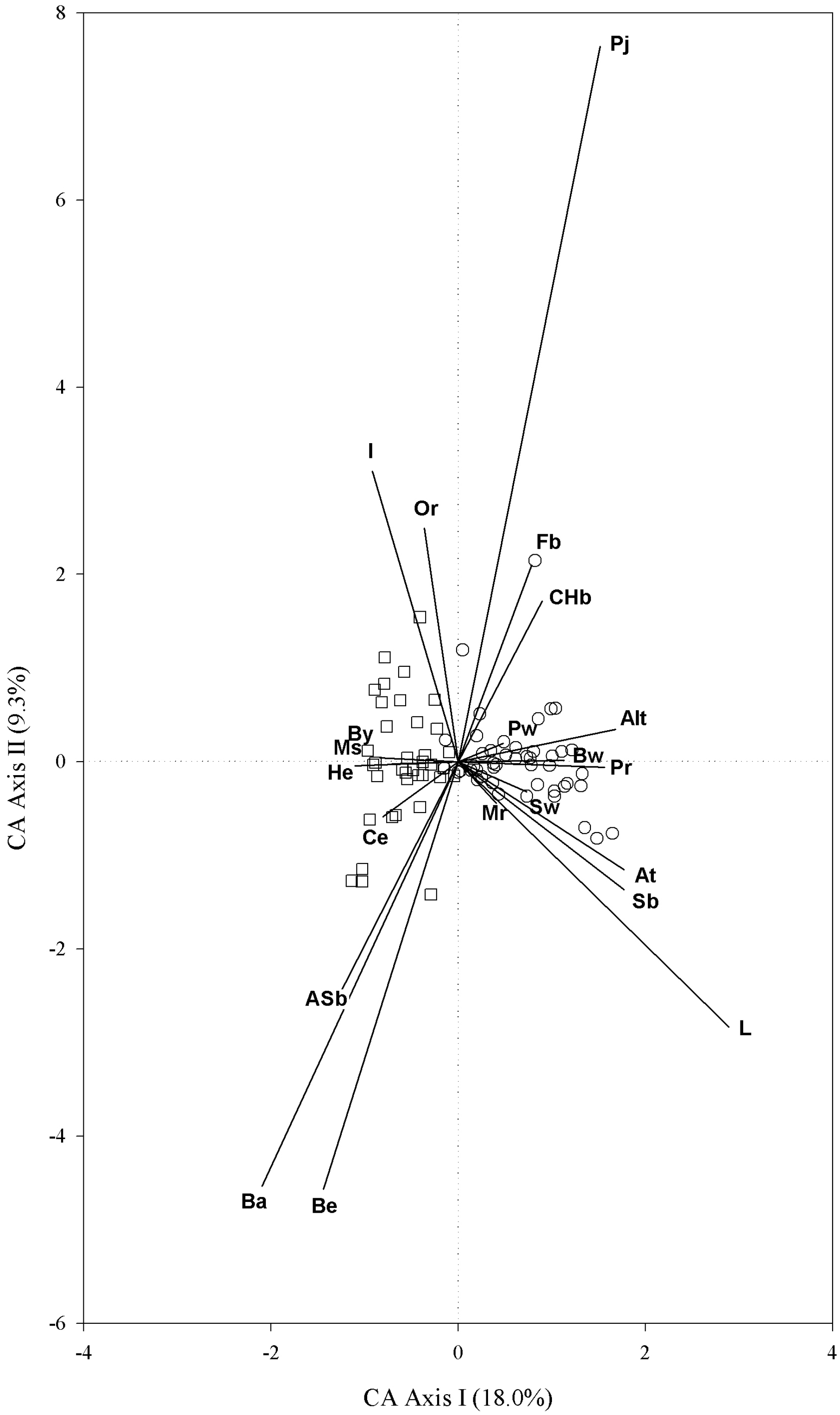


Figure 2.3 K-means clustering and correspondence analysis (CA) according to canopy species composition (n = 106). Type 1 scaling type was used to illustrate the differences among sites. The circles and squares denote the two site types detected by K-means. Species codes are: ASb, black ash (*Fraxinus nigra*); Ba, basswood (*Tilia americana*); Be, American beech (*Fagus grandifolia*); Bw, white birch (*Betula papyrifera*); By, yellow birch (*Betula alleghaniensis*); CHb, black cherry (*Prunus serotina*); Ce, eastern whitecedar, (*Thuja occidentalis*); Fb, Balsam fir (*Abies balsamea*); He, eastern hemlock (*Tsuga canadensis*); I, ironwood (*Ostrya virginiana*); L, tamarack (*Larix laricina*); Ms, sugar maple (*Acer saccharum*); Mr, red maple (*Acer rubrum*); Or, red oak (*Quercus rubra*); Alt, large tooth aspen (*Populus grandidentata*); Pj, jack pine (*Pinus banksiana*); Pr, red pine (*Pinus resinosa*); At, trembling aspen (*Populus tremuloides*); Pw, eastern white pine (*Pinus strobus*); Sb, black spruce (*Picea mariana*); and Sw, white spruce (*Picea glauca*).


Pj
I
Or
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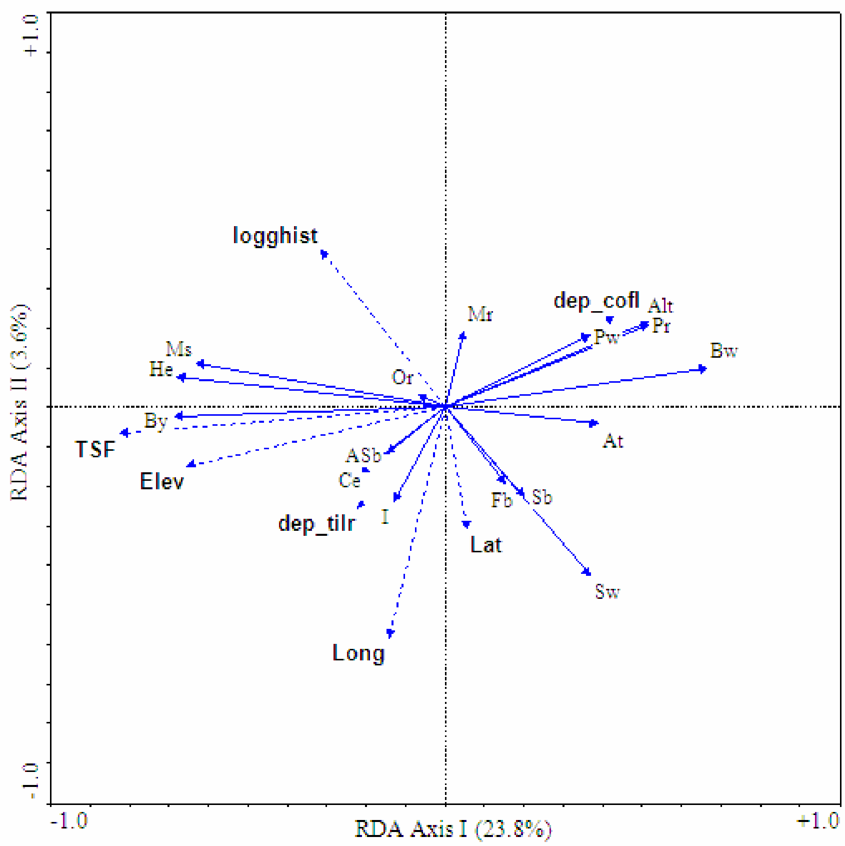
ASb
L
Ba
Be
8
6
4
2
0
-2
-4
-6
-4
-2
0
2
4
CA Axis II (9.3%)
CA Axis I (18.0%)

Figure 2.4 Bi-plot of redundancy analysis (RDA) of canopy species composition (n = 106) with significant environmental and site history variables. Type 2 scaling was used to highlight differences among the species. The environmental and site history variables are elevation (Elev), longitude (Long), latitude (Lat), presence of coarse fluvioglacial deposits (dep_cofl), presence of undifferentiated rocky glacial tills (dep_tilr), time-since-fire (TSF) and logging history (logghist). Species codes as per Fig. 2.3.



CHAPITRE 3

FIRE AND THE RELATIVE ROLES OF WEATHER, CLIMATE AND LANDSCAPE CHARACTERISTICS IN THE GREAT LAKES-ST. LAWRENCE FOREST OF CANADA

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3.1 ABSTRACT

Question: In deciduous-dominated forest landscapes, what are the relative roles of fire weather, climate, human and biophysical landscape characteristics for explaining variation in large fire occurrence and area burned?

Location: The Great Lakes-St. Lawrence forest of Canada.

Methods: We characterized the recent (1959-1999) regime of large (≥ 200 ha) fires in 26 deciduous-dominated landscapes and analyzed these data in an information-theoretic framework to compare six hypotheses that related fire occurrence and area burned to fire weather severity, climate normals, population and road densities, and enduring characteristics such as surficial deposits and large lakes.

Results: Three-hundred ninety two large fires burned 833 698 ha during the study period, burning on average 0.07% ± 0.42% (± SD) of forested area in each landscape annually. Fire activity was strongly seasonal, with most fires and area burned occurring in May and June. A combination of antecedent-winter precipitation, fire season precipitation deficit/surplus and percent of landscape covered by well-drained surficial deposits best explained fire occurrence and area burned. Fire occurrence varied only as a function of fire weather/climate variables, whereas area burned was also explained by percent cover of aspen and pine stands, human population density and two enduring landscape characteristics: percent cover of large water bodies and glaciofluvial deposits.

Conclusion: Understanding the relative role of these variables may help design adaptation strategies for forecasted increases in fire weather severity by allowing prioritization of landscapes according to enduring characteristics and management of their composition so that substantially increased fire activity would be necessary to transform landscape structure and composition.

3.2 RESUME

Question : Dans des paysages dominés par des forêts feuillues, quels sont les rôles relatifs des caractéristiques de temps, de climat, des caractéristiques humaines et biophysiques du paysage pour expliquer la variation de l’occurrence du feu et la superficie brûlée ? Aire d’étude: La forêt des Grands-Lacs/Fleuve St-Laurent du Canada.

Méthodes : Nous avons caractérisé le régime récent (1959-1999) des grands feux (≥ 200 ha) dans 26 paysages dominés par des forêts feuillues et avons analysé ces données selon la théorie de l’information pour comparer six hypothèses qui mettent en relation l'occurrence de feu et la superficie brûlée avec les conditions météorologiques propices au feu, le climat, les densités de la population et des chemins et les caractéristiques permanentes, telles que les dépôts de surface et la présence de grands lacs.

Résultats : 392 grands feux ont brûlé 833 698 ha pendant la période d'étude, brûlant en moyenne 0,07 % ± 0,42 % (± écart-type) de la superficie couverte de forêts dans chaque paysage annuellement. L'activité du feu était fortement saisonnière, avec la plupart des feux brûlant en mai et juin. Une combinaison des précipitations de l’hiver précédent, du déficit/excédent de précipitations pendant la saison du feu et de la proportion de paysage couvert par les dépôts de surface bien drainés a mieux expliqué l'occurrence du feu et la superficie brûlée. L'occurrence du feu a changé seulement en fonction des variables climatiques, tandis que la superficie brûlée a également été expliquée par la couverture de peuplements de tremble et de pin, la densité de population humaine et deux caractéristiques permanentes du paysage : la présence de grands lacs et de dépôts fluvioglaciaires.

Conclusion : Pouvoir comprendre le rôle relatif de ces variables peut aider à concevoir des stratégies d'adaptation pour des augmentations prévues des conditions météorologiques propices au feu, par exemple, par permettant la priorisation des paysages selon leurs caractéristiques permanentes et de gérer leur composition de sorte que l'activité sensiblement accrue du feu soit nécessaire pour transformer la structure et la composition du paysage.

3.3 INTRODUCTION

The presence of extreme weather conditions conducive to fire ignition and spread is paramount in determining the occurrence and final size of large fires (Johnson & Larsen 1991; Diez et al. 1993). For example, drought is associated with both large fire incidence and annual area burned (AAB) in several different forest types (Balling et al. 1992; Girardin et al. 2004; Pereira et al. 2005). Other critical weather variables affecting AAB include temperature, wind speed, and relative humidity (Flannigan & Harrington 1988; Bessie & Johnson 1995; Taylor et al. 2004).

Fire weather is however not the only determinant of fire frequency or area burned. Over the course of one or several fire seasons, area burned can depend on a variety of landscape characteristics, including the arrangement and type of forest fuels, ignitions, topography, suppression effort, and abundance and distribution of firebreaks such as lakes or agricultural land (Hély et al. 2001; Ryan 2002; Lefort et al. 2003). While these variables have been the focus of much boreal forest ecology, we found no studies examining relationships among fire, weather and landscape characteristics in the northern hardwood forests of eastern North America where, although relatively rare, these fires do occur (Whitney 1987; Drever et al. 2006).

Our study goals were (i) to characterize the recent history of fires and (ii) evaluate competing hypotheses regarding the relative importance of fire weather, climate and biophysical characteristics in explaining variation of fire occurrence and AAB in the Great Lakes-St. Lawrence forest region. Our hypotheses were:

1. *Re-charge hypothesis.* Fire occurrence and AAB depend on moisture input and retention capacity of the landscape, as a function of precipitation accumulated during the winter before the fire season, precipitation deficit/surplus (total precipitation - potential evapotranspiration) during the fire season, and the percentage of the landscape covered by sorted, well-drained glaciofluvial or undivided surficial deposits.
2. *Fire break hypothesis.* Fire occurrence and AAB depend on the amount and type of landscape features that have low flammability and reduce fire spread,

i.e. aspen (*Populus* spp.) stands, large lakes and rivers, urban spaces, cropland or other non-burnable surfaces, and the percentage of the landscape covered by wetlands and ridged landforms.

