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INFLUENCE DE LA VÉGÉTATION ET DU CLIMAT DANS LE COMPORTEMENT DES INCENDIES EN FORÊT BORÉALE MIXTE CANADIENNE

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RÉSUMÉ

La mosaïque de la forêt boréale mixte canadienne est constituée de peuplements feuillus, mélangés, et résineux, dont la composition, la structure, et l'agencement spatial résultent principalement de l'impact des incendies naturels successifs. Or, peu d'informations existent sur les facteurs responsables du départ d'un feu, sur le comportement du feu pendant l'incendie, ni sur les facteurs influençant la mortalité post-incendie dans cet écosystème. Pourtant, la compréhension des incendies est primordiale dans l'optique d'une gestion durable écosystémique des forêts inspirée des perturbations naturelles. Cette thèse a pour objectifs de caractériser la végétation et le climat en tant que facteurs de l'environnement du feu, d'analyser leurs rôles dans le comportement du feu, et d'étudier la mortalité post-incendie des principales essences de la forêt boréale mixte.

L'hypothèse générale est que la composition variable des peuplements de forêt boréale mixte, en place avant l'incendie, produit divers comportements du feu, entraînant une mortalité différenciée à l'échelle des peuplements, et une structure du paysage en mosaïque dont les propriétés varient de celles d'un paysage monospécifique. L'étude prend place au Québec en Abitibi dans les régions de Duparquet et de Val Paradis. L'environnement du feu est réduit à la végétation et au climat, en fixant constante la topographie (peuplements sélectionnés uniquement sur argiles mésiques à faibles pentes).

L'étude des débris ligneux et leur évolution temporelle au cours de la succession (jusqu'à 230 ans après le dernier feu dans la région de Duparquet) est la première du genre en forêt boréale mixte au Canada et les résultats vont à l'encontre des principaux modèles d'accumulation de matière morte des forêts nord américaines. En forêt boréale mixte, les patrons temporels d'accumulation des débris ligneux au sol suivent une courbe sigmoïde et sont influencés par la composition de la canopée et le temps depuis le dernier feu, alors que les chicots suivent une courbe de croissance progressive et ne sont influencés que par le temps depuis le dernier feu. A l'opposé, la plupart des autres écosystèmes forestiers présentent, suite à un feu, des accumulations selon une courbe en "U". Cette différence majeure s'explique notamment en forêt boréale mixte par le remplacement des espèces dominantes au sein de la canopée, des productivités spécifiques différentielles, des taux de décomposition élevés, et l'occurrence de perturbations naturelles telles que les épidémies d'insectes représentant des pulses épisodiques d'apport de débris ligneux. Ces derniers étant reconnus comme des habitats privilégiés pour de nombreux organismes, l'aménagement durable écosystémique devra donc développer des méthodes permettant de conserver dans les peuplements exploités suffisamment de débris ligneux, au sol et dressés, dont le nombre et la composition reflèteront la structure de la mosaïque naturelle.

La susceptibilité au feu d'un peuplement ("stand fire hazard"), définie pour cette étude comme le départ potentiel du feu et agissant sur la sévérité post-incendie, est influencée par la composition et la structure du peuplement. Ce résultat est mis en évidence à Duparquet (Abitibi) par l'analyse des combustibles de surface simultanément en place dans chacun des 48 peuplements. Les différences significatives trouvées portent sur les combustibles fins (branches et arbustes) dont les charges sont plus importantes dans les

peuplements conifères que dans les feuillus. Si la succession est reconstruite en regroupant les peuplements étudiés en quatre types (feuillus, mixtes-feuillus, mixtes-conifères, et conifères), les résultats suggèrent que la susceptibilité au feu des peuplements croit au cours de la succession avec l'augmentation des résineux. L'impact de la variabilité des combustibles est testé par la comparaison de simulations du comportement du feu provenant du système BEHAVE. Ce simulateur a été choisi en raison de la nécessité d'incorporer les charges des différents combustibles de surface. Les simulations sont réalisées pour des feux de printemps et d'été avec une fenêtre météorologique couvrant des risques de feu faibles à extrêmes. Les résultats suggèrent qu'au départ du feu, les peuplements feuillus supportent une intensité dégagée plus faible et une propagation de la flamme plus lente que les peuplements mixtes ou conifères.

La végétation et le climat influencent significativement le comportement du feu en forêt boréale mixte. Ce résultat provient des simulations réalisées à l'aide des systèmes BEHAVE et FBP (Fire Behavior Prediction) sur les peuplements de Duparquet, et il alimente la polémique concernant le rôle des différents facteurs de l'environnement du feu. Les résultats des ANOVA suggèrent que le climat a un effet prédominant comparé à la végétation. Toutefois, il existe dans la mosaïque naturelle des peuplements très différents et non pris en compte dans cette étude. L'influence du facteur végétation sur le comportement du feu doit donc être sous-estimée dans les résultats présentés. Dans le cadre d'un aménagement durable écosystémique il serait pertinent de considérer à long terme l'effet du climat et de ses changements sur l'occurrence et le comportement des feux, ainsi que sur les changements au niveau de la végétation. De plus, si les effets des feux naturels servent de base au développement de pratiques sylvicoles, l'analyse du comportement du feu présentée suggère d'appliquer des coupes de tailles variables et des pratiques agissant sur les sols avec des sévérités différentes. En effet, les surfaces brûlées et les intensités dégagées augmentent proportionnellement avec la surface terrière des conifères.

Les comparaisons entre les comportements du feu observés lors de brûlages expérimentaux et ceux prédits par les systèmes BEHAVE et FBP révèlent que les prédictions quantitatives de BEHAVE sont inadaptées à la forêt boréale mixte. Ce résultat ne remet pas en question l'emploi de BEHAVE dans l'étude de la susceptibilité au feu puisque celui-ci est le seul système basé sur les charges des combustibles, et qualitativement les systèmes FBP et BEHAVE présentent la même classification des peuplements. A l'opposé, les valeurs prédites par le FBP sont proches des observations réelles. Ce résultat confirme la pertinence de l'emploi futur du FBP pour la forêt boréale mixte, et encourage l'amélioration des équations traitant du comportement du feu dans les peuplements mixtes.

La mortalité post-incendie, analysée à Val Paradis (Abitibi) suite au feu de printemps de 1997, confirme les résultats des chapitres précédents en montrant que la composition des peuplements influence l'intensité dégagée pendant le feu. Les peuplements feuillus brûlent en effet avec des intensités plus faibles que les peuplements mixtes ou conifères. L'analyse des taux de mortalité confirme les classifications passées présentant le peuplier faux-tremble comme l'espèce la moins résistante au feu comparée à l’épinette noire et au pin gris. Cette thèse suggère chez ces deux conifères l'existence d'interactions efficaces entre la résistance cambiale et celle du houppier *via* la hauteur de la base du houppier : plus celle-ci est basse, plus elle protège la tige et le cambium de l'exposition directe à la chaleur dégagée par le front de flammes. L'influence positive de cette variable dans la résistance au feu est formulée pour la première fois, et elle semble s'opposer à la faible résistance potentielle due aux écorces fines des conifères étudiés. Toutefois, au-delà d'une intensité maximale tous les arbres de toutes les espèces meurent peu après l'incendie. Le diamètre à hauteur de poitrine ou la hauteur totale des arbres sont les variables dendrométriques qui, associées à la hauteur de charbon sur le tronc, prédisent le mieux la probabilité de mortalité. Ces modèles logistiques servent à la création de nomogrammes spécifiques pouvant aider les aménageurs forestiers pour certaines pratiques sylvicoles comme la récupération de bois après incendie et les brûlages dirigés.