3. *Human influence and access hypothesis.* Fire occurrence and AAB vary as a function of population and road densities. In landscapes with high human and road densities, increased human health and property concerns as well as increased detection, access and prioritization by fire fighting personnel result in lower AAB than in landscapes where road and human densities are relatively less.
4. *Fuels hypothesis.* Fire occurrence and AAB depend on the relative coverage of flammable fuel types such as boreal spruce stands vs. that of less-flammable fuel types such as aspen stands, agricultural matrix, and urban centres.
5. *Fire weather hypothesis.* Fire occurrence and AAB are a function of maximum fire weather severity as estimated by the yearly maximum value of the Canadian Drought Code (Turner 1972). Larger and more frequent fires burn during years when fire weather is severe.
6. *Amenable climate hypothesis.* Fire occurrence and AAB vary as a function of climatic conditions amenable to growth of the deciduous component in the mixed forests of our region: growing season length (landscapes with growing seasons that begin early in the spring allow the development of relatively less-flammable deciduous canopies), precipitation and potential evapotranspiration, and lightning density during the fire season.

Contrasting the roles of enduring landscape characteristics such as surficial deposits relative to ephemeral severe weather events in explaining variation in fire occurrence and area burned can shed light on which landscapes have a fire regime most intimately tied to weather and thereby may be most susceptible to climate change-related alterations in fire activity e.g. Bergeron et al. (2006).

3.4 METHODS

Study area: Great Lakes-St. Lawrence forest region

The Great Lakes-St. Lawrence forest region (Fig. 3.1; modified from (Rowe 1972)) has forests transitional between the boreal forest to the north and Carolinian forests to the south. The region has generally low relief with rolling and forested hills, many lakes, wetlands, outwash plains and other glacial features. Elevation ranges from zero to approximately 1120 m above sea level, with a mean of 361 m. It is variously classified as the Mixedwood Shield (CEC 1997), the western Sugar maple-Yellow birch bioclimatic domain in Québec (Robitaille & Saucier 1998) and the Laurentian Mixed Forest Province in the United States (Bailey 1995). Sugar maple (*Acer saccharum* Marsh.) and yellow birch (*Betula alleghaniensis* Britt.) typically dominate mesic, mid-slope sites. Other common tree species include eastern hemlock (*Tsuga canadensis* (L.) Carr.), white birch (*B. papyrifera* Marsh.), aspen, white spruce (*Picea glauca* (Moench) Voss), balsam fir (*Abies balsamea* (L.) Mill.), red pine (*Pinus resinosa* Ait.) and white pine (*P. strobus* L.). Forests cover the vast majority (98%) of the land area. The total area of the study region was 40 596 481 ha.

Units of observation

We used ecodistrict-year i.e. a given year in an ecodistrict, as the observational unit. Ecodistricts (Fig. 3.1), the finest level of the classification in the Canadian Ecological Framework, are landscape units characterized by distinctive assemblages of relief, geology, landforms and soils, vegetation, water, fauna, and land use (Ecological Stratification Working Group 1996). Mean ecodistrict size was 1 353 190 ± 1 011 262 ha (± SD). The fire season consisted of the months between April and October, inclusive.

Proportion annual area burned (PAAB)

Annual area burned for each ecodistrict was calculated from the Canadian Large Fire Database (LFDB). The LFDB is a compilation of all large forest fires in Canada

(1959-1999) as reported by provincial and territorial agencies as well as Parks Canada (Canadian Forest Service 2002). Although the LFDB includes only fires > 200 ha, these fires account for approximately 97 % of area burned in Canada (Stocks et al. 2002). Our analysis included both human and lightning-caused fires as well as fires of unknown origin. We modeled proportion of total forested area burned each year in each ecodistrict (PAAB) as the dependent variable to account for size differences among ecodistricts. Using only large fires reduced the effect of increasing efficiency in fire detection and suppression during the period of record, as well as the confounding influence on the relationships between the explanatory and fire variables arising from very small, localized fires that do not respond to environmental conditions (Lefort et al. 2004).

Independent variables

We examined four types of independent variables: human, biophysical, climate and fire weather (Table 3.1). While these independent variables are interrelated and some variables are included in more than one model, we ensured correlations among the independent variables were below levels high enough to cause collinearity problems within our models (Burnham & Anderson 2002). For the forest fuels, we lumped the three pine classes (red and white pine, mature jack pine, and immature jack pine) into a general pine type as each class represented only a small fraction of each ecodistrict. For road density, we used the 2001 Road Network file to estimate road density as it is the earliest available spatial database that provides coverage of both rural and urban areas for the three provinces in the study area.

To estimate yearly precipitation and the severity of fire weather, we compiled daily total precipitation and maximum temperature data from April to October for 1959 to 1999 from Environment Canada meteorological stations across the study area (Fig. 3.1). Variables for each ecodistrict were based on stations within a 50-km radius of the ecodistrict centroid, as estimated with ARCVIEW 3.2a (ESRI 1996), using the station closest to the centroid first and then filling in missing data with more distant stations. Three ecodistricts (407, 409, 412) were not included in this analysis because no stations exist within 50 km of their centroids.

We screened and corrected the temperature and precipitation data sets for missing daily values. For temperature, a missing daily datum was replaced with the corresponding value from the closest nearby (< 50 km) meteorological station. If such a station was not present or also missing data for the day in question, the mean monthly temperature for that month was used as a substitute. Missing precipitation data were replaced with the value for the same day from nearby stations or mean monthly values as described above.

Canadian Drought Code

We estimated severity of fire weather using the Canadian Drought Code (DC). The DC provides a daily numerical estimate of the weather conditions conducive to fire involving deep, organic, duff layers of forest soil or heavy forest fuels (Turner 1972). This index is calculated on a day-to-day cumulative basis to maintain a ledger of stored moisture by accounting for daily losses through evapotranspiration and gains from precipitation. See Turner (1972); Van Wagner (1987); and Girardin et al. (2004) for computational details. The DC is a significant predictor of fire frequency and area burned in the southern boreal forest of Canada (Girardin et al. 2004). We used yearly maximum DC (based on daily calculations) in our analyses because the occurrence and size of large fires have been shown to be more intimately related to extremes of fire weather rather than to measures of its central tendency (Moritz 1997). Since DC values typically show a seasonal trend – a slowly increasing value reaching an apex in mid- to late August, then either declining or staying the same (McAlpine 1990) – we used only years for which data from the entire fire season were available. Moreover, since the DC is especially responsive to daily precipitation (i.e. one rainfall event can influence the index calculation for several days afterwards; Girardin et al. (2004)), any years with more than five days of missing precipitation data were excluded from the analysis.

Model fitting and selection

Our fire data were markedly non-normal, exhibiting a large clump of values at zero and skewed non-zero values. To deal with this structure, we used a mixed-distribution, mixed-effects model for repeated measures data with clumping at zero and correlated random effects (Tooze et al. 2002; Martin et al. 2005). The mixed-distribution approach combines a model of the occurrence probability of non-zero values (the 'occurrence model', based on logistic regression using all ecodistrict-years) with the probability distribution of nonzero values (the 'area burned' model, based on lognormal regression using ecodistrict-years where PAAB > zero). Using this modeling approach allowed the simultaneous assessment of how different independent variables affected both the occurrence probability of fire and the area burned given that a fire was observed. Mixed-effects models incorporate fixed effects of standard explanatory variables with random effects (correlation and non-constant variability) resulting from repeated, random unit observations (different years) on the same unit (ecodistrict). Both the occurrence and area burned model components incorporated ecodistrict as a random effect.

We used the MIXCORR macro (Tooze et al. 2002) in SAS 8.2 (SAS Institute Inc. 2001). This method relies on maximum likelihood methods for model fitting, estimating the effect of independent variables on the occurrence probability and mean of nonzero values, and estimating parameters for both model components. A lognormal probability distribution was used to characterize the error structure of the area burned model. The model components were combined when calculating the overall likelihood for mixed-distribution model, which is a product of the likelihoods of each component.

The model forms were:

Occurrence: logit(Probability of Fire) = $a_1 + b_1{*}X_1 + c_1{*}X_2 + d_1{*}X_3 \ldots + i_1{*}X_j + u_1$ [1]

Area burned: log(PAAB) = $a_2 + b_2{*}X_1 + c_2{*}X_2 + d_2{*}X_3 \ldots + i_2{*}X_j + u_2 + \sigma_2^2$, [2]

where a_1 and a_2 are intercept parameters; b_1 and b_2 are slope parameters for independent variable X_1; c_1 and c_2 are slope parameters for independent variable X_2, and so on; u_1 is the occurrence random unit effect; u_2 is the area burned random unit effect; and σ_2^2 is the variance of the residuals. In addition, the MIXCORR macro determines a parameter of covariance, ρ, of random effects between the occurrence and area burned components of the model mixture. We considered explanatory variables with slope estimates for which the 95% CI excluded zero as having strong support from the data for an effect on our dependent variable.

We compared the six models based on our hypotheses using an information-theoretic approach, where the goal is to make inferences from data that minimize information loss by selecting the most parsimonious models from a suite of candidate models (Burnham & Anderson 2002). This approach also allowed elucidating what independent variables were essential for providing parsimonious fit. Akaike's Information Criterion modified for small samples (AICc) was calculated for each model, as well as the difference in AICc between each model and the model with the minimum AIC (ΔAICc) and the Akaike weight (w) (Burnham & Anderson 2002). The Akaike weights estimated the probability that each model is the best of the suite and provided the basis for computing the weighted averages of parameter estimates. Model averaging was performed to account for model selection uncertainty (Burnham & Anderson 2002).

3.5 RESULTS

Fire occurrence and area burned

During the fire seasons of 1959 to 1999, 392 large fires burned 833 698 ha in the Great Lakes-St. Lawrence forest of Canada. Fires ≤ 1000 ha were the most frequent (66% of all fires) yet accounted for only 13% of the total area burned (Fig. 3.2a). Conversely, fires > 10 000 ha in size accounted for 49% of the total area burned; one 113 514-ha fire in the western section of the region comprised 14% of the total area burned. Fires showed strong seasonality, with most fires and area burned occurring in May and June (Fig. 3.2b). Most (55%) of the area burned resulted from lightning-caused fires (Fig. 3.2b). Except for punctuated spells of high fire activity such as occurred in 1976, 1980 and 1984, fires were quite variable and generally 6-8 fires burned less than 60 000 ha each year across the study region (Fig. 3.2c). Large fires burned a small fraction of ecodistrict forest area per year, with a mean of 0.07% (SD = 0.42%; range = 0-6.8%) of ecodistrict forest area per year. Lightning density across the study region showed a seasonal trend characterized by an abrupt increase in June, a maximum in July and a subsequent gradual decrease to October (Fig. 3.3). Mean monthly maximum Drought Code across the region increased to a peak in August with a slow decrease until October (Fig. 3.3).

Model selection and parameter estimation

The recharge hypothesis (model 1) received the highest relative support of all the hypotheses considered, having an Akaike weight of 0.75 (Table 3.2). The next best model was model 5, the fire weather hypothesis, which had an Akaike weight of 0.22, meaning that, based on the evidence ratio (evidence of best model: model of interest), the strength of evidence is 3.4 times greater for the recharge hypothesis than for the fire weather hypothesis. All other models had Akaike weights $\leq$0.01, and thus had relatively much lower support from the data.

Model-averaged parameter values suggested several variables had a strong effect on the probability of fire occurrence and on proportion of area burned yearly (Table 3.3). Of the four general classes of variables analyzed, only fire weather and climate variables affected fire occurrence. Growing season length (GSL) and seasonal precipitation deficit/surplus (PptSurpDef, Fig. 3.4) had a negative effect on yearly proportion burned in both the occurrence and area burned components of the model. Positive effects for two other climate-related variables, maximum DC (DC_max) and total previous-winter precipitation (WinterPpt), were supported by the data for the occurrence but not area burned model component. All the landscape variables for which the data provided strong support related to area burned but not fire occurrence. Of these, human population density (PopDen) and percent of the landscape covered by aspen stands (AspenPerc), by large water bodies (WaterPerc), and by pine stands (PinePerc) showed a negative effect on yearly proportion burned, while percent of the landscape covered by glaciofluvial plains had a positive effect (GlacioFlvPlainPerc) (Table 3.3).

The data provided support for a random ecodistrict effect in the occurrence model (u_1), indicating that considerable spatial variation existed in the probability of fire occurrence, i.e. some ecodistricts burned with much greater frequency than others (Table 3.3). The covariance parameter (ρ) between the random effects in the occurrence and area burned model components was 0.29 (Table 3.3), suggesting ecodistricts that burned more often than others did not necessarily burn larger proportions of the forested landscape.

3.6 DISCUSSION

Recent large fires in the Great Lakes-St. Lawrence forest

Fires greater than 200 ha burned approximately 2.5% of the region's forests over the 41-yr period of study. Annual fire activity represented a small fraction of individual landscapes across the region, on average affecting 0.07% of the forest area in each ecodistrict. This fraction is comparable to the annual burn rate reported since 1940 for the mixedwood (0.05%) and deciduous (0.04%) forests in the Québec portion of the study region (Bergeron et al. 2006). It also corresponds well to modern burn rates documented for the northern hardwood landscapes of Michigan, USA (0.03% between 1985-2000; Cleland et al. 2004) and northern New England (0.02% between 1961-1978; Fahey & Reiners 1981). In all these cases, burn rates are an order of magnitude less than the recent burn rates typically reported for boreal landscapes north of the region; for example, Bergeron et al. (2006) documented average yearly burn rates since 1940 for Québec landscapes in Abitibi and central Québec of approximately 0.26%.

In terms of seasonality, the peak of fire activity in the Great Lakes-St. Lawrence forest preceded the July peak in lightning density, occurring during May and June (Fig. 3.2b). A similar seasonal fire trend was observed for recent natural fires in the deciduous forests of the central Appalachian mountains of the United States (Lafon et al. 2005). This situation contrasts what typically occurs in the Canadian boreal forest (Stocks et al. 2002) and western United States (Westerling et al. 2003), where fire frequency and area burned are highest during the hottest and driest period of the annual climatological cycle in July and August.

Fire hazard in our study region thus seems strongly associated with deciduousness, as evidenced by how fires in the study region differ both in their extent and seasonality from adjacent boreal landscapes, despite having similar trends in fire weather severity (Girardin et al. 2004). This association has a temporal dimension related to the onset of broadleaf emergence and growth as well as a spatial dimension related to the relative abundance of deciduous stands in a given landscape. As compared to conifers, deciduous trees and stands possess structural and fuel attributes that impart low flammability, including low bulk density of the canopy, leaves with high moisture content, low concentrations of flammable resins and oils, discontinuity of fuels between the forest floor and tree crowns, high rates of decomposition for coarse woody debris, and relatively fire-retardant fine fuels and litter (Philpot 1970; Van Wagner 1977; Hély et al. 2000; Frelich 2002). The development of a deciduous canopy decreases wind speeds and penetration of solar irradiance to the forest floor, limiting desiccation of forest fuels (Schroeder & Buck 1970; Lafon et al. 2005). Moreover, deciduous stands can limit the intensity and spread of large fires (Hély et al. 2001; Vazquez et al. 2002; Wang 2002). Therefore, from an evolutionary perspective, given the seasonal peak of fire weather severity in our region occurs during July and August – when the deciduous canopy is fully emerged and the forests are least likely to burn – deciduousness in this region may have evolved not only as a drought adaptation (Walter 1973), but also to provide deciduous species with adaptive traits that repel the fires that likely would have accompanied periods of drought.

Hypotheses selection

The recharge hypothesis, where fire occurrence and area burned are a function of precipitation surplus/deficit, glaciofluvial deposits, and total precipitation accumulated during the antecedent winter, had the highest level of data support. The recharge hypothesis was the only model that captured spatial variation related to fuel type and landscape moisture retention (as determined by surficial deposits) with spatial and temporal variation in the climatic conditions that affect when fires burn and how large they become during a given fire season. Moreover, including antecedent precipitation allowed capturing variation of fire hazard attributable to lagged weather effects, effects increasingly recognized as important to long-lead forecasting of fire activity (Westerling et al. 2002; Crimmins & Comrie 2004). The high support for the recharge hypothesis, relative to the rest, provided evidence that large fires in the Great Lakes-St. Lawrence forests depend neither solely on extreme fire weather and burn irrespective of fuel conditions ("the weather hypothesis") nor are solely a function of how fuel type variation influences spread or severity ("the fuels hypothesis") (Cumming 2001); rather, both types of variables are integral to understanding how large fire activity varies in time across our heterogeneous region.

Relative role of independent variables

Only fire weather/climate variables influenced whether fires occurred in a given ecodistrict-year, whereas these variables as well as landscape attributes explained proportion of area burned. Our data suggest precipitation deficit/surplus was the most important independent variable of our set in explaining fire occurrence and proportion burned (Table 3.3; Fig. 3.4). Precipitation deficit/surplus integrates variation in fuel moisture and seasonal water paucity. When precipitation exceeds potential evapotranspiration (PET), precipitation deficit/surplus indicates the amount of moisture available for storage in forest fuels and the seasonal conditions necessary to maintain an attenuated fire threat. When precipitation is lower than PET, actual evapotranspiration is equal to precipitation and precipitation deficit/surplus reflects seasonal water paucity as well as the variation in fire hazard attributable to forest fuel dryness. Precipitation deficit/surplus may also reflect other aspects of fuel conditions. For example, precipitation deficit/surplus is a primary driver of the accumulation of needles and other fine fuels on the floor of European forests (Kurz-Besson et al. 2006) and high evapotranspiration has been associated with a decreased aspen cover (Sexton et al. 2006). Moreover, since actual and potential evapotranspiration are indicators of environmental energy available for tree growth and for sustaining high tree species diversity (Currie & Paquin 1987), precipitation deficit/surplus may also reflect regional variation in the relative abundance of deciduous versus coniferous species related to evapotranspiration.

Growing season length (GSL) was inversely related to both fire occurrence and area burned (Table 3.3). As with precipitation deficit/surplus, GSL integrates climatic and fuel conditions. Relative to landscapes where the growing season is shorter, a longer growing season allows the earlier development of the deciduous component of stands and landscapes (Richardson et al. 2006) and a concurrent decrease in fire hazard. Many southern deciduous tree species have their northern limit within the Great Lakes-St. Lawrence forest region, including American beech (*Fagus grandifolia* Ehrh.), green ash (*Fraxinus pennsylvanica* Marsh.) and American basswood (*Tilia americana* L.) (Burns & Honkala 1990). Longer growing seasons favour the growth of these species and allow them to form the secondary component of the species mix across stands at the competitive exclusion of white spruce, balsam fir and conifers prevalent in more mixed forests to the north (Nichols 1935; Rowe 1972).

Drought code and antecedent winter precipitation influenced only fire occurrence. Our finding that yearly maximum drought code values are positively related to fire occurrence corroborates work in the southern Canadian boreal forest demonstrating the relationship between the condition of deep forest fuels and fire frequency (Girardin et al. 2004). A positive relationship between winter precipitation and fire occurrence was unanticipated (Table 3.3). We had expected high pre-fire season winter precipitation would more fully charge the landscape with moisture and thereby decrease fire occurrence, as has been shown across a variety of forest types in the western United States (Westerling et al. 2002). In the Great Lakes-St. Lawrence forests, above-average precipitation during the antecedent winter may mean large snow loads or ice accumulation increase branch ablation and tree fall of winter damage-susceptible species such as sugar maple or yellow birch (Croxton 1939), thereby increasing dead vegetation during the following fire season. Alternatively, high winter precipitation may cause higher-than-average moisture retention in mineral soil layers, thereby increasing production in the following fire season of the fine fuels necessary to support large fires, as has been shown in some fuel-limited western forests (Kipfmueller & Swetnam 2000; Westerling et al. 2002).

Other variables – large water bodies, aspen stands, glaciofluvial deposits and human population density – for which our data provided evidence for an effect on area burned are well documented as influencing fire size and spread. For example, several authors have documented large lakes and other water bodies are effective firebreaks and have a negative effect on area burned (Heinselman 1973; Grimm 1984; Zhang et al. 1999). Aspen stands are natural firebreaks, either by circumventing fires into adjacent conifer stands or causing crown fires to extinguish or drop to the surface (Howard 1996), a phenomenon reflected in their lower fire frequency relative to nearby conifer stands (Cumming 2001). Coarse deposits such as glacial outwash or ice-contact tills typically engender soils with drainage and texture qualities that have low total and seasonal availability of moisture (Hack & Goodlett 1960); in the Great Lakes-St. Lawrence forest region, these deposits are associated with high fire activity and conditions favourable for the growth of pines and other fire-adapted species (Whitney 1986; Drever et al. 2006). Our finding that population density negatively affected percent area burned is incongruent with the pattern observed in the deciduous-dominated forests of northern New England, USA, where percent area burned between 1909 and 1959 either remained roughly constant or increased with population density (Fahey & Reiners 1981). In our study region, observed decreases in area burned with increasing population density may be related to the presence of agricultural and firebreaks as well as to better access and more active fire suppression effort.

Our fuel map captured only spatial variation of forest fuels and represented the end state of changes occurring over the period of study, thereby precluding analyses of potential feedbacks between fuels and fire that can drive fire regime in decadal time scales (Ryan 2002). This lack of temporal dynamism may be behind the unexpected result that percent pine cover of each ecodistrict (PinePerc) showed a negative relationship with PAAB (Table 3.3). However, our finding may also reflect the decrease in abundance of white and red pine communities that has been documented throughout their range (Zhang et al. 1999; Radeloff et al. 1999; Thompson et al. 2006). Thus, many pine-dominated stands may have burned but failed to regenerate to pine dominance. Moreover, given the low amount of each landscape burned per year and that our study focused only on a 41-year period, temporal variation in fuel types within ecodistricts was arguably less important than the relative coverage of different fuel types in explaining fire occurrence and area burned.

3.7 CONCLUSION

Characterizing the relative roles of climatic and landscape variables may help design strategies for climate change adaptation. Studies of future climate change indicate a potential increase in fire weather severity, increased area burned, and a longer fire season across Canada (Flannigan et al. 2005), although important regional variation is expected (Flannigan et al. 1998). Therefore, in this region, increasing ecological resilience – the capacity of an ecosystem to absorb disturbance and undergo change while still maintaining its essential functions, structures, identity and feedbacks (Holling 1973; Peterson et al. 1998) – to climate change-induced alterations in fire regime could involve modifying the landscape through various silvicultural practices that influence the abundance and distribution of deciduous fuel types. Indeed, such 'fire smart' strategies already inform planning of fire risk abatement in different jurisdictions, principally through the management of aspen stands in time and space (Hirsch et al. 2001; Le Goff et al. 2005). Prioritization of landscapes in which to implement such strategies could be based on prevalence of the various enduring landscape attributes identified here as being important influences on area burned e.g. landscapes with few large water bodies and high percent cover of glaciofluvial deposits. Managing for increased resilience to climate change-related increases in fire could entail increasing the abundance or modifying the distribution of deciduous stands in these landscapes. This approach would mean that, compared to the present, substantially longer or more severe fire seasons and increased fire activity would be necessary to transform landscape structure and composition into alternate configurations.

3.8 ACKNOWLEDGEMENTS

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3.9 TABLES

Table 3.1 Description of independent variables for each ecodistrict.

Variable name	Description	Reference(s)
Human		
PopDen	Population density (people per km^2). We estimated temporal variability by adjusting 1991 estimates according to trends in rural population growth in the provinces where ecodistricts occur.	Population database of the National Ecological Framework for Canada (Marshall et al. 1999); (Statistics Canada 2006)
RoadDen	Density (km per km^2 of land area) of all primary, secondary and tertiary roads, as derived from 2001 Road Network File.	(Statistics Canada 2001)
Biophysical		
AspenPerc	Percent of ecodistrict covered by Aspen fuel type.	(Nadeau et al. 2005)
BorealMixPerc	Percent of ecodistrict covered by Boreal Mixedwood fuel type.	(Nadeau et al. 2005)
BorealSprucePerc	Percent of ecodistrict covered by Boreal Spruce fuel type.	(Nadeau et al. 2005)
GlacioFlvComplexPerc	Percent of ecodistrict covered by Glaciofluvial Complexes, i.e. sand, gravel and locally diamicton deposited by glacial meltwaters as undifferentiated ice contact stratified drift and	Surficial geology database of the National Ecological Framework for Canada (Marshall et al. 1999)

	outwash.	
GlacioFlvPlainPerc	Percent of ecodistrict covered by Glaciofluvial Plains, i.e. sand and gravel deposited by glacial meltwaters as outwash sheets, valley trains, and terrace deposits.	Surficial geology database of the National Ecological Framework for Canada (Marshall et al. 1999)
GrassPerc	Percent of ecodistrict covered by Grass fuel type.	(Nadeau et al. 2005)
NonburnablePerc	Percent of ecodistrict covered by nonburnable surfaces, including urban developments, agricultural fields, and water.	(Nadeau et al. 2005)
PinePerc	Percent of ecodistrict covered by immature, mature as well as red and white pine stands.	(Nadeau et al. 2005)
RidgedPerc	Percent of ecodistrict covered by long, narrow elevations of the surface, usually sharp crested with steep sides; ridges may be parallel, subparallel, or intersecting.	Local surface form database of the National Ecological Framework for Canada (Marshall et al. 1999)
UndividedPerc	Percent of ecodistrict covered by areas composed of > 75% rock outcrops.	Surficial geology database of the National Ecological Framework for Canada (Marshall et al. 1999)

WaterPerc	Percent of ecodistrict covered by large lakes and other water bodies (> approximately 1 km^2).	Land and Water Areas database of the National Ecological Framework for Canada (Marshall et al. 1999)
WetlandPerc	Percent of ecodistrict covered by swamps, bogs or fens.	Local surface form database of the National Ecological Framework for Canada (Marshall et al. 1999)
Climate and fire weather		
DC_max	Maximum value for Drought Code (unitless) during fire season.	Derived from station data (see Methods) and calculated as per (Turner 1972)
GSL	Growing season length (days), as determined by the number of days between the first and last day of the year when mean daily air temperature ≥ 5°C.	Canadian Ecodistrict Climate Normals 1961-1990 database of the National Ecological Framework for Canada (Marshall et al. 1999)
PE_FS	Potential evapotranspiration (mm) for the fire season (April to October), as calculated using the	Canadian Ecodistrict Climate Normals 1961-1990 database

	Penman method for a standard crop i.e. a continuous cover of short green plants exhibiting growth unlimited by nutrients, that shades the ground completely and has a negligible effect on evaporative water loss (1983); (Marshall et al. 1999).	of the National Ecological Framework for Canada (Marshall et al. 1999)
PptSurpDef	Difference between total precipitation between April to October (as determined from meteorological stations) and PE_FS (mm).	Total precipitation derived from station data (see Methods); PE_FS derived from Canadian Ecodistrict Climate Normals 1961-1990 database of the National Ecological Framework for Canada (Marshall et al. 1999)
StrikeDen	Density of cloud-to-ground lightning (number of strikes per km^2 during April to October) averaged from seven years (1999-2005) of occurrence data gathered from a network of ground-based sensors by the Meteorological Service of	Pers. comm., William Burrows, Environment Canada

	Canada. Detection efficiency > 90%, with a location accuracy of 0.5 km.	
TotalP_FS	Total seasonal precipitation (mm) during April to October, as determined from 1961-1990 climate normals derived by interpolation from all available meteorological stations.	Canadian Ecodistrict Climate Normals 1961-1990 database of the National Ecological Framework for Canada (Marshall et al. 1999)
WinterPpt	Total precipitation (mm) during December, January, and February before each fire season.	Derived from station data (see Methods).