Les résultats présentés dans cette thèse confirment donc l'influence de la composition locale de la végétation sur le comportement du feu en forêt boréale mixte, et ils évaluent la mortalité post-incendie des principales essences. Les simulations et modèles proposés visent à mieux prédire la variabilité naturelle qui devrait être préservée par l'aménagement écosystémique au sein des peuplements et à l'échelle de la mosaïque forestière. L'effet des changements climatiques sur la forêt boréale mixte est finalement analysé en considérant les interactions entre le comportement du feu, le climat, et la végétation.

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CHAPITRE I

INTRODUCTION GÉNÉRALE

Le biome boréal est le plus vaste biome circumpolaire au monde et s'étend du 47 au 58 degré latitude nord en Amérique du Nord (Larsen 1980). Au Québec, la forêt boréale mixte correspond au domaine bioclimatique de la *sapinière à bouleau blanc* (Ordre des ingénieurs forestiers du Québec 1996). Elle représente un écosystème extrêmement important, tant par sa position géographique que par les intérêts économiques et culturels qu'elle suscite aujourd'hui. Alors que les exploitations forestières et minières ont dominé depuis le début du XX^{è} siècle, les activités tendent à se diversifier pour acquérir un caractère récréatif et social. La forêt boréale mixte, couvre 123 700 km^2 au Québec, soit 8 % du territoire québécois (Ordre des ingénieurs forestiers du Québec 1996), et elle occupe la frange méridionale du biome boréal principalement représenté par les *pessières à épinette noire* (433 600 km^2 soit 28 % du Québec (Ordre des ingénieurs forestiers du Québec 1996)). La *sapinière à bouleau blanc* se situe donc à la transition entre les forêts tempérées au sud et les pessières à épinette noire au nord. Sa répartition actuelle résulte de la compétition naturelle, passée et actuelle (Solomon 1992), entre les espèces qui la composent et celles des écosystèmes voisins (Weber et Flannigan 1997) pour les conditions environnementales climatiques et édaphiques. La forêt boréale mixte apparaît à l'échelle du paysage comme une mosaïque naturelle constituée d'une juxtaposition de peuplements feuillus, résineux ou mélangés, agencés selon le type de dépôt de surface ainsi que les effets des perturbations naturelles successives.

Les perturbations naturelles à grande échelle sont reconnues comme facteurs contrôlant la dynamique de la forêt boréale (Shugart *et al*. 1992). Parmi ces perturbations, le feu est le facteur qui influence le plus le développement de l'écosystème boréal (Johnson 1992; Bergeron *et al*. 1999a) avec en moyenne 1,3 millions ha de forêt brûlée par année (Weber et Stocks 1998). Les composantes du *temps* (saison, durée de la perturbation, fréquence, intervalle, et cycle) et de l'*espace* (superficie touchée et niveaux d'organisation atteints) du régime de feu (Johnson 1992; Weber et Flannigan 1997) ont souvent été étudiées, tant en forêt boréale mixte (Roussopoulos 1978; Furyaev *et al*. 1983; Clark 1988;

Bergeron 1991; Flannigan 1993; Gauthier *et al*. 1996; Bergeron *et al*. 1999a) qu’en pessière (Heinselman 1971; Larsen 1980; Van Wagner 1983; Wein et McLean 1983; Payette et Gagnon 1985; Overpeck *et al*. 1990). Par contre, la composante *amplitude* (Van der Maarel 1993) qui représente l'intensité et la sévérité et se traduit aussi par le type de feu (de surface versus de couronne) a, quant à elle, reçu jusqu’ici peu d’attention en forêt mixte. Or, l’analyse de l’amplitude du régime du feu correspond à l’étude du comportement du feu pendant l’incendie et à ses effets sur la mortalité et la régénération post-incendie. Un exemple révélateur du manque d’information en forêt boréale mixte est le nombre élevé de feux naturels ou prescrits en pessière à épinette noire (31 et 25 respectivement) ayant servi à modéliser le comportement du feu lors de la création de la Méthode canadienne de prévision du comportement des incendies de forêt en 1984 (Forestry Canada Fire Danger Group 1992), alors qu’aucune donnée était disponible pour la forêt mixte (Forestry Canada Fire Danger Group 1992; Hirsch 1996). Cette lacune persiste encore aujourd’hui alors que la compréhension du comportement du feu et des incendies est reconnue comme étant primordiale dans une optique de gestion durable des forêts.

La notion de gestion durable des forêts est née suite à la prise de conscience mondiale à la fin des années 70 de la dégradation toujours croissante des ressources planétaires causée par les activités anthropiques. Différents événements mondiaux clés ont eu lieu suite à la parution de la Stratégie mondiale de la conservation (UICN 1980). Il y a eu la Commission Brundtland en 1987, la Conférence des Nations Unies sur l’Environnement et le Développement durable (CNUED) à Rio de Janeiro en 1992, le Processus de Montréal en 1994, et la Déclaration de Santiago en 1995 (Service canadien des forêts 1996; Conseil Canadien des ministres des forêts 1997). Les orientations issues de ces rencontres successives ont porté sur la conservation de la biodiversité et la mise en place d’une politique de gestion durable avec un cadre commun de critères et indicateurs. De nombreux auteurs ont rapidement conclu que la conservation de la biodiversité devait préférentiellement et concrètement être axée sur la conservation des écosystèmes (Franklin 1993; Gauthier *et al*. 1996; Gouvernement du Québec 1996). En effet, l’hypothèse posée est qu’il est plus facile de préserver les milieux qui engendrent et soutiennent les espèces que de vouloir suivre toutes les espèces une à une ainsi que leurs interactions (Gouvernement du

Québec 1996), cette démarche s'étant révélée peu efficace dans le passé. Dans l'optique de l'aménagement durable des forêts pour la conservation de la diversité des écosystèmes, l'une des voies de recherche et d'exploitation retenue et prometteuse vise à s'inspirer de la dynamique naturelle des mosaïques forestières et notamment des perturbations naturelles qui les structurent (Attiwill 1994; Galindo-Leal et Bunnell 1995) afin de développer des pratiques sylvicoles qui s'inspireraient de leurs effets (Bergeron et Harvey 1997; Bergeron *et al*. 1999b). Il devient donc capital de bien connaître le comportement des incendies pour mettre en place une gestion durable et efficace en forêt boréale mixte. Ces connaissances doivent porter tant sur les conditions favorables au sein de la végétation et des conditions climatiques qui déclenchent un feu à l'échelle du paysage, que sur le comportement du feu au sein des peuplements, ou que sur les effets du feu sur la mortalité et la régénération post-incendie aux différentes échelles de perceptions.