Table 3.2 Results of model selection. Neg2LL indicates the negative 2 log likelihood; K, number of parameters; AIC_c, Akaike Information criterion corrected for small sample sizes; ΔAIC_c, AIC_c difference between a given model and the one for which the strength of evidence is highest; *w*, Akaike weight, the estimated probability a particular model of the set provides the most parsimonious data fit.

Model	Hypothesis	Independent Variables	Neg2LL	*n*	K	AIC_c	ΔAIC_c	*w*
1	*Re-charge*	WinterPpt, PptSurpDef, GlacioFlvComplexPerc, GlacioFlvPlainPerc, UndividedPerc	-922.91	904	16	-890.296	0	0.755
5	*Fire weather*	DC_max	-904.03	904	8	-887.869	2.427	0.224
6	*Amenable climate*	GSL, TotalP_FS, StrikeDen, PE_FS	-910.75	904	14	-882.275	8.022	0.0137
2	*Fire break*	AspenPerc, NonburnablePerc, WaterPerc, RidgedPerc WetlandPerc	-912.94	904	16	-880.326	9.970	0.005
3	*Human influence and access*	RoadDen, PopDen	-896.79	904	10	-876.547	13.749	0.001
4	*Fuels*	PinePerc, BorealSprucePerc, AspenPerc, GrassPerc, BorealMixPerc, NonburnablePerc	-912.53	904	18	-875.754	14.542	0.001

Table 3.3 Parameter estimates and standard error (SE) as averaged over all the models. Variables are described in Table 3.1. A t-value > |1.96| (in **bold**) indicates a parameter has 95% CI that do not include zero. 'Occurrence' indicates the parameter estimate for the logistic model component while 'Area burned' indicates its estimate for the lognormal component. Random effects are standard deviation estimates related to ecodistricts.

	Occurrence			Area burned		
Independent Variable	Estimate	SE	t	Estimate	SE	t
Human						
PopDen	-0.022	0.088	-0.25	-0.216	0.082	**-2.65**
RoadDen	-0.199	0.566	-0.35	0.295	0.478	0.62
Biophysical						
AspenPerc	-0.023	0.013	-1.78	-0.037	0.011	**-3.36**
BorealMixPerc	-0.013	0.032	-0.42	-0.053	0.033	-1.61
BorealSprucePerc	-0.019	0.029	-0.63	-0.047	0.029	-1.59
GlacioFlvComPerc	0.017	0.040	0.42	0.071	0.038	1.85
GlacioFlvPlaPerc	0.022	0.024	0.93	0.043	0.022	**1.97**
GrassPerc	0.009	0.046	0.20	-0.056	0.045	-1.24
NonburnablePerc	0.006	0.016	0.35	0.023	0.014	1.63
PinePerc	-0.027	0.031	-0.87	-0.054	0.027	**-2.01**
RidgedPerc	0.031	0.037	0.84	-0.044	0.030	-1.49
UndividedPerc	-0.009	0.017	-0.51	0.037	0.020	1.87
WaterPerc	-0.022	0.013	-1.63	-0.032	0.013	**-2.39**
WetlandPerc	0.000	0.009	0.02	-0.002	0.007	-0.30
Climate and fire weather						
DC_Max	0.003	0.001	**3.38**	0.002	0.001	1.66
GSL	-0.082	0.029	**-2.77**	-0.084	0.032	**-2.61**
PE_FS	0.009	0.006	1.48	0.010	0.007	1.54
PptSurpDef	-0.003	0.001	**-3.62**	-0.002	0.001	**-2.65**
StrikeDen	0.458	0.823	0.56	-0.450	0.718	-0.63
TotalP_FS	0.001	0.003	0.39	-0.005	0.003	-1.47
WinterPpt	0.003	0.001	**1.98**	0.001	0.002	0.79
Other						

Intercept ($a_{1,}$ a_2)	-2.065	0.431	**-4.79**	-7.470	0.443	**-16.85**
Residual (σ_2^2)				1.590	0.190	**8.36**
Random effect ($u_{1,}$ u_2)	0.386	0.187	**2.06**	0.294	0.174	1.68
Covariance (ρ)	0.288	0.152	**1.90**			

3.10 FIGURES

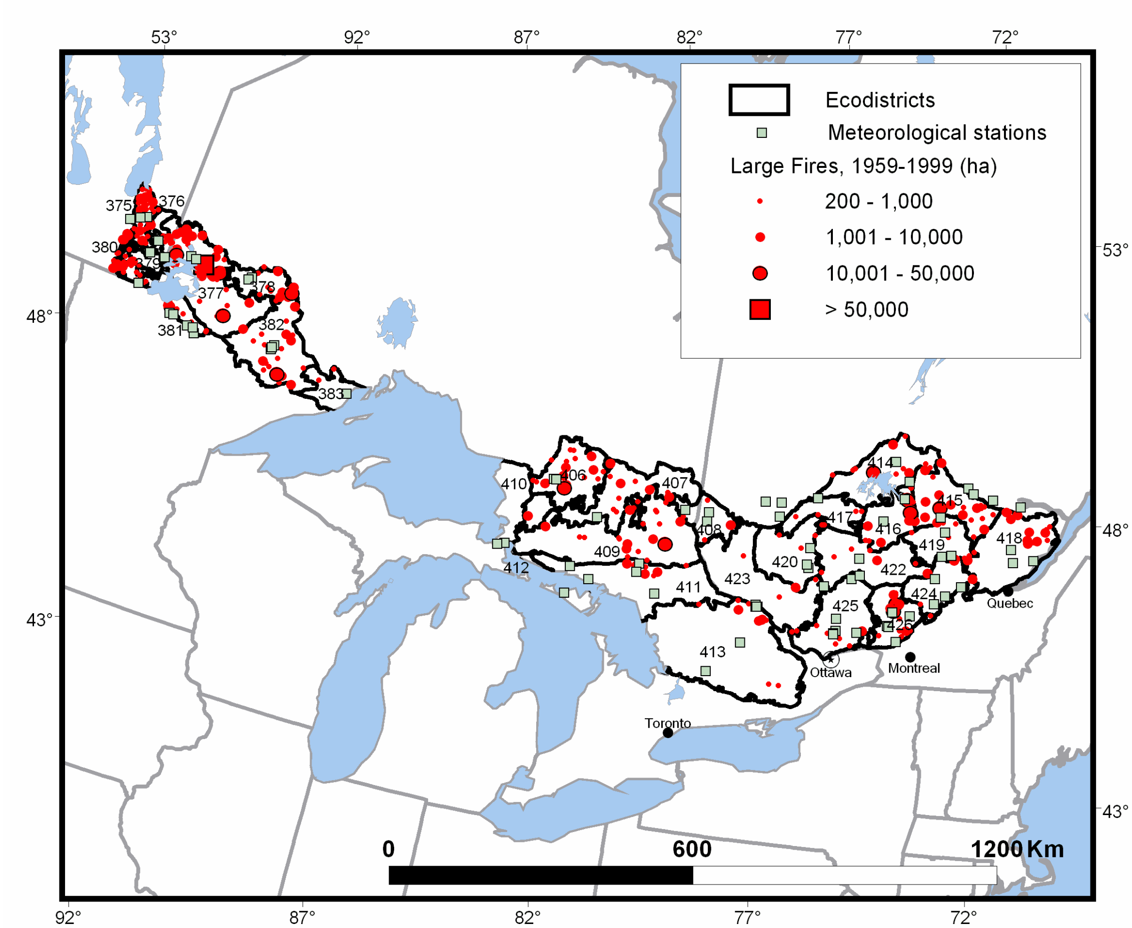

Figure 3.1 Study region, location of meteorological stations used, and distribution of large fires analyzed.

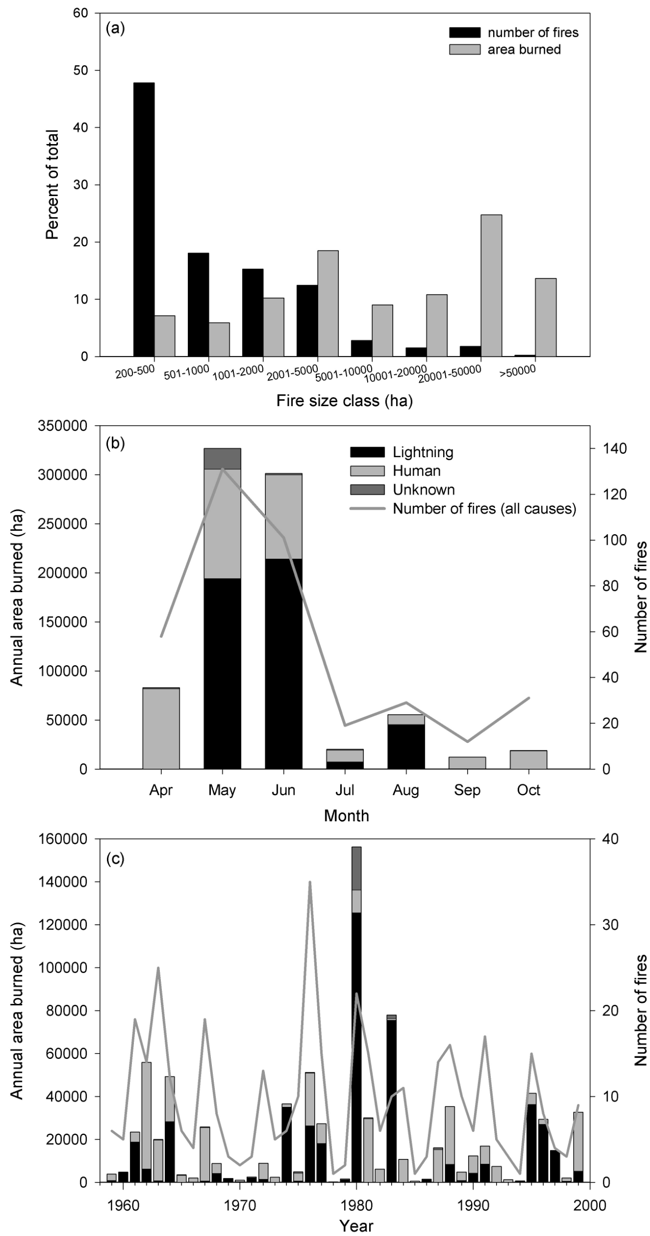

Figure 3.2 Annual area burned and fire occurrence of fires > 200 ha (1959-1999) by size class as a percent of total (a) and by month (b) and year (c). The legend in (b) applies for (c).

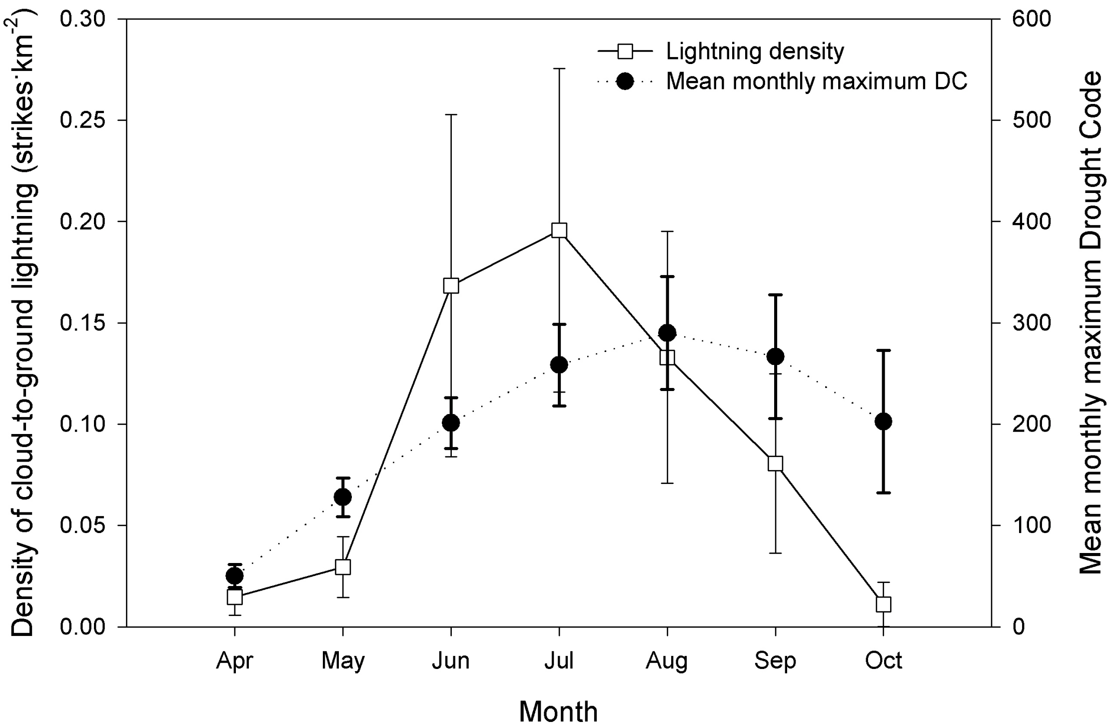

Figure 3.3 Mean monthly density (± SD) of cloud-to-ground lightning and mean monthly maximum Drought Code (± SD) in the Great Lakes-St. Lawrence forest region.

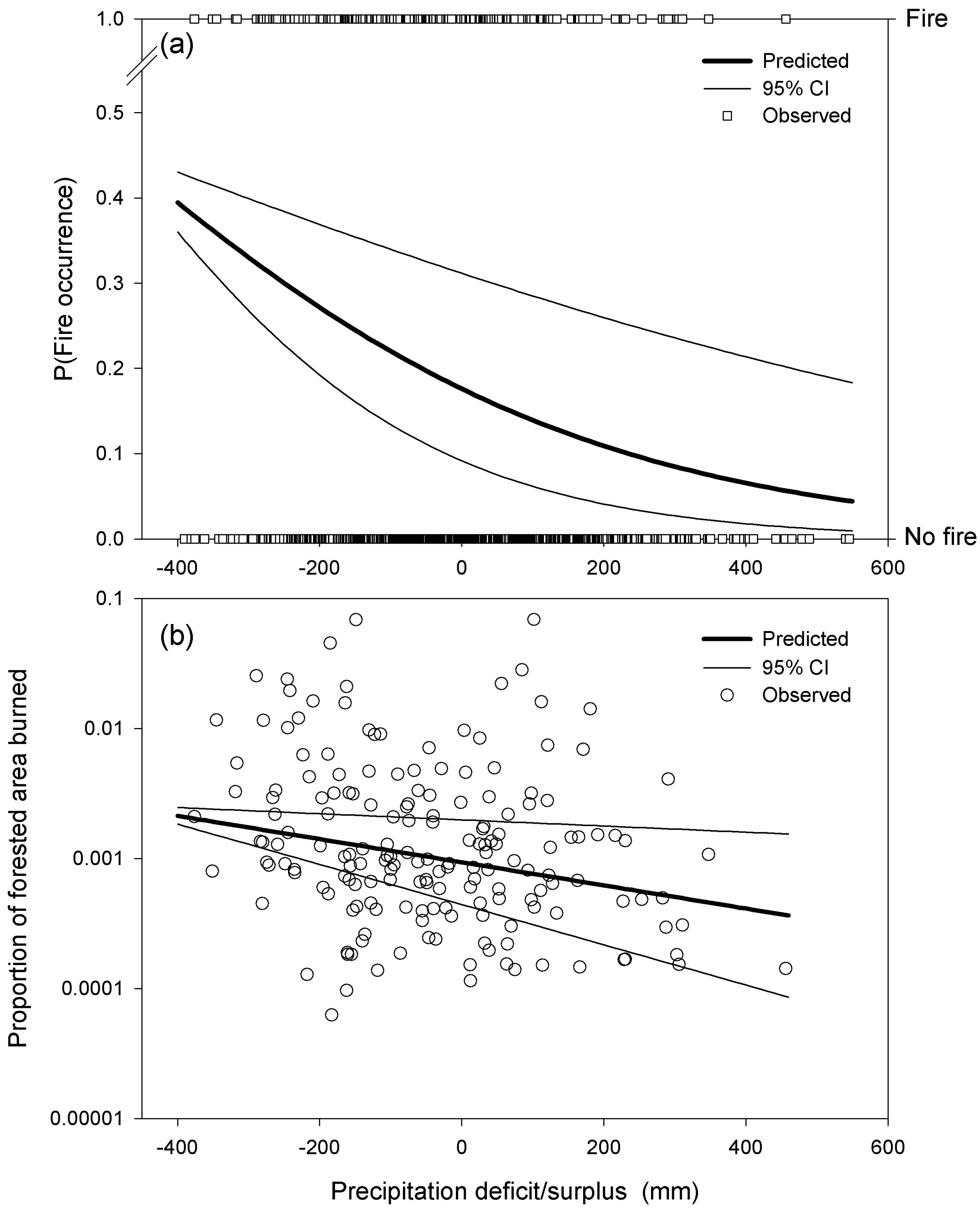

Figure 3.4 (a) Probability of fire occurrence and (b) proportion of forested area burned as a function of precipitation deficit/surplus during fire season (April-October).

CHAPITRE 4

EFFECTS OF CLIMATE CHANGE ON FIRE FOR A NORTHERN HARDWOOD LANDSCAPE IN TÉMISCAMINGUE, QUÉBEC

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4.1 ABSTRACT

We investigate how future climate change may affect fire in a deciduous forest landscape in Témiscamingue, Québec, Canada, in a three-part process. First, using an information-theoretic approach, we evaluate of the relative role of different weather variables in explaining large fire (> 200 ha) occurrence and area burned across the Great Lakes-St. Lawrence forest region between 1959 and 1999. Second, we examine how these weather variables varied historically in Témiscamingue and, third, how they may change from the present to 2100 according to three scenarios of climate change based on the third-generation Canadian Global Circulation Model. Mean monthly maximum temperatures during the fire season (April-October) and weighted sequences of dry spells best explained fire occurrence and area burned across the Great Lakes-St. Lawrence forest. Since 1910, mean monthly maximum temperatures have exhibited stationarity in Témiscamingue while dry spell sequences have decreased, indicating reduced frequency and length of dry spells. All three future climate scenarios show an increase in mean monthly maximum temperatures and a decrease in dry spell sequences during the 21st century, overall implying a slight increase in annual area burned. Despite this increase, effects of climate change on fire will not likely affect forest structure and composition as much as succession or harvesting and other disturbances.

4.2 RÉSUMÉ

Nous étudions comment le futur changement de climat peut affecter le feu dans un paysage de forêt feuillue dans Témiscamingue, Québec, Canada, dans un processus divisé en trois parties. D'abord, en utilisant une approche de la théorie de l'information, nous évaluons le rôle relatif des différentes variables climatiques en expliquant l'occurrence des grands feux (> 200 ha) et la superficie brûlée à travers la région des Grands-Lacs/Fleuve St-Laurent du Canada entre 1959 et 1999. En second lieu, nous examinons comment ces variables climatiques ont changé historiquement au Témiscamingue. Finalement, nous évaluons comment les feux peuvent changer

entre aujourd'hui et 2100 selon trois scénarios de changements climatiques basés sur le modèle global canadien de circulation. Les températures maximales mensuelles moyennes pendant la saison de feu (avril-octobre) et les périodes de sécheresse ont mieux expliqué l'occurrence de feu et la superficie brûlée à travers la région d'étude. Depuis 1910, la moyenne des températures maximales mensuelles n'a montré aucune tendance, tandis que les périodes de sécheresse ont diminué, indiquant une réduction de la fréquence et de la longueur des périodes de sécheresse. Chacun des trois futurs scénarios de climat montre une augmentation des températures maximales mensuelles moyennes et une diminution des périodes de sécheresse pendant le 21^{e} siècle, impliquant ainsi une légère augmentation de la superficie brûlée. En dépit de cette augmentation, les effets des changements climatiques sur le feu n'affecteront probablement pas la structure et la composition des forêts autant que la succession ou les perturbations, telle que la récolte.

4.3 INTRODUCTION

Forecasts of climate change for Canada indicate potential increases in the frequency of severe fire weather, area burned, ignitions and fire season length (Flannigan et al. 2005). This general pattern is expected to exhibit strong regional variation, with northern and western regions relatively more affected than southern and eastern regions (Flannigan et al. 2005). Although possible effects of climate change on fire have been little studied in the northern hardwood forests of eastern North America, there is some indication that fire frequency at the northern limit of the hardwood forests may actually decrease in the future, even with a tripling of atmospheric CO_2 concentrations (Bergeron et al. 2006). Such a decrease would be a continuation of the trend in fire frequency observed for this region during the last 400 years (Drever et al. 2006).

Our study objectives were: (i) to simultaneously test three different hypotheses that relate weather variables to the occurrence probability of fire and area burned in a biome, the Great Lakes-St. Lawrence forest region (Rowe 1972) - where fire is rare, (ii) characterize past trends in the weather variable(s) that best predicted fire occurrence and area burned for a northern hardwood landscape in Témiscamingue, Québec, Canada, and (iii) to evaluate how various scenarios of climate change will potentially affect fire occurrence and area burned in this landscape. Our three hypotheses were:

(a) Water balance – Fire occurrence and area burned are a function of precipitation surplus or deficit, the difference between total precipitation during the fire season and potential evapotranspiration. More and larger fires will occur with precipitation deficits than with surpluses (Drever et al., *submitted*).

(b) Temperature and dry spells – Fire occurrence and area burned are a function of mean monthly maximum temperature and weighted dry spell sequences. The higher the temperature and weighted dry spell sequences, the higher the number of fires and area burned (Flannigan and Harrington 1988).

(c) Fire weather – Fire occurrence and area burned vary as a function of the severity of fire weather, as estimated by the maximum value of the Canadian Drought Code (Turner 1972) during a given fire season. Higher Drought Code values are associated with more fires and increased area burned (Girardin et al. 2004).

4.4 METHODS

Study areas: Great Lakes-St. Lawrence forest region and Témiscamingue, Québec

The Great Lakes-St. Lawrence forest region of Canada (Fig. 4.1, modified from Rowe (1972)) extends from south-eastern Manitoba to the Saguenay valley north of the St. Lawrence River in Québec. Also termed the Mixedwood Shield (CEC 1997), this region has low relief with rolling and forested hills, many lakes, wetlands, outwash plains and other glacial features. Mean elevation is approximately 360 m, ranging from zero to 1 120 m above sea level. Total area of the study region is 40 596 481 ha; forests cover the vast majority (98%) of the land area.

The study landscape in Témiscamingue (~ 46º40’N; 78º45’W) occurs near the northern limit of Great Lakes-St. Lawrence forest region (Fig. 4.1). This forest is classified as the western Sugar maple-Yellow birch bioclimatic domain (Robitaille and Saucier 1998). Sugar maple (*Acer saccharum* Marsh.) and yellow birch (*Betula alleghaniensis* Britt.) dominate mesic, mid-slope sites. Other characteristic tree species of this forest are red pine (*Pinus resinosa* Ait.) and white pine (*P. strobus* L.), but eastern hemlock (*Tsuga canadensis* (L.) Carr.), white birch (*B. papyrifera* Marsh.), aspen (*Populus* spp.), white spruce (*Picea glauca* (Moench) Voss), and balsam fir (*Abies balsamea* (L.) Mill.) are also common.

Covering 179 300 ha, the Témiscamigue landscape is delineated on the west by the Ottawa River and its remaining boundaries correspond to 'ecological districts' i.e. relatively homogeneous units based on topography, elevation, surficial deposits, parent materials, and hydrology (Robitaille and Saucier 1998). The area has a subpolar, continental climate with cold winters and warm summers. Mean annual temperature is 4.4^{o}C, based on the nearest meteorological station (Barrage Témiscamingue (46°42'N, 79°6'W, 181 m above sea level)). Mean annual precipitation is 963 mm, with 23% (226 mm) falling as snow. Topography is characterized by gentle to moderately steep hills with rounded and often rocky summits. Surficial deposits are principally undifferentiated or rocky glacial tills, parent materials metamorphic or sedimentary, and soils principally humo-ferric podzols (Brown 1981). Mean elevation is 311 m above sea level and slopes average approximately 9%. Forests extend across almost all the land, while lakes and other large water bodies cover about 20% of the total area. While several large fires burned in Témiscamingue at the end of the 19th century (Drever et al. 2006), fires have been infrequent during the 20th century (Fig. 4.2). Industrial harvesting began in earnest about 1870 and today represents the dominant disturbance type, affecting on average about 2 500 ha per year (Fig. 4.2).

Fire variables

The observational unit for analyzing the relationship between fire occurrence, area burned and weather variables was ecodistrict-year, namely, a given year in a given ecodistrict. Ecodistricts are areas delineated by distinct assemblages of relief, geology, landforms, soils, vegetation, faunal use, water bodies and land use types (Ecological Stratification Working Group 1996).

Fire occurrence and annual area burned for each ecodistrict were determined from the Canadian Large Fire Database (Canadian Forest Service 2002). This 41-yr database documents all fires > 200 ha across Canada that occurred between 1959 and 1999, accounting for 97 % of the total area burned in Canada (Stocks et al. 2002). We analyzed all fires irrespective of cause. In our analyses, we modelled proportion of total forested area burned each year (PAAB) in each ecodistrict as the dependent variable to account for differences in ecodistrict size. Using only large fires reduced the confounding effects arising from small or localized fires that do not respond to environmental conditions (Lefort et al. 2004) as well as the influence of increasing efficiency in fire detection and suppression during the period of record.

The fire data were non-normal and exhibited a large clump of values at zero and skewed non-zero values. This data structure precipitated the use of a mixed-distribution, mixed-effects model for repeated measures data with clumping at zero and correlated random effects (Tooze et al. 2002; Martin et al. 2005). This approach involves a logistic regression using all ecodistrict-years to model the probability of fire occurrence ('fire occurrence model') with a lognormal regression to model the probability distribution of nonzero values using ecodistrict-years when PAAB > zero (the 'area burned' model). Mixed-effects models incorporate fixed effects of standard explanatory variables with random effects (correlation and non-constant variability) resulting from repeated, random unit observations (different fire seasons) on the same unit (ecodistrict). Both the occurrence and area burned model components incorporated ecodistrict as a random effect. The analysis involved the MIXCORR macro (Tooze et al. 2002) in SAS 8.2 (SAS Institute Inc. 2001). This method uses maximum likelihood methods for fitting the models, estimating independent variable effects on the occurrence probability and mean of nonzero values, and parameter estimation for both model components. We used a lognormal probability model to characterize the error structure of the area burned model. The two model forms were combined by calculating the overall likelihood for mixed-distribution model, which is a product of the likelihoods of each model component.

The model forms were:

[1] Occurrence: logit(Probability of Fire) = $a_1 + b_1 * X_1 + c_1 * X_2 + u_1$,

[2] Area burned: log(PAAB) = $a_2 + b_2 * X_1 + c_2 * X_2 + u_2 + \sigma_2^2$,

where a_1 and a_2 are intercept parameters; b_1 and b_2 are slope parameters for independent variable X_1; c_1 and c_2 are slope parameters for independent variable X_2; u_1 is the occurrence random unit effect; u_2 is the area burned random unit effect; and σ_2^2 is the variance of the residuals. The MIXCORR macro also determines a covariance parameter, ρ, of the random effects between the occurrence and area burned model components. Explanatory variables with slope estimates for which the 95% CI excluded zero were considered as exhibiting strong support from the data for an effect on our dependent variables.