La végétation, en tant que combustible, les conditions climatiques, ainsi que la topographie constituent les trois facteurs qui composent l'environnement du feu (Schroeder et Buck 1970; Agee 1997), et chacun d'eux est décrit selon plusieurs variables (Brown et Davis 1973). Ces facteurs sont en constante interaction, et un incendie se déclare puis se propage lorsque les conditions environnementales dites favorables au feu sont atteintes. Toutefois, une polémique existe concernant le rôle de chacun des trois facteurs (Agee 1997) car certaines études concluent que la végétation (Bessie et Johnson 1995) peut ne pas avoir de rôle significatif.

1.1. AVANT L'INCENDIE: LES CONDITIONS FAVORABLES

1.1.1. Le climat

Les conditions météorologiques représentent le facteur de l'environnement du feu qui change le plus rapidement au cours du temps et qui influence le plus fortement l'apparition puis le comportement d'un incendie (Schroeder et Buck 1970; Trabaud 1989). La saison des feux en forêt boréale est déterminée par la réorganisation estivale de la circulation atmosphérique au dessus de la forêt. Le courant d'air froid de l'Arctique, stable en hiver au dessus de la forêt boréale, est repoussé vers le nord par l'entrée de courants d'air chaud plus instables en provenance des Tropiques ou du Pacifique (Johnson 1992). Ces courants d'air chaud peuvent dessécher rapidement les combustibles et générer des éclairs. A cette circulation atmosphérique, active sur une échelle de plusieurs mois, se superpose une circulation atmosphérique d'Ouest en Est beaucoup plus rapide (de l'ordre de quelques jours à quelques semaines). Cette circulation rapide est le résultat de l'alternance régulière dans la troposphère de dépressions et d'anticyclones près du front polaire. Les conditions météorologiques propices au départ d'incendies sont rencontrées lorsque l'alternance s'arrête et qu'un anticyclone stationne plus d'une semaine sur une même zone. Ce phénomène de blocage de masse d'air est appelé anomalie de hautes pressions (Johnson 1992). Les températures élevées, associées à une faible humidité et à de légers vents, dessèchent alors rapidement les combustibles. Lorsque le blocage disparaît, une dépression chargée d'humidité et de chaleur sous forme d'orage envahit la zone (Overpeck *et al*. 1990). Les incendies déclenchés par la foudre sont alors alimentés par des vents violents crées au niveau du front, et le feu se propage rapidement sur de vastes superficies. Les incendies déclenchés par la foudre représentent en moyenne 35% du nombre total d'incendies, et couvrent environ 85% de la surface totale brûlée. Cependant 2 à 3% des feux de plus de 200 ha pouvent contribuer jusqu'à 98% de la surface annuelle brûlée (Weber et Stocks 1998).

1.1.2. La végétation

La végétation représente quant à elle, le second facteur qui change au cours du temps dans l'environnement du feu, mais elle est le facteur primordial, car sans combustible il n'y a pas de feu possible (Brown et Davis 1973). A l'échelle du paysage, la mosaïque forestière de la sapinière à bouleau blanc est constituée de la juxtaposition de peuplements se distinguant les uns des autres par leur superficie, leur forme, leur composition, et leur âge depuis le dernier feu (Bergeron 2000). La succession secondaire post-incendie de cette forêt a été analysée principalement en terme de composition (Bergeron et Dubuc 1989). Pour la région étudiée, ces auteurs ont ainsi montré qu'au niveau de la canopée s'opérait un remplacement dans la dominance des espèces en fonction du temps. Les espèces héliophiles telles que le peuplier faux-tremble (*Populus tremuloides* Michx.), le bouleau blanc (*Betula papyrifera* Marsh.), et le pin gris (*Pinus banksiana* Lamb.), atteignent les premières la canopée et dominent le peuplement pendant 100 ans environ après le passage du feu, l'espèce dominante après l'incendie étant fonction de la composition du peuplement avant incendie (Weber et Stocks 1998). Par la suite, les espèces plus tolérantes à l'ombre telles que le sapin baumier (*Abies balsamea* (L.) Mill.), l'épinette blanche (*Picea glauca* (Moench) Voss)), l'épinette noire (*Picea mariana* (Mill.) BSP), et le cèdre blanc (*Thuya occidentalis* L.) atteignent la canopée et transforment peu à peu le peuplement feuillu en peuplement mixte (100 à 150 ans après le feu), puis en peuplement conifère (200 ans après l'incendie). La composition du peuplement tendra alors vers une dominance du sapin puis du cèdre jusqu'à ce qu'un nouveau feu survienne et redémarre la succession secondaire à son stade initial (Bergeron et Dubuc 1989; Bergeron et Dansereau 1993; Bergeron 2000). Ce changement de composition dans la canopée au cours du temps est fidèlement traduit dans l'évolution de la biomasse arborescente vivante. En effet, la biomasse arborescente vivante totale augmente en effet tant que les peupliers dominent le peuplement et que leur diamètre augmente (Paré et Bergeron 1995). Dès que les conifères commencent à remplacer les peupliers, la biomasse vivante arborescente diminue pour devenir relativement faible vers 200 ans après le dernier feu, lorsque le peuplement est devenu coniférien. L'explication vient entre autre de la productivité des peupliers qui est la plus forte par rapport aux autres essences présentes (Paré et Bergeron 1995). Cette biomasse arborescente vivante n'aura cependant pas d'effet lors du départ d'un incendie car elle représente des combustibles vivants et de gros diamètres. Par contre, si l'incendie devient un feu de couronne, le feuillage des arbres et les branches contribueront alors directement à la propagation du feu (Brown et Davis 1973).

L'ignition d'un incendie requiert sur le sol des combustibles fins et secs, soit les branches et les brindilles mortes, la litière, les herbacées et les arbustes de petits diamètres, morts ou desséchés (Brown *et al*. 1982; Schimmel et Granström 1997). En effet, les combustibles morts réagissent plus rapidement aux variations d'humidité que les combustibles vivants (Valette 1990), et la dessiccation des combustibles est d'autant plus rapide que leur diamètre est petit (Burgan et Rothermel 1984; Johnson 1992). Toutefois la charge, la composition, ainsi que l'arrangement spatial des débris ligneux auront des influences différentes sur la disponibilité des combustibles (susceptibilité au feu d'un peuplement), et sur le comportement du feu au cours de l'incendie (McRae *et al*. 1979; Schimmel et Granström 1997). Concernant le sous bois, De Grandpré *et al*. (1993) ont montré que la composition de la végétation variait relativement peu au cours de la succession, alors que l'abondance des espèces tendait à augmenter. Toutefois, ces auteurs n'ont pas quantifié ce changement en terme de variation de biomasse, aspect très important pour l'étude du départ du feu et de sa propagation. La biomasse des arbustes a été quantifiée spécifiquement dans de nombreuses régions dans le passé (Post 1969; Telfer 1969; Brown 1976; Roussopoulos 1978), grâce à des relations allométriques entre le diamètre basal et la biomasse (Brand et Smith 1985; Buech et Rugg 1995). Cependant, il a souvent été mentionné que les relations trouvées présentaient une forte variabilité régionale susceptible de rendre invalide l'utilisation de telles équations dans une autre région (Roussopoulos 1978). Or, les espèces arbustives de la sapinière n'ont jamais été analysées au Québec. Les mousses, quant à elles, sont reconnues pour être un très bon combustible (Brown et Davis 1973) car elles ont la capacité, même vivantes, de supporter une perte importance de leur teneur en eau au cours d'une période de sécheresse. Cependant, ce type de combustible n'excède pas 3% du couvert forestier dans les peuplements «riches» en mousses (De Grandpré *et al*. 1993) et ne constitue donc pas en sapinière un type de combustible important. La litière, autre type de combustible impliqué lors de l'ignition de la flamme puis dans sa propagation (Brown et Davis 1973), n'augmente ni en charge (Paré *et al*. 1993), ni en épaisseur (De Grandpré *et al*. 1993) au cours du temps dans la région étudiée. Finalement, les débris ligneux telles les brindilles et les branches de diamètre inférieur à 7.6 cm (Burgan et Rothermel 1984) constituent un combustible très important dans l'ignition et la propagation du feu (Roussopoulos 1978; Harmon *et al*. 1986; Johnson 1992). Cependant aucune étude a été réalisée à ce jour sur les débris ligneux dans la forêt boréale mixte du Québec.