Independent weather variables

We computed the weather variables by first compiling daily total precipitation and maximum temperature data (April to October, 1959-1999) from meteorological stations across the Great Lakes-St. Lawrence region (Fig. 4.1) from Environment Canada. These daily data were attributed to a particular ecodistrict by first selecting stations within 50 km of the ecodistrict centroid, as determined with ARCVIEW 3.2a (ESRI 1996), and then filling in missing data with the nearest stations . The temperature and precipitation data sets were screened and corrected for missing daily values. A missing daily temperature datum was replaced with the corresponding value from the closest nearby (< 50 km) meteorological station. In cases when such a station was not present or also missing data for the day in question, the mean monthly temperature for that month was used. We replaced missing precipitation data with the value for the same day from nearby stations or mean monthly values as described above.

Since we sought to forecast weather-fire relationships from GCM data, we limited our analyses to temperature- or precipitation-based variables, as other variables such as wind speed and relative humidity that influence fire risk are relatively less reliable (i.e. show lower correlations between observed and modelled values) or are not available as GCM output. We evaluated four weather variables: precipitation surplus/deficit (PptSurpDef); mean monthly maximum temperature (MonthlyMeanTx); the sum across the fire season of weighted dry spell sequences (WeightedDrySpell); and the maximum value of the Canadian Drought Code for each fire season. While these independent variables are interrelated, we ensured correlations among them were below levels high enough to cause collinearity problems within our models (Burnham and Anderson 2002). Precipitation surplus/deficit is the difference between total precipitation accumulated during a fire season (April to October) and potential evapotranspiration (PET). PET values (mm) for the fire season were obtained from the Canadian Ecodistrict 1961-1990 Climate Normals database of the National Ecological Framework for Canada (Marshall et al. 1999). To estimate precipitation surplus/deficit, we determined total fire season precipitation and subtracted ecodistrict PET values from this mean. Mean monthly maximum temperature is the average of the maximum daily temperature values for each month of the fire season. The sum of weighted dry spell sequences, SEQQ (Flannigan and Harrington 1988), for each fire season was calculated as follows:

$$SEQQ = \sum_{j=1}^{m} w_j$$

where

$$w_j = \sum_{j=1}^{n_j} i = n_j (n_j + 1)/2$$

where n_j is the number of days in the jth sequence in a fire season, $j = 1, 2 \ldots m$, and m is the total number of dry spell sequences over a given fire season.

The Canadian Drought Code (DC) estimates fire weather severity by providing a daily numerical index of the weather conditions conducive to fire involving deep organic layers of forest duff or heavy forest fuels (Turner 1972). It is calculated on a cumulative day-to-day basis and maintains a ledger of stored moisture by accounting for daily losses through evapotranspiration and gains from precipitation. See Turner (1972), Van Wagner (1987), and Girardin et al. (2004) for computational details. The DC is a significant predictor of fire frequency and area burned in the southern boreal forest of Canada (Girardin et al. 2004). Our analyses used yearly maximum DC, based on daily calculations, because occurrence and size of large fires are more intimately related to extremes of fire weather rather than to central tendency measures (Moritz 1997). Typically, DC estimates exhibit a seasonal trend – a slowly increasing to an apex in mid- to late August with a subsequent decline or roughly constancy to October (McAlpine 1990); we therefore used only years for which data were available for all of the fire season. Moreover, since the DC is especially responsive to precipitation (i.e. the index calculation may be affected for several days by one rainfall event (Girardin et al. 2004)), we excluded from the analysis any years with more than five days of missing precipitation data.

Model selection

We evaluated our three hypotheses that relate fire and weather variables using an information-theoretic framework (Burnham and Anderson 2002). This approach assessed which hypothesis (model) best fit the data and allowed determining which independent variables were responsible for this fit. We calculated Akaike's Information Criterion modified for small samples (AICc) for each model, the difference in AICc between each model and the model with the minimum AIC (ΔAICc), and the Akaike weight (w) (Burnham and Anderson 2002). Akaike weights estimated the probability that each model is the best of the model suite, and thus provided the relative level of data support for each model. Akaike weights also provided the basis for computing the weighted averages of parameter estimates. Model averaging was used to account for model selection uncertainty; this involves presenting parameter estimates and their standard errors as weighted averages, using Akaike weights as weighting factors normalized to 1 for the subset of models where that parameter appeared (Burnham and Anderson 2002).

Trend characterization

To characterize the historical trends of weather variables related to fire in our study landscape, we compiled daily maximum and total precipitation data from the Barrage Témiscamingue meteorological station. The variables with most explanatory capacity for fire occurrence and area burned, as determined from the model selection analysis above, were derived from these data. We modeled trends using PROC LOESS, a local regression method, in SAS 8.2 (SAS Institute Inc. 2001). The smoothing parameter (the data fraction used in each local regression) was chosen automatically by minimizing a bias-corrected Akaike's Information Criterion (AIC_{c1}) that incorporates both local model complexity and goodness of fit (Cohen 1999). In addition, we used PROC REG in SAS 8.2 to evaluate overall trends.

SRES Scenarios

In its Special Report on Emissions Scenarios, the Intergovernmental Panel of Climate Change developed a set of standardized scenarios of future global change for researchers investigating climate change impacts, adaptation, and mitigation (Nakicenovic and Swart 2000). These scenarios are alternative potential futures of how global greenhouse gas emissions will evolve from 1990 to 2100, based on different assumptions regarding demographic trends, socio-economic development, and technological change. All scenarios are equally valid and should not be considered forecasts or predictions. We analyzed the following scenarios:

- **A1B**: This scenario describes a future of very rapid economic growth, a global population peak in mid-century with a subsequent decline, and the rapid introduction of new and more efficient technologies. The technological emphasis is a balance between fossil fuel-intensive and non-fossil fuel energy sources, resulting in a mid-century peak of global emissions of carbon dioxide at roughly double 1990 levels with a subsequent decline to ~1.5 times 1990 levels by 2100.

● **A2**: This scenario describes a heterogeneous world where self-reliance and preservation of local identities shape global development patterns. Trends in fertility across regions converge very slowly and global population increases continuously to 2100. Economic development is primarily regional and per capita economic growth and technological changes are more fragmented and slower than in A1B. Global emissions of carbon dioxide increase continuously to 2100, reaching more than triple 1990 levels.

● **B1**: This scenario describes a convergent world with the same global population peak in mid-century as A1, but with rapid changes in economic structures toward a service and information economy, reductions in material intensity and the introduction of clean and resource-efficient technologies. Global emissions of carbon dioxide peak in 2050 at ~1.5 times 1990 levels and thereafter decline to about half of 1990 levels by 2100.

Global circulation model and output data

The above scenarios were evaluated in an atmospheric global circulation model (GCM) which provided the forecasted temperature and precipitation data. The GCM, entitled the Third Generation Coupled Global Climate Model (CGCM3, version T47), is a three-dimensional, coupled atmosphere-ocean simulation model that comprehensively represents the circulation among the atmosphere, oceans, and land surface. CGCM3 is a spectral model with a surface grid of cells with a spatial resolution of approximately 3.75° latitude/longitude and 31 vertical atmospheric layers. Its ocean grid shares the same land mask as the atmosphere, but has four ocean grid cells underlying every atmospheric grid cell. Resolution is 1.85° of latitude and longitude, with 29 vertical levels. Model output data were obtained from the Canadian Centre for Climate Modelling and Analysis.

To characterize future regional climate of Témiscamingue, we established a reference area centred on the study landscape that intersected three GCM grid cells. This procedure was performed because daily GCM output can occasionally exhibit small irregularities in predicted values for a given grid cell; incorporating cells neighbouring the cell of interest reduces the associated error. We calculated area-weighted averages of simulated temperature and precipitation based on the proportions of each cell within the reference area. GCM precipitation data showed biases in both intensity and frequency of rainfall when compared to observed historical data i.e. higher frequency of small precipitation events. We therefore used a downscaling technique, local intensity scaling (Schmidli et al. 2006), to correct the precipitation data for these biases. Trends were determined using the local regression procedure described above. In several cases, an inverse or natural log transformation was necessary to normalize the residuals; predicted data were back-transformed to ease interpretation.

Future fire in Témiscamingue

To forecast effects of climate change in Témiscamingue, we used a Monte Carlo approach based on the regressions that best explained fire occurrence and area burned. Each forecasted year, beginning in 2004, we determined the occurrence probability of fire based on regressions using the projected climate variables. In each iteration (n = 1000), we first generated a probability from a random number generator drawing from a uniform distribution between 0 and 1. If this random probability was less than the occurrence probability of fire given the forecasted climatic conditions for that year, fire was deemed to occur and the proportion of area burned determined according to the forecasted climatic conditions. Both the occurrence and area burned regressions incorporated error terms. Proportion burned per year was finally averaged across all iterations; these means were then multiplied by Témiscamingue's total area to forecast annual area burned. Trends were characterized with the local regression procedure outlined above. Computations were programmed in R v. 2.3.1 (R Development Core Team 2006) and SAS 8.2.

4.5 RESULTS

Model selection and weather variables

The temperature-dry spell hypothesis overwhelmingly received the highest level of relative support from the data, having a > 99% probability of providing the best data fit (Table 4.2). The other two models had Akaike weights < 0.001, and thus had relatively much lower support from the data. We found strong evidence of an effect on fire occurrence from all four independent variables (Table 4.3). Drought Code maxima (DC_max), mean of monthly temperature maxima (MonthlyMeanTx) and weighted dry spell sequences (WeightedDrySpell) showed a positive effect while precipitation surplus/deficit (PptSurpDef) had a negative effect on fire occurrence. Only weighted dry spell sequences and precipitation surplus/deficit showed strong evidence of an effect on proportion of area burned, with the former having a positive effect and the latter a negative effect (Table 4.3).

Trends in weather variables and future annual area burned

For the mean of historical monthly temperature maxima, local regression analysis indicated the most parsimonious fit was provided with a smoothing parameter of 1.0, analogous to a straight linear fit (AIC_{c1} = 134.85; Fig. 4.3a), although no overall trend was detected ($p = 0.955$). Forecasted temperatures for the A1B scenario (Fig. 4.3a) were best fit with a smoothing parameter of 1.0 (AIC_{c1} = 143.56) and showed a significant upwards trend overall ($df = 95$; $r^2 = 0.37$; $p < 0.0001$). The A2 scenario forecasted temperatures best fit with a smoothing parameter of 0.62 (AIC_{c1} = 115.40) with a significant upwards trend ($df = 95$; $r^2 = 0.64$; $p < 0.0001$). The temperature trend for the B1 scenario was best fit with a smoothing parameter of 1.0 (AIC_{c1} = 161.97) and a significant upwards trend ($df = 95$; $r^2 = 0.13$; $p = 0.0004$).

The seasonal sum of weighted dry spell sequences fluctuated over time, with peaks in 1922, 1947, and 1997 (AIC_{c1} = 813.98, smoothing parameter = 0.116; Fig. 4.3b). A general decrease was detected between 1910-2004 ($df = 72$; $r^2 = 0.08$; $p = 0.02$), indicating a decrease in the length and frequency of dry spells since 1910. In terms of future forecasts, all the scenarios had trends best fit with smoothing parameters of 1.0, except for CGCM3: B1 (AIC_{c1} = -198.26, smoothing parameter = 0.73). All scenarios showed a slight downwards, non-significant overall trend in the seasonal sum of dry spell sequences (Fig. 4.3b).

Based on the Monte Carlo simulations, the three scenarios forecasted an increasing trend in annual area burned (Fig. 4.4). All were best fit with a smoothing parameter of 1.0 (AIC_{c1} = -40.74, -82.58, -50.74 for scenarios A1B, A2, B1, respectively). Scenario A1B showed a weak, but significant increase in annual area burned (df = 95; $r^2 = 0.04$; $p = 0.0437$), as did scenario A2 (df = 95; $r^2 = 0.27$; $p < 0.0001$). The trend for scenario B1 was not significant ($p = 0.1291$).

4.6 DISCUSSION

Model selection and weather variables

A combination of mean monthly maximum temperature and seasonal sums of weighted sequences of dry spells best explained variation in fire occurrence and area burned across the Great Lakes-St. Lawrence region. Weighted dry spell sequences as well as precipitation surplus/deficit – both precipitation-based variables – explained both fire occurrence and area burned, while the mean monthly maximum temperature explained only fire occurrence. Several ecodistricts may have similar temperatures but quite different precipitation patterns during a given fire season; therefore, at the ecodistrict scale, temperature may not correlate as well as precipitation with fuel conditions that influence final fire size. Several authors have reported that prolonged sequences of dry hot days, particularly before and after emergence of broad leafed canopies, significantly predict fire occurrence and area burned (Flannigan and Harrington 1988; Nash and Johnson 1996; Carcaillet et al. 2001). The significant effect of Drought Code maxima on fire occurrence was unsurprising; Drought Code is a good predictor of fire frequency in the southern boreal forest of central Canada (Girardin et al. 2004).

Our finding of a significant effect of precipitation surplus/deficit on fire occurrence and area burned was more surprising; seasonally or monthly cumulated estimates of rainfall typically do not predict fire frequency or area burned, as it is the timing and frequency of rainfall that are paramount in influencing the fuel conditions related to fire ignition and spread (Flannigan and Harrington 1988). However, by including potential evapotranspiration in its calculation, precipitation surplus/deficit may reflect both the weather conditions necessary for fire to occur (e.g. lightning activity is highest during dry periods when high pressure dominates (Nash and Johnson 1996)) as well as a landscape-specific moisture balance that indicates the amount of precipitation available for storage in forest fuels.

Trends in fire weather and area burned

In terms of historical trends, the decrease in dry spell sequences we observed corroborates reported reductions in drought frequency for Québec's southern boreal zone since the end of the Little Ice Age (Bergeron and Archambault 1993; Lefort et al. 2003) as well as increases in the amount and frequency of summer precipitation for between 1900-1998 (Zhang et al. 2000). Similar to our results, Zhang et al. (2000) did not detect a significant 20th century trend in maximum temperatures in the Témiscamingue area.