1.1.3. La topographie

Contrairement aux deux autres facteurs, la topographie est un facteur quasi permanent de l'environnement du feu pour une zone donnée. Il est dont possible de déterminer ou de prévoir son influence (Trabaud 1989). Les zones élevées sont, par example plus susceptibles d'attirer les impacts de foudre (Flannigan et Wotton 1991), et donc d'être le lieu de départ d'un incendie. La topographie exerce également un effet direct et indirect sur le comportement des incendies. En effet, la topographie influence directement les conditions micro et méso-météorologiques qui, à leur tour, influencent la teneur en eau du combustible et de la vitesse du vent près du sol (Schroeder et Buck 1970; Trabaud 1989). L'exposition d'un terrain, correspondant à l'orientation que ce terrain occupe par rapport à la direction géographique à laquelle il fait face, a une grande influence indirecte sur le départ d'un feu et son comportement. En effet, l'exposition d'un versant influence la quantité de chaleur locale reçue et donc la teneur en eau des combustibles vivants ou morts. De façon générale, les versants sud et sud-ouest présentent donc les conditions les plus favorables pour une inflammation rapide et la propagation des feux car ils reçoivent un ensoleillement plus direct qui augmente les températures de l'air et des combustibles, et qui diminue leur teneur en eau. De plus, ces versants sont battus plus souvent par des vents desséchants. Pour évaluer le comportement du feu, il faut également en principe interpréter correctement deux grands critères altitudinaux: l'altitude par rapport au niveau de la mer et l'altitude relativement à la région environnante (Trabaud 1989). Toutefois, la forêt boréale mixte étudiée étant une région relativement plate, seule l'analyse de l'altitude par rapport à la région environnante pourrait être pertinente dans le cadre de peuplements situés différemment selon la pente. L'augmentation de la déclivité favorise une plus grande vitesse de propagation, les feux se propageant plus rapidement en escaladant un versant que sur un terrain plat. En effet, les combustibles situés en haut de pente reçoivent une plus grande chaleur par rayonnement et convection (Jean 1992), et des vents ascendants sont généralement engendrés par la chaleur naturelle dégagée par le feu (Trabaud 1989).

1.2. PENDANT L'INCENDIE: LE COMPORTEMENT DU FEU

1.2.1. De l'ignition à l'extinction d'un feu

Le début d'un incendie est représenté par le moment où une source d'énergie ou de chaleur supérieure à 100°C entre en contact avec un combustible de telle sorte que celui-ci se dessèche complètement et s'échauffe (Johnson 1992). Si la température atteinte dépasse 200°C, le combustible commence à se décomposer chimiquement sous l'effet de la chaleur (phase de pyrolyse, avec tâches noires rougissantes) et les produits chimiques tels que les terpènes, les huiles essentielles, et les résines se volatilisent (Brown et Davis 1973; Trabaud 1976). L'ignition de la flamme se produit lorsque ces composés chimiques volatilisés sous forme de mélange gazeux explosent (300 à 400°C). La combustion avec flamme se poursuit jusqu'à ce que les derniers composés chimiques de poids moléculaires faibles à modérés avec des chaleurs de combustion élevées se soient volatilisés et enflammés (Trabaud 1976). Lorsqu'il ne reste plus que la lignine et ses composantes, produits chimiques très stables et de poids moléculaires élevés, la combustion s'effectue à très haute température, non plus dans l’air sous forme de flamme, mais plutôt sous forme de rougeoiement des braises par oxydation à la surface du solide. Ces braises représentent le produit de décomposition du carbone, constituant principal de la lignine (Brown et Davis 1973; Johnson 1992).

1.2.2. Le comportement du feu

La propagation d’un feu est le résultat d’une suite d’ignitions de flamme et de combustion des particules de proche en proche, lors du transfert de chaleur qui a lieu pendant la combustion avec flamme (Pyne 1984), principalement sous forme de convection et de radiation (Brown et Davis 1973; Johnson 1992). L'intensité dégagée, la vitesse de propagation et la quantité de combustible consumée constituent les trois composantes primaires du comportement du feu alors que la surface brûlée représente une composante secondaire (Andrews 1986; Hirsch 1996). Toutefois, pour être explicites, toutes ces composantes doivent être mesurées une fois l'équilibre de propagation de la flamme atteint. En effet, juste après son ignition, la flamme tend à accélérer durant les premiers moments de la propagation (Pyne 1984; Forestry Canada Fire Danger Group 1992). Par la suite, la vitesse se stabilise et l'état d'équilibre est alors atteint (Johnson 1992; Bilgili 1995; Hirsch 1996). A partir de ce moment, le comportement du feu et ses composantes sont constantes, à moins que le feu en progressant dans l'espace ne rencontre un nouveau type de combustible, auquel cas, les composantes du comportement du feu se réajustent.

Ainsi, la vitesse de propagation du front de flamme (en m/min) est la distance parcourue perpendiculairement au front de flamme par unité de temps ou encore le rapport entre l’énergie disponible pour échauffer le combustible et l’énergie nécessaire pour l’ignition du combustible (Johnson 1992), elle-même étant fonction des propriétés chimiques et thermo-physiques des composantes du combustible.

L’intensité du front de flamme (en kW/m) représente quant à elle, le taux d’énergie dégagée par unité de masse consumée par unité de longueur du front de flamme (Johnson 1992). Son calcul nécessite donc la prise en compte de la masse de combustible consumée, de sa chaleur de combustion (énergie dégagée correspondant en moyenne à 18700 kJ/kg, Johnson (1992)), et de la vitesse de propagation (McRae *et al*. 1979).

La superficie brûlée permet de voir les effets du feu et de son comportement au niveau du paysage car elle détermine la taille, la forme, et l'arrangement spatial des différentes parcelles en régénération.

Van Wagner (1983) propose cinq types de feux basés sur l'intensité dégagée: les feux de très faible intensité(< 10 kW/m), ceux de surface à contre vent (100 - 800 kW/m), ceux de surface sous le vent (200 - 15000 kW/m), ceux de couronne passifs ou intermittents (8000 - 30000 kW/m), et finalement les feux de couronne actifs ou indépendents (jusqu'à 100000 kW/m). Parmi ces divers types de feux, la forêt boréale mixte, de par sa constitution, pourrait difficilement être le lieu des feux de très faible intensité, car ceux-ci ne se propagent normalement que dans les humus très profonds (>30 cm d'épaisseur). Etant donné que l'épaisseur moyenne de l'humus en sapinière dépasse rarement 10 cm (De Grandpré *et al*. 1993; Bergeron 2000) les feux de très faible intensité auront tendance à s'éteindre avant d'avoir atteint l'état d'équilibre. Par contre, les quatre autres types de feux sont susceptibles de se propager dans le paysage de la forêt boréale mixte. En général la forme finale d'un feu est une ellipse simple ou double, d'autant plus allongée (rapport longueur/largeur élevé) que la vitesse du vent est rapide ou que la topographie est prononcée (Green 1983; Green *et al*. 1983; Pyne 1984). La zone interne, mais proche du périmètre de l'ellipse, brûlera selon une combustion avec flammes, alors que le centre de l'ellipse brûlera, après le passage du front de flamme, sous forme de rougeoiements avant de s'éteindre (Pyne 1984). La zone périphérique de l'ellipse sous le vent sera le lieu du front de flamme et se propagera à une plus grande vitesse que le reste du périmètre (Pyne 1984).

La vitesse de propagation et l'intensité du feu permettent de comprendre l'effet du comportement du feu sur la mortalité et la régénération post-incendie car lorsqu'elles sont combinées, elles reflètent la sévérité du feu. La sévérité du feu est une variable écologique qui traite des conséquences du passage du feu. La sévérité du feu est d'autant plus grande que le feu se propage lentement ou/et que l'intensité dégagée est élevée (Johnson and Miyanishi 1995).

1.2.3. Méthodes d'analyse du comportement du feu

Il existe principalement trois façons d'analyser le comportement du feu. Dans tous les cas il est primordial d'avoir une description qualitative (composition et arrangement spatial) et quantitative (charges) des différents types de combustibles, avant et après le passage du feu:

1. Suite au déclenchement d'un feu naturel, une opportunité d'analyse apparaît (Forestry Canada Fire Danger Group 1992): les mesures de vitesse de propagation de la flamme, de quantité de combustible consumée, et de superficie brûlée prises au cours de l'incendie sont ensuite confrontées à des mesures prises dans des peuplements similaires de la région ayant échappé au feu et qui serviront de mesures pré-incendie.
2. Un feu prescrit est déclenché sur une certaine superficie pour laquelle on a préalablement mesuré les différents types de combustibles (Stocks 1987; Forestry Canada Fire Danger Group 1992). Les mesures prises au cours de la propagation du feu seront ensuite reliées aux mesures pré et post-incendie.
3. Les caractéristiques des peuplements sont mesurées en détail (soit la composition de la canopée, les types de combustibles, leur arrangement spatial et leurs charges) et elles sont incorporées dans un simulateur de comportement du feu (Andrews 1986; Forestry Canada Fire Danger Group 1992; Bessie et Johnson 1995; Bilgili 1995). Dans ce cas, on obtient des prédictions du comportement du feu qui pourront être comparées à de réelles observations, si elles existent (Schimmel et Granström 1997). Les simulateurs du comportement du feu développés en Amérique du Nord se basent soit sur des données empiriques (cas de la Méthode Canadienne de prévision du comportement des incendies de forêt (Forestry Canada Fire Danger Group 1992)), soit sur les propriétés physiques et chimiques des combustibles (cas du système BEHAVE: Burgan et Rothermel (1984), et Andrews (1986)).

1.3. APRES L'INCENDIE: MORTALITE DES ARBRES

Les effets occasionnés par le passage du feu sont aussi complexes que le comportement du feu lui-même car nous l'avons vu, les facteurs impliqués sont régis par de multiples variables, et le phénomène a lieu dans un environnement vivant complexe, mettant en jeu diverses strates de végétation (Van Wagner 1977, 1983; Stocks 1987; Shugart *et al*. 1992; Schimmel et Granström 1997). Parmi les principales actions du feu en forêt nous pouvons citer la disparition de matériel ligneux consumé, les dommages portés aux végétaux (pouvant causer leur mortalité) ainsi qu'aux sols, dûs au dégagement de chaleur pendant l'incendie, et finalement la production de résidus pouvant entraîner des modifications au niveau des caractéristiques physico-chimiques du sol principalement (Brown et Davis, 1973, Smith *et al*. 1998). Bien que tous les compartiments de la forêt soient affectés plus ou moins rapidement et qu'ils interagissent ensemble, nous nous contenterons de présenter plus en détail les effets du feu sur les espèces arborescentes.

1.3.1. Les parties touchées par le feu

Les racines, la tige, et la couronne sont les trois zones chez les arbres qui sont susceptibles d'être endommagées lors du passage du feu (Brown et Davis 1973; Ryan 1998). De plus il peut y avoir un effet combiné des dommages occasionnés aux différents organes (Ryan et Reinhardt 1988; Ryan 1998). Dans tous les cas, la blessure puis la mort sont causées par l'augmentation de la température interne des cellules (Brown et Davis 1973). La température létale moyenne des cellules est de 60 - 70°C (Van Wagner 1973; Johnson 1992), et la mort des tissus survient d'autant plus vite que cette température interne est atteinte rapidement et que l'exposition à cette température se prolonge (Brown et Davis 1973).

1.3.2. La résistance au feu: variables caractéristiques et différences spécifiques

Les différences de résistance au feu observées entre les espèces et au sein de chacune d'elles, s'expliquent par les caractéristiques de la couche isolant les parties exposées au feu.

Ainsi, plus les racines sont implantées profondément dans le sol minéral, plus celui-ci les isole de la chaleur dégagée par le front de flamme et moins elles sont endommagées (Brown et Davis 1973; Peterson et Arbaugh 1989). Si l'humus est également très épais, il peut ne brûler que superficiellement, protégeant ainsi les racines. Toutefois, les feux en forêt boréale étant pour la plupart très intenses (Heinselman 1971; Van Wagner 1983; Johnson 1992), il est très fréquent d'observer suite au passage du feu, des zones de sol minéral exposé autour de la base des troncs (Johnson 1992). Dans ce cas là, la combustion de l'humus prolonge le temps de résidence de la flamme et des températures élevées, et les racines tendent à être endommagées (Brown et Davis 1973; Ryan et Frandsen 1991). Par contre, ces surfaces de sol minéral exposées sont favorables à la germination des graines des principales espèces rencontrées en forêt boréale (Johnson 1992).

Concernant la tige, plus l'écorce d'un arbre est épaisse et compacte, plus le phloème et le cambium sont protégés de la chaleur dégagée par la flamme le long du tronc, et moins ce dernier est endommagé (Brown et Davis 1973). L'épaisseur de l'écorce varie selon les espèces (Brown et Debyle 1987; Ryan et Reinhardt 1988; Ryan 1998), et au sein d'une même espèce, elle augmente avec l'augmentation du diamètre des arbres (Brown et Debyle 1987; Peterson et Arbaugh 1989; Ryan et Frandsen 1991). La présence de fissures et craquelures dans l'écorce diminuera cependant son effet isolant (Brown et Davis 1973; Ryan et Frandsen 1991). Les arbres de petits diamètres subissent en général une mortalité cambiale tout autour de leur circonférence entraînant automatiquement leur mortalité (Gutsell et Johnson 1996). Par contre, les arbres de plus gros diamètres peuvent présenter une mortalité cambiale partielle. La zone morte forme éventuellement une cicatrice et se présente alors sous forme triangulaire dont la base est parallèle au sol. Elle est toujours située sur le côté de l'arbre sous le vent (par opposition au côté exposé au vent), car la chaleur dégagée par le front de flamme y est plus élevée et persiste plus longtemps suite à la création et au maintien dans cette zone de deux vortex de chaleur (Gutsell et Johnson 1996). L'aubier, partie vivante en périphérie du tronc et située sous l'écorce, poursuit son développement dans les années subséquentes, formant ainsi une excroissance sur le pourtour de la cicatrice.