For forecasted trends, the GCM output indicated a general increase in mean monthly temperature maxima and a roughly downwards trend in weighted dry spell sequences (Fig. 4.3). Taken together, future trends in these weather variables pointed to an increase in annual area burned for Témiscamingue (Fig. 4.4). These findings corroborate other studies examining future fire risk and area burned in Canada in general and west central Québec in particular. In terms of fire risk, Flannigan et al. (2000; 2001) showed roughly a 20% increase in fire risk in west central Québec by 2060 relative to a reference period (1985-1994 or 1960-1980, respectively), as estimated using the mean seasonal severity rating or Fire Weather Index of the Canadian Fire Weather Index System (Van Wagner 1987). Similarly, McAlpine (1998) described an increase from low to moderate seasonal fire weather severity in the parts of Ontario adjacent to Témiscamingue under a $2XCO_2$ scenario relative to a 1980-89 reference period. Conversely, other studies indicated little to no change for west central Québec under a $2XCO_2$ scenario (Flannigan et al. 1998; Stocks et al. 1998). In terms of proportion burned, our finding corroborates Flannigan et al. (2005) who showed a 50% increase in area burned for the region by 2080-2100, a period roughly corresponding to the tripling of atmospheric CO_2 concentrations. Our forecast contradicts Bergeron et al. (2006) who suggested that climate change will result in a continuation of the historical lengthening of fire cycles in southern Témiscamingue, even at $3xCO_2$ concentrations. However, their analysis was based on regressions that relied on wind speed and relative humidity, rather than temperature or precipitation. In summary, it appears that future fire activity in Témiscamingue will depend on the extent that temperature increases overwhelm the effect of reduced dry spell sequences resulting from increased fire season precipitation. Moreover, future fire activity will depend on how forecasted increases in fire season length (Stocks et al. 1998; Flannigan et al. 2000) extend into the leafless periods during spring and fall, the time during which deciduous forests are most susceptible to burning (Drever et al., *submitted*; Lafon et al. 2005).

Because our forecasts of annual area burned rely on regressions based on measures of central tendency, we precluded exclusion of extreme fire events. Therefore, our forecasts of future fire activity should be evaluated as general potential trends, rather than precise representations of future area burned. Moreover, the period that forms the basis of our weather-fire regressions is characterized by low fire activity relative to historical rates (i.e. 19^{th} century and previously) (Weir et al. 2000; Bergeron et al. 2006). However, this fire paucity is perhaps not necessarily a shortcoming in terms of forecasting because it seems likely that fire suppression will continue into the foreseeable future.

Management implications

Our evidence indicates climate change will not dramatically increase fire activity in

Témiscamingue. Compared to succession or harvesting (Fig. 4.2), wind and other disturbances, climate-related alterations in fire will not have major effects on forest structure and composition. This finding implies a decrease in the resilience of pine-leading stands by a continuation of the historically low frequency of fire, a disturbance on which these stands depend for competitive advantage vis-à-vis shade tolerant hardwoods on well-drained mesic sites. This issue is especially pertinent for red pine, as the persistence of this species depends on a combination of infrequent, high-severity fire and moderately frequent surface fires of low to moderate intensity (Bergeron and Brisson 1990; Carey 1993). Without such disturbances, red pine, and to some extent white pine, will likely continue to experience the range reduction already observed throughout their range (Radeloff et al. 1999; Zhang et al. 1999; Thompson et al. 2006). Therefore, if maintaining the historical diversity of various stand types is a landscape-level objective, managers should implement and monitor silvicultural treatments that create adequate conditions for the regeneration of pine, even on mesic sites e.g. partial cuts that open up the canopy followed by prescribed surface burns to expose mineral soil and liberate nutrients sequestered in the humus (Van Wagner and Methven 1978).

4.7 ACKNOWLEDGEMENTS

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4.8 TABLES

Table 4.1 Model descriptions.

No.	Model name	Independent variables	Variable description
1	Water balance	PptSurpDef	Precipitation surplus/deficit
2	Temperature and dry spell	MonthlyMeanTx WeightedDrySpell	Mean maximum temperature for fire season months Sum of weighted dry spell sequences across fire season
3	Fire weather severity	DC_Max	Maximum value of Drought Code for fire season.

Table 4.2 Results of model selection. Neg2LL indicates the negative 2 log likelihood; K, number of parameters; AIC_c, Akaike Information criterion corrected for small sample sizes; ΔAIC_c, AIC_c difference between a given model and the one for which the strength of evidence is highest; *w*, Akaike weight, the probability a model provides the most parsimonious fit to the data; r, Pearson's correlation coefficient between observed and predicted values for the occurrence and area burned model components.

Model no.	Neg2LL	*n*	K	AIC_c	ΔAIC_c	*w*	r (Occur)	r (Area burned)
2	-921.11	907	10	-900.86	0	0.999	0.34	0.55
1	-902.88	907	8	-886.72	14.145	< 0.001	0.29	0.54
3	-900.68	907	8	-884.52	16.343	< 0.001	0.30	0.54

Table 4.3 Parameter estimates and standard error (SE) for independent variables as averaged over all the models. See Table 4.1 for variable descriptions. A t-value > |1.96| (in **bold**) indicates a parameter has 95% CI that do not include zero. 'Occurrence' indicates the parameter estimate for the logistic model component while 'Area burned' indicates its estimate for the lognormal component. Random effects are standard deviation estimates related to ecodistricts.

	Occurrence				Area burned			
Variable	Estimate	SE	n	t	Estimate	SE	n	t
Independent								
DC_Max	0.003	0.001	1	**3.369**	0.002	0.001	1	1.665
MonthlyMeanTx	0.174	0.072	1	**2.405**	0.003	0.064	1	0.041
PptSurpDef	-0.002	0.001	1	**-3.155**	-0.002	0.001	1	**-2.681**
WeightedDrySpell	0.001	0.001	1	**2.780**	0.001	0.000	1	**3.178**
Regression								
Covariance	0.159	0.156	3	1.019				
Intercept	-7.053	1.853	3	**-3.806**	-7.780	1.676	3	**-4.641**
Residual	1.584	0.186	3	**8.537**				
Random Effect	0.456	0.221	3	**2.065**	0.234	0.156	3	1.501

4.9 FIGURES

Figure 4.1 Great Lakes-St. Lawrence forest region, location of meteorological stations used, distribution of large fires analyzed, and location of the Témiscamingue landscape.

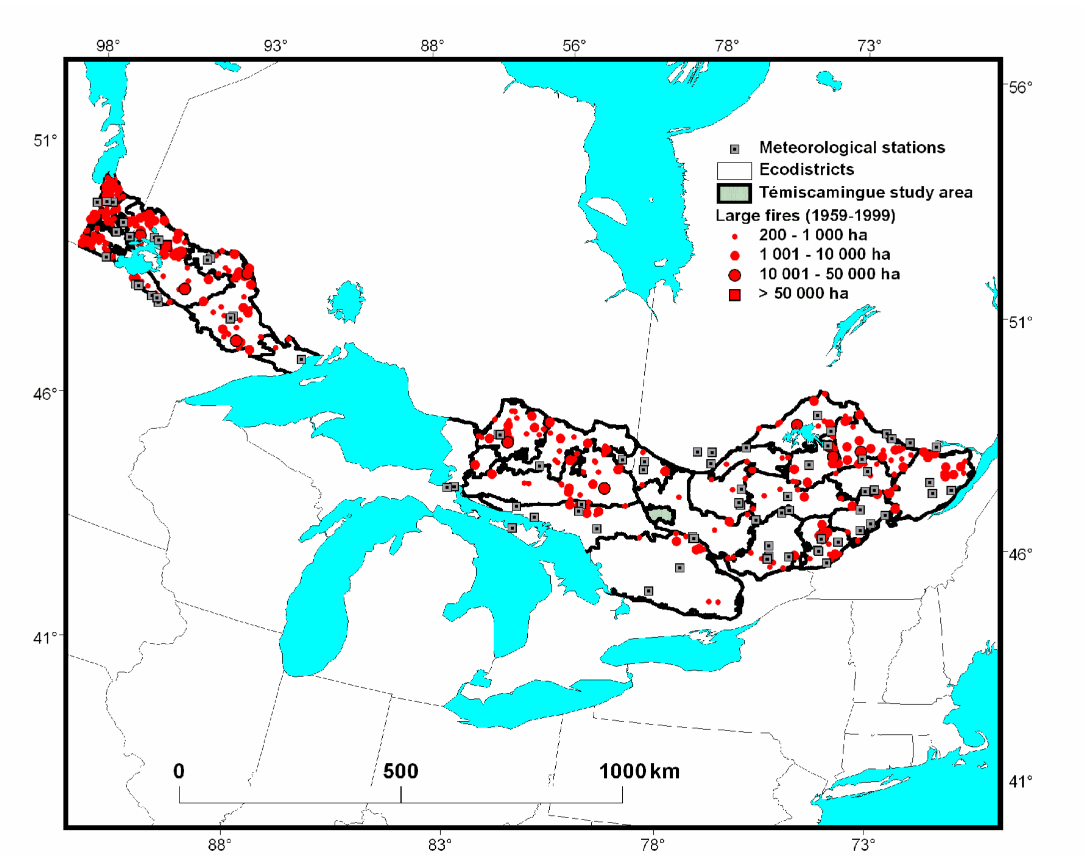


Figure 4.2 Recent history of fire and forest harvesting (inset) in Témiscamingue, Québec. Fire data are derived from Drever et al. 2006. Harvesting data (line across figure indicates the mean) are derived from the forest inventory database of the Ministère des Ressources naturelles du Québec.

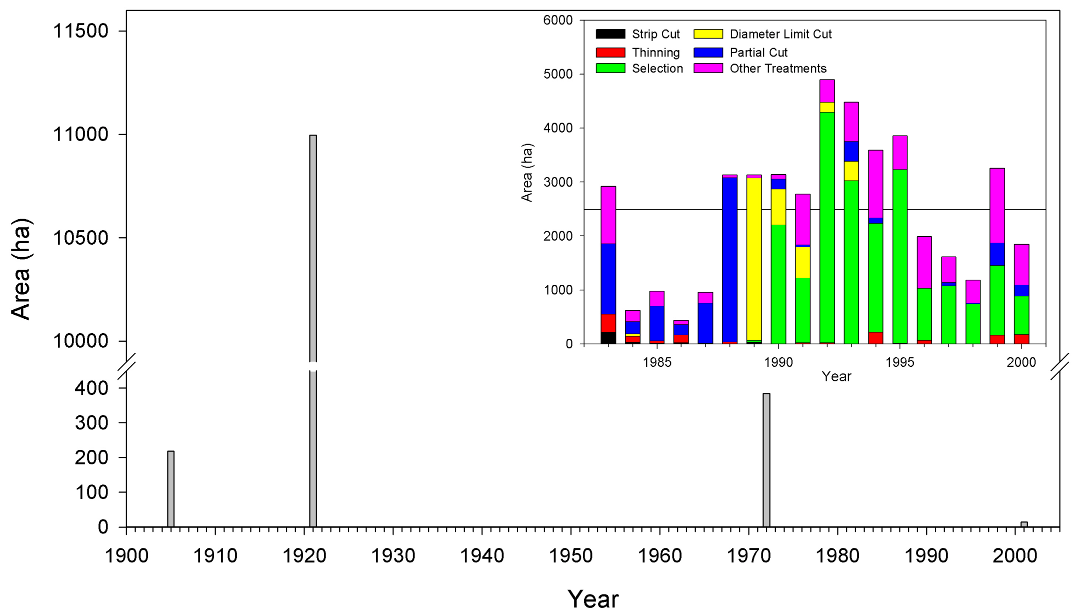


Figure 4.3 Past and forecasted trends in (a) mean of monthly temperature maxima and (b) seasonal sum of weighted dry spell sequences.