Finalement, il a été démontré sur les espèces de l'ouest de l'Amérique du nord que plus le feuillage de la couronne est éloigné de la source de chaleur, moins il a de risque de s'enflammer (Brown et Davis 1973; Peterson 1985). A cela, s'ajoute la présence ou non de branches basses qui peuvent servir de continuum à la flamme (Brown et Davis 1973), lui permettant ainsi de gagner la couronne facilement. La composition chimique du feuillage, sa teneur en eau, ainsi que les changements phénologiques (Brown et Davis 1973; Johnson 1992; Harrington 1993) sont autant de variables supplémentaires qui déterminent son inflammabilité, et peuvent ou non favoriser sa combustion (Brown et Davis 1973).

La plupart des études sur la mortalité post-incendie ont été réalisées dans l'ouest de l'Amérique du Nord et ont principalement porté sur des conifères tels que *Pseudotsuga menziesii* (Ryan et Reinhardt 1988; Ryan *et al*. 1988; Peterson et Arbaugh 1989; Keane *et al*. 1990), *Pinus ponderosa* (Thomas et Agee 1986; Keane *et al*. 1990; Swezy et Agee 1991;

Harrington 1993; Regelbrugge et Conard 1993; Ryan 1998), *Pinus contorta* (Peterson 1985; Reinhardt et Ryan 1988; Swezy et Agee 1991; Ryan 1998), *Picea engelmannii* (Ryan et Reinhardt 1988), *Abies lasiocarpa* (Perterson 1985; Ryan et Reinhardt 1988), et *Abies concolor* (Thomas et Agee 1986; Swezy et Agee 1991). A l'opposé, peu d'espèces feuillues ont été retenues pour ce type d'études: la mortalité post-incendie de *Populus tremuloides* a été analysée par Brown et Debyle (1987), alors que celles de certains chênes associés à des érables ou à des pins ont été étudiées par Huddle et Pallardy (1999), Harmon (1984) et Regelbrugge et Conard (1993). Contrairement au nombre important d'études traitant de la régénération post-incendie des espèces de la forêt boréale mixte (Bergeron et Dubuc 1989; Zasada *et al*. 1992; Bergeron et Brisson 1994; Galipeau *et al*. 1997; Simard *et al*. 1998), la mortalité post-incendie de ces espèces a été très peu étudiée.

1.3.3. Les méthodes d'analyses de la mortalité post-incendie.

Les données de mortalité sont souvent binaires (arbre mort ou vivant) et sont donc analysées grâce à des modèles de régression logistiques faisant intervenir, comme variables explicatives, les variables dendrométriques (diamètre, épaisseur de l'écorce, hauteur de l'arbre, hauteur de la base de la couronne) et les variables reliées aux dommages occasionnés au feuillage, aux tiges, et aux racines (Peterson et Arbaugh 1986; Brown et Debyle 1987; Regelbrugge et Conard 1993; Ryan 1998). Les variables traduisant les dommages portés à la couronne sont souvent les plus explicatives (hauteur, volume, ou pourcentage du houppier brûlé), alors que peu de variables portant sur les dommages occasionnés aux racines ont été testées (Ryan 1998). Les dommages occasionnés à la tige sont souvent significatifs dans les modèles lorsqu'ils sont décrits par la hauteur du charbon sur l'écorce (Peterson et Arbaugh 1986, 1989; Ryan et Reinhardt 1988). Toutefois, un grand nombre d'études ne valident pas les modèles logistiques trouvés par l'emploi de données nouvelles et indépendantes, critère pourtant primordial dans le domaine de la modélisation.

De la même façon, rares sont les études qui mettent en relation les variables du comportement du feu, telles que l'intensité dégagée ou la longueur de la flamme, avec les effets occasionnés aux arbres (Van Wagner, 1973; Brown et Debyle 1987; Reinhardt et Ryan, 1988; Ryan et Frandsen 1991). Pourtant de telles relations pourraient être employées comme l’ont fait Reinhardt et Ryan (1988) pour développer des outils d’aménagement forestiers (les nomogrammes) mettant en relation les modèles logistiques de mortalité et les caractéristiques du comportement du feu. Ces outils pourraient être utilisés dans différents buts : pour améliorer les prévisions de récupération de bois mort après le passage d’un feu naturel dans une zone exploitée, utiliser des brûlages dirigés pour diminuer la charge potentielle de combustible et prévenir à moyen terme le risque d’incendie, ou pour préparer le terrain pour la régénération.

1.4. OBJECTIF GÉNÉRAL

Bien que l’on reconnaisse que les incendies forestiers contrôlent la dynamique de la forêt boréale mixte, il existe peu d’informations sur les facteurs qui favorisent le départ d’un feu, sur le comportement du feu pendant l’incendie, ni sur les facteurs qui influencent la mortalité post-incendie. Ainsi cette thèse a pour objectifs de caractériser les facteurs constituant l’environnement du feu, d’analyser leurs rôles respectifs dans le comportement du feu, et de clarifier le processus de mortalité post-incendie des principales essences de la forêt boréale mixte.

L’hypothèse générale de ce projet est que la composition variable des peuplements de forêt boréale mixte, en place avant l’incendie (feuillus, mixtes, ou résineux), produit différents comportements du feu, entraînant une mortalité différenciée à l’échelle des peuplements, et une structure du paysage en mosaïque, dont les propriétés varient de celles d’un paysage monospécifique.

L'étude se déroule au Québec en Abitibi principalement dans la forêt d'enseignement et de recherche du lac Duparquet (Figure 1.1) et dans la région de Val Paradis.

1.5. OBJECTIFS SPÉCIFIQUES

Les débris ligneux de petits diamètres constituent de très bons combustibles de surface car, morts et secs, ils s’enflamment les premiers, alimentent la flamme, et favorisent ainsi sa propagation. A l’opposé, la combustion des débris ligneux de plus gros diamètres, de par des teneurs en eau beaucoup plus élevées, ne se déclenche que lorsque le feu dépasse une certaine intensité. De plus la teneur en eau et la nature chimique des particules peuvent influencer significativement le délai d’inflammation ainsi que la propagation du feu. La connaissance de la composition en débris ligneux (charges, distribution des diamètres, et espèces) au sein des peuplements composant la mosaïque forestière est donc primordiale pour comprendre les risques potentiels de départ de feu (disponibilité des combustibles), et le comportement du feu lors d'un incendie. Toutefois, il semble que les débris ligneux et leurs caractéristiques en forêt boréale mixte n'aient jamais été étudiés. C'est pourquoi, le premier chapitre traite de la nature des débris ligneux (composition spécifique, distribution des diamètres, et charges) dans les peuplements et de leurs changements au cours de la succession secondaire. L'hypothèse est que les changements locaux de composition (au sein de la canopée) et les taux de décomposition rapides des espèces étudiées pourraient influencer les modèles d'accumulation des débris ligneux dans le temps. La forêt boréale mixte serait alors un écosystème particulier présentant des modèles d'accumulation locale différents des modèles généralement trouvés.

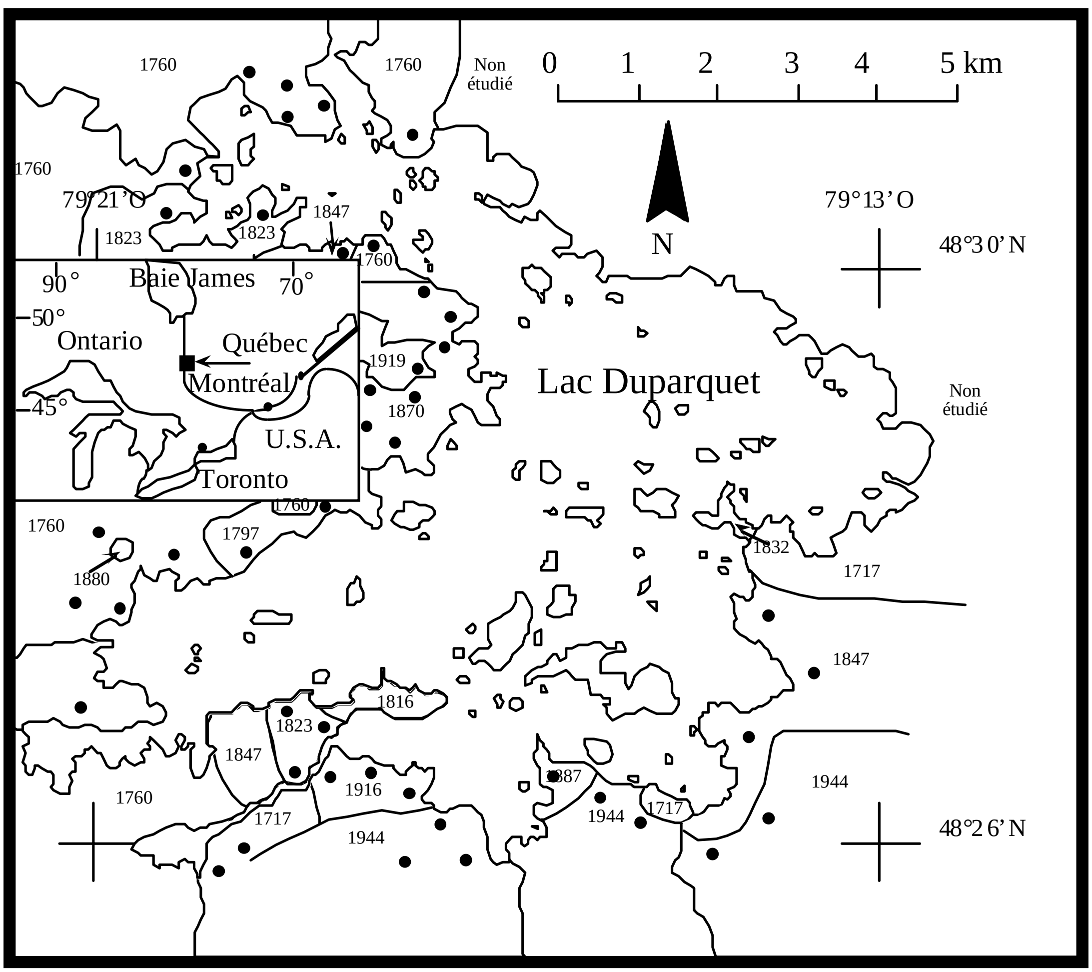

Figure 1.1 Carte des berges du Lac Duparquet (Québec, Canada) présentant les sites d'échantillonnage et les années de feu.

Outre les débris ligneux, la litière et l'humus constituent avec les herbacées et les arbustes autant de combustibles de surface potentiels selon leurs compositions, leur arrangement spatial, et leurs charges. Différentes études ont analysé certains de ces facteurs d'un point de vue de leur composition et de leurs changements au cours de la succession (Roussopoulos 1978; De Grandpré *et al*. 1993; Paré *et al*. 1993). Cette thèse consacre le second chapitre à l'étude simultanée de tous ces combustibles de surface, en terme de composition, de charges, et de leurs changements au cours de la succession, symbolisée par quatre types de peuplements, et à l'analyse de leur effets sur la susceptibilité au feu ("Fire hazard"). La susceptibilité au feu d'un type de peuplement est d'autant plus importante que celui-ci présente simultanément des charges importantes de particules de petits diamètres, ou/et riches en produits chimiques volatiles inflammables (huiles essentielles, résines des conifères...). Le risque potentiel de départ du feu, fondé sur la disponibilité et la qualité des différents combustibles, est ensuite testé par l'utilisation d'un système de prédiction du comportement du feu (le système BEHAVE), basé sur la nature et la charge des différents combustibles de surface (Burgan et Rothermel 1984).

Une fois la végétation caractérisée selon ses différents types de combustibles au sein des peuplements, les rôles de la végétation et du climat sont étudiés lors de l'analyse du comportement du feu à l'échelle du peuplement afin d'évaluer l'influence des caractères locaux à l'echelle du paysage. La topographie, représentant le troisième facteur de l'environnement du feu, est considérée comme une constante puisque les peuplements sélectionnés sont localisés sur des terrains plats. L'hypothèse de travail de ce troisième chapitre est que les deux facteurs ont un rôle significatif mais que la végétation, par la présence de peuplements feuillus et mixtes, peut influencer négativement le comportement du feu. Pour cette étude du rôle de la végétation et du climat dans le comportement du feu, deux approches avaient été retenues à l'origine du projet: des brûlages dirigés et l'utilisation de simulateurs de comportement du feu à l'échelle du peuplement. Toutefois, cette thèse ne présente que les résultats des simulations, les brûlages dirigés n'ayant pu jusqu'à ce jour être réalisés en raison non seulement de conditions météorologiques défavorables, mais aussi d'une incompréhension partielle subsistant encore quant à l'apport d'une telle expérience à la communauté scientifique et forestière oeuvrant au Québec en forêt boréale.

Le dernier chapître aborde la mortalité post-incendie *via* l'effet de la composition des peuplements sur la mortalité des arbres. En effet, suite au feu naturel survenu en juin 1997 en forêt boréale mixte dans la région de Val Paradis (Abitibi, Québec), la résistance au feu des principales essences est d'abord analysée en fonction de l'intensite dégagée au sein des différents peuplement (feuillus, mixtes, et résineux). D'après l'analyse des trois premiers chapitres, les peuplements feuillus et mixtes ont du brûler avec des intensités de feux inférieures à celles produites dans les peuplements résineux. Par la suite, les variables dendrométriques caractéristiques sont testées pour chaque espèce, et la mortalité post-incendie est analysée en relation avec les données du comportement du feu. Les essences conifériennes sont supposées être plus résistantes au feu que les essences feuillues, mais elles doivent être plus sensibles par leur hauteur totale et la base de houppier que les peupliers. Les régressions logistiques retenues sont alors utiliséés dans le but de créer des nomogrammes spécifiques.

CHAPITRE II

COARSE WOODY DEBRIS IN THE SOUTHEASTERN CANADIAN BOREAL FOREST: COMPOSITION AND LOAD VARIATIONS IN RELATION TO STAND REPLACEMENT

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2.1. RÉSUMÉ

Les caractéristiques quantitatives et qualitatives des débris ligneux au sol et debout (CWD) sont analysées en relation avec l'âge et la composition des peuplements de la forêt boréale mixte canadienne. Quarante-huit peuplements, âgés de 32 à 236 ans, et régénérés après feu, sont échantillonnés sur les argiles mésiques. La méthode quadrant centré sur le point est utilisée pour analyser la composition de la canopée et sa structure (arbres morts et vivants). La méthode de la ligne d'intersection sert à échantillonner tous les débris ligneux couchés. La charge totale au sol, la densité moyenne des chicots et leur volume par peuplement sont similaires aux charges rencontrées dans différents écosystèmes boréaux. Des régressions linéaires et non linéaires montrent que le temps depuis le dernier feu et la composition de la canopée expliquent significativement les changements temporels enregistrés par les débris au sol, alors que seul le temps depuis le dernier feu intervient significativement dans les changements concernant les chicots. Les modèles d’accumulation des CWD dans le temps sont différents du modèle en "U" en raison de l'absence d'une diminution de charge initiale, du taux de décomposition rapide des espèces (responsables d'une accumulation à court terme), et de la succession par le remplacement des espèces dans le temps.

2.2. ABSTRACT

Quantities and structural characteristics of coarse woody debris (CWD) (logs and snags) are examined in relation to stand age and composition in the Canadian mixedwood boreal forest. Forty-eight stands originating after fire (from 32 to 236 years) are sampled on mesic clay deposits. The point centered quadrant method is used to record canopy composition and structure (living trees and snags). The line intersect method is used to sample logs of all diameters. Total log load, mean snag density and volume per stand are similar to other boreal stands. Linear and non-linear regressions show that time since fire and canopy composition are significant descriptors for log load changes, whereas time since fire is the only significant factor for snag changes. Coarse woody debris accumulation models through time since fire are different from the U-shaped model because the first initial decrease from residual pre-disturbance debris is missing, the involved species have rapid decay rates with no long-term accumulation, and the succession occurs from species replacement through time.

2.3. INTRODUCTION

Coarse woody debris (CWD), as standing dead trees (snags) and fallen boles and branches (logs), contribute to the structure, microhabitat diversity, and nutrient cycling of forests (Harmon *et al*. 1986; Lambert *et al*. 1980; Stauffer and Best 1980). The functional importance of CWD depends not only on their quantities, but also on their size distribution, degree of decay, species, position (snags vs. logs) and spatial arrangement (Harmon *et al*. 1986). Insight into the dynamics of CWD will guide managers in understanding the impacts of current management regimes on the CWD cycle and their consequences on biodiversity, stand health and ecosystem productivity (Harmon *et al*. 1987; Keddy and Drummond 1996; Lee *et al*. 1997; Sturtevant *et al*. 1997).

The availability of CWD in early stages of forest development is almost entirely dependent on individual stand history (Spies *et al*. 1988), i.e., pre-disturbance and disturbance-generated debris, and residual standing trees. Coarse woody debris within the accumulation stage are generated by the regenerating stand, and CWD are related to the standing forest structure, stem growth and mortality (Harmon *et al*. 1986), disturbance return interval, and decay rates. However, previous attempts relating standing tree structure (living basal area) to CWD have often yielded poor results (Sturtevant *et al*. 1997; Muller and Liu 1991). Several studies of CWD within forest chronosequences have described a general U-shaped temporal pattern observed in northern hardwoods (Bormann and Likens 1994), wave-regenerated balsam fir (*Abies balsamea*) (Lambert *et al*. 1980), and Douglas-fir - western hemlock (*Tsuga heterophylla*) forests (Agee and Huff 1987; Spies *et al*. 1988). This temporal pattern seems to characterize the dynamics of undisturbed forests (Lambert *et al*. 1980), naturally disturbed forests (Lee *et al*. 1997; Sturtevant *et al*. 1997; Tyrrell and Crow 1994) and managed forests that have been clear-cut (Bormann and Likens 1994; Sturtevant *et al*. 1997). In general, debris levels are high following the initial stand disturbance. Residual debris then decline over time, while there is a gradual additional input from the regenerating stand. As stands mature, tree mortality increases and contributes to the CWD reservoir. Debris levels usually peak during a transitional stage as the even-aged stand senesces into a more uneven-aged structure. Finally, levels decline resulting in the reverse J-shaped diameter distribution of uneven-aged forests (Harmon *et al*. 1986; Spies *et al*. 1988).

In the Canadian mixedwood boreal forest, the forest vegetation is characterized by a gradual change in canopy species dominance following major disturbances such as fires (Bergeron and Dubuc 1989; Payette 1992; Roussopoulos 1978). All tree species regenerate within a few years following the fire, but the different growth rates create a successive dominance in the canopy from shade-intolerant deciduous species to shade-tolerant coniferous species. Paré and Bergeron (1995) have shown that the species replacement and the occurrence of spruce budworm outbreaks affect the above ground biomass. As trees are the CWD producers, it would be interesting to see if CWD accumulation through time follows the same pattern as compared to the aboveground biomass. Our objectives in this study were twofold. First, we wanted to document the CWD quantities, species composition and distributions to determine the appropriate CWD accumulation models for the Canadian mixedwood boreal forest. Secondly, we wanted to test the effect of stand composition and time since fire as potential factors that could simultaneously influence changes in CWD components. Our hypothesis is that in the southeastern Canadian boreal forest, temporal patterns will differ from the traditional U-shaped model because there is a species replacement with time, and the involved species produce different CWD loads that have rapid but different decay rates.

2.4. MATERIALS AND METHODS

2.4.1. Study area description

The study area is located around Lake Duparquet (Bergeron *et al*. 1995), located in the Clay Belt of northwestern Quebec (48°30'N, 79°20'W), a large physiographic region characterized by lacustrine clay deposits left by the proglacial lakes Barlow and Ojibway (Vincent and Hardy 1977). The area surrounding Lake Duparquet has forests that have never been commercially harvested. Lake Duparquet is situated at the southern limit of the boreal forest in the Missinaibi-Cabonga section (Rowe 1972), which is characterized by balsam fir (*Abies balsamea* (L.) Mill.) as the dominant species. Black spruce (*Picea mariana* (Mill). BSP), paper birch (*Betula papyrifera* Marsh.), white spruce (*Picea glauca* (Moench) Voss) and trembling aspen (*Populus tremuloides* Michx.) also represent important components in the forest. The mean annual temperature is 0.6° C; mean annual precipitation is 822.7 mm;

and mean annual frost-free period is 64 days. However, freezing temperatures may occur throughout the year (Environment Canada 1993).

2.4.2. Data collection

Sampling design

Forty-eight stands initiated after lethal fires dating from 32 to 236 years ago were selected on gently sloping mesic clay deposits. Time of stand initiation has been determined in previous dendrochronological