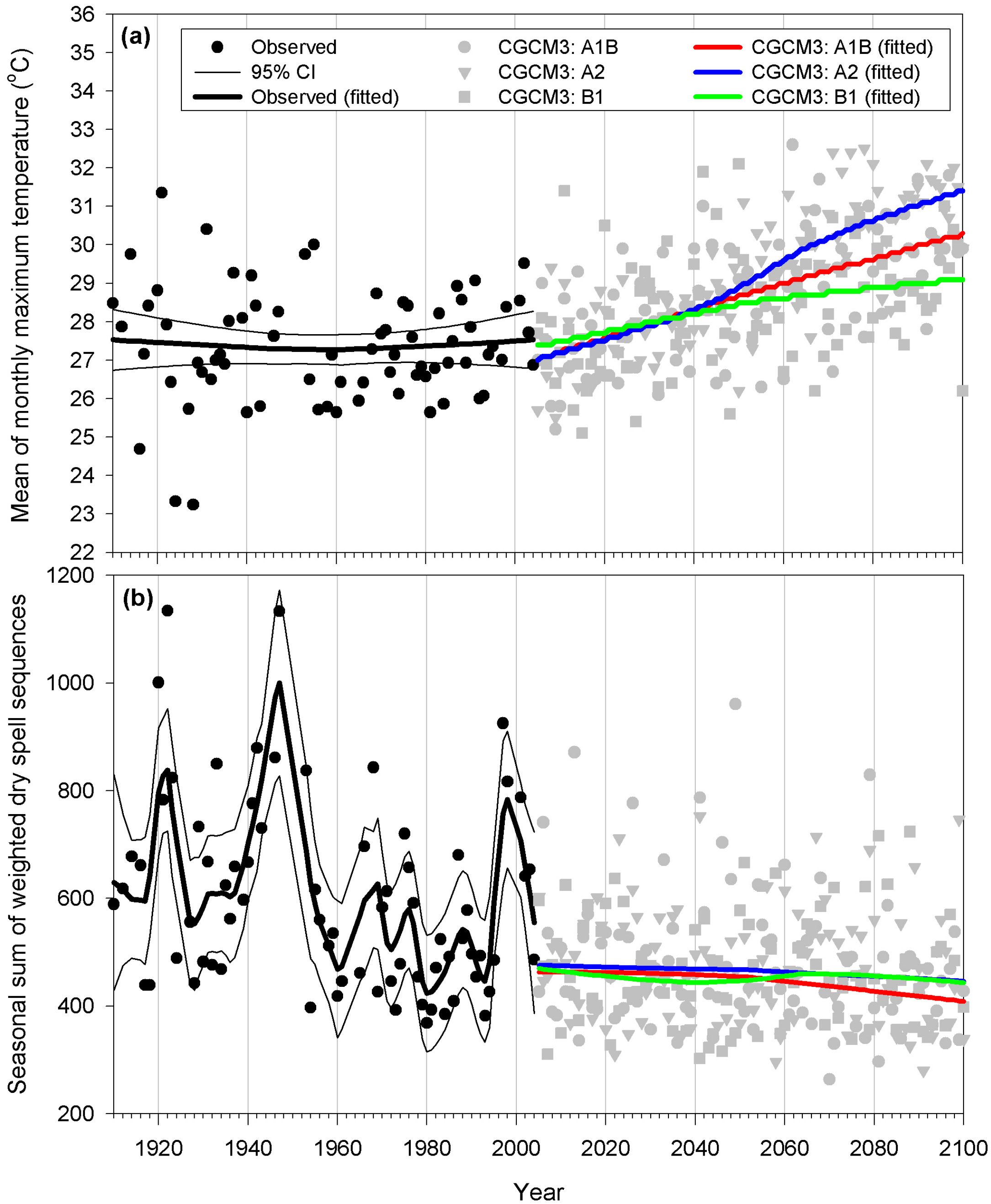
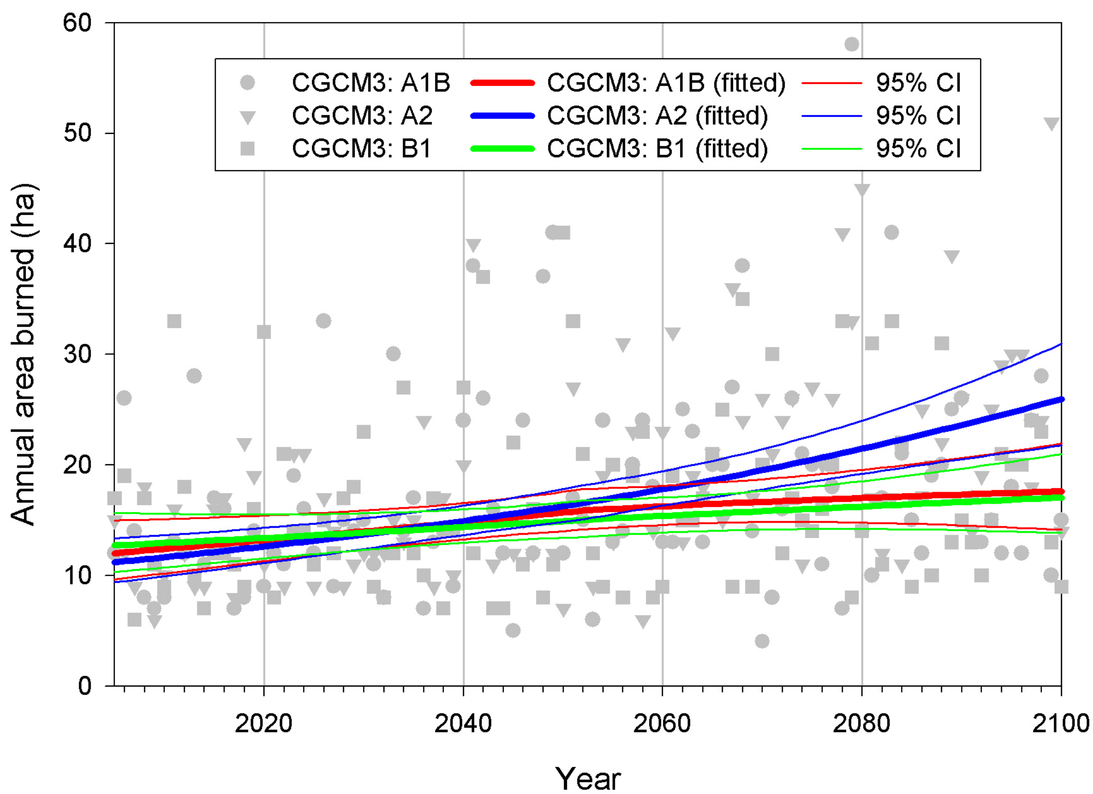


Figure 4.4. Forecasted annual area burned for Témiscamingue landscape based on three scenarios of climate change.



CONCLUSION GENERALE

CONTRIBUTIONS PRINCIPALES DE LA THESE

Cette conclusion résume les principales contributions de la thèse à la littérature sur la résilience écologique, la caractérisation des régimes de feux, les rapports entre le feu et le climat et les impacts des changements climatiques. La thèse :

• passe en revue de manière exhaustive la littérature concernant la résilience écologique et sa pertinence avec le paradigme sous-tendant la gestion durable des forêts: la sylviculture basée sur les perturbations naturelles. Cette revue synthétise les résultats d'une littérature typiquement limitée à l'écologie des communautés, à la biologie des populations et à l'écologie de paysage et place les éléments dans un contexte d'écologie forestière et de sylviculture. Cette synthèse sera utile aux chercheurs et aux aménagistes concernés par la gestion de systèmes complexes et adaptatifs dans lesquels la récolte de bois est devenue la perturbation dominante et qui sont de plus en plus soumis à des changements globaux tels que les changements climatiques, l'introduction d'espèces envahissantes ou l'altération des cycles biogéochimiques.

• caractérise la fréquence de feu et les relations entre la composition et le feu dans la forêt feuillue du Québec. À l'exception de la description détaillée des types de forêts trouvés au Témiscamingue de Brown (1981) et des travaux de l'*Institut Québécois d'Aménagement de la Forêt Feuillue* (Nolet et al. 1999 ; Nolet et al. 2001), je n'ai trouvé aucune littérature révisée par des pairs sur le sujet pour cette région. Une des contributions majeures de cette thèse est donc la documentation d'un processus écologique fondamental, le feu, et de son influence sur la composition en espèces de la canopée. Cette documentation est importante pour mettre en application une gestion basée sur les perturbations naturelles. Bien que les perturbations de petite taille, soit la dynamique de trouées, sont celles qui couvrent la majorité de l'aire, la composition en espèces dans ce paysage est le résultat de la combinaison des changements de succession et des perturbations qui remplacent des peuplements entiers telles que le feu ou le chablis total. La dominance de l'érable, de la pruche et d'autres espèces qui caractérise les peuplements développés en l'absence de perturbation est maintenue par leur tolérance à l'ombre. Or, la dominance des assemblages de tremble-bouleau-pin se produit quand le feu et d'autres perturbations sont fréquents et assez sévères pour créer les environnements ouverts qui permettent l'établissement et la croissance de ces espèces. La compréhension des processus par lesquels la récolte de bois peut maintenir les différentes espèces qui exigent la gamme des conditions créées par ce spectre de perturbations constitue le défi central des gestionnaires qui cherchent à conserver la diversité d'espèces d'arbres dans les peuplements sous aménagement.

• contient la seule analyse de feux couvrant tout le biome Grands Lacs-St. Laurent et les rôles causaux joués par le climat et certains attributs de paysage. Elle se fonde sur une nouvelle approche statistique pour analyser des données du feu : modélisation avec des modèles mixtes et effets mixtes. Cette approche répond à un problème fondamental fréquemment rencontré dans des études d'historique des feux, particulièrement dans les secteurs où le feu est rare : une abondance de valeurs zéro en combinaison avec des valeurs différentes de zéro qui ne sont pas normalement distribuées. La modélisation avec des modèles mixtes traite cette structure de données en combinant une analyse d'occurrence du feu avec celle de la superficie brûlée, procurant ainsi un pouvoir statistique élevé. La modélisation des effets mixtes permet également de mesurer l'influence des caractéristiques fixes de paysage et celles qui varient avec le temps, une approche qui permet de discuter de la résilience de la forêt dans un contexte de changements climatiques. En outre, j'ai utilisé une approche de la théorie de l'information pour évaluer le poids de diverses hypothèses concernant les facteurs causaux derrière des régimes récents du feu : aucune autre étude sur les feux n'a utilisé cette approche.

Les résultats de ce chapitre peuvent être employés pour formuler des stratégies pour s'adapter aux augmentations de la sévérité des conditions météorologiques propices au feu, par exemple, en fournissant une base pour classer les paysages selon des caractéristiques fixes telles que la proportion du territoire couvert par des dépôts fluvioglaciaires. Plus le nombre de caractéristiques fixes associées à l'activité élevée du feu est élevé, plus le paysage devrait recevoir une priorité élevée que dans l'attribution des mesures de mitigation et d'adaptation. D'ailleurs, les résultats peuvent être employés pour gérer la composition de sorte qu'une activité sensiblement accrue du feu soit nécessaire pour transformer des paysages en configurations alternatives.

• explore les effets possibles des changements climatiques sur le feu pour un paysage particulier dans le biome feuillu. La conclusion centrale de cette analyse suggère que d'aujourd'hui à l'an 2050, la rareté de grands feux qui a caractérisé l'histoire du feu dans ce paysage se maintiendra. Au delà de 2050, il est possible que les augmentations d'évènements de feux associés à la hausse de la température deviennent importantes. Jusque là, tous les effets sur la structure et la composition de la forêt résultants des changements climatiques seront éclipsés par des interventions humaines et peut-être par des perturbations causées par le vent et les insectes.

FUTURES DIRECTIONS DE RECHERCHE

Des futures directions possibles pour cette recherche incluent :

(a) l'incorporation d'échelles multiples dans l'analyse régionale du feu. Ceci est important pour comprendre le rôle de l'échelle dans la façon dont les variables de climat et de paysage influencent l'occurrence et la superficie brûlée. Il serait intéressant d'évaluer l'hypothèse que l'occurrence de feu et la superficie brûlée sont influencées par la même hiérarchie de variables à different échelles pour le biome forestier des Grands Lacs/St. Laurent. À l'échelle du paysage, le feu est influencé par les conditions météorologiques propices au feu et les variables compositionnelles (les espèces, les types de débris ligneux) mais à des échelles plus grandes (e.g. écorégion), il est influencé par des variables physiographiques (par exemple, la géologie surficielle) et de climat.

(b) obtention d'une meilleure précision de l'échelle de temps utilisée dans l'analyse du rôle causal du climat qui affecte l'occurrence et la superficie brûlée. Au lieu d'employer des données de feu et de météorologie agrégées à l'échelle annuelle, il serait intéressant de comprendre les interactions entre le feu et les conditions métérologiques au moment où se produit la déflagration, par exemple la longueur des périodes sèches qui ont précedé l'événement ou la présence de foudre.

(c) une exploration en détail du rôle du climat antécédent sur l'activité du feu. En d'autres termes, pour des prévisions améliorées de l'activité du feu, il serait utile de comprendre comment les conditions de climat et de débris ligneux de l'année précédente affectent l'occurrence du feu.

(d) employer les meilleures prévisions du climat disponibles. Le rendement du modèle global de circulation utilisé a une résolution large. L'obtention des données d'un modèle régional de climat (MRC) permettrait de mieux comprendre la façon dont des phénomènes plus locaux de temps et de climat, tels que les effets climatologiques influencés par les Grand Lacs, agissent au Témiscamingue.

(e) comprendre les implications des modifications du régime de feu sur la structure des classes d'âge et la composition du paysage forestier. Ceci pourrait comporter l'utilisation de modèles dynamiques de paysage, dans lesquels le module de feu dépendrait des extrants produit par le MRC, ce qui permettrait de simuler de façon stochastique les processus de contagion des feux. Il faudrait aussi intégrer dans le modèle les interventions humaines qui sont devenues le processus le plus imporant qui structure le paysage et le vent qui de tout temps a fortement influencé les forêts du Témiscamingue.

IMPLICATIONS POUR LA GESTION DES FORETS

Plusieurs des implications practiques de cette thèse pour les aménagistes forestiers sont décrites au chapitre 1. Cependant, plus globalement, on peut tirer deux idées pratiques importantes pour l'aménagiste forestier cherchant à gérer des forêts tout en maintenant leur résilience écologique face aux incertitudes provenant du changement global de climat et dans un contexte où la récolte de bois et d'autres effets anthropiques constituent les principaux agents du changement :

- La première est que les changements climatiques prévus pour le Témiscamingue, probablement, n'auront pas un effet très important sur le régime du feu. Il est fort probable que les changements climatiques auront un effet plus important aux niveaux des épidémies d'insectes, sur le chablis et l'envahissement d'espèce exotiques indésirables.
- La deuxième est que pour maintenir la résilience de nos écocystèmes, il est important d'aménager ceux-ci de façon à y maintenir le plus de complexité possible et cela à tous les niveaux. Il faudrait donc adopter une diversité de pratiques forestières au niveau du peuplement et du paysage forestier et modifier ces pratiques au fur et à mesure que de nouvelles informations deviennent disponbiles (gestion adapatative). La diversification des pratiques permet d'augmenter la résilience des forêts, ce qui répond globalement aux objectifs d'un aménagement durable des forêts.

Nous entrons dans une phase de changement climatique important et rapide et la compréhension de son influence sur la dynamique de nos forêts nous permettra de mieux s'adapter à ces nouvelles conditions. Il est clair que la rapididité de ces changements climatiques exigera de l'humain des interventions pour en minimiser les impacts. Cette recherche constitue donc une amorce d'un effort beaucoup plus large que le Canada devra entreprendre pour minimiser les impacts des changements climatiques sur nos forêts.

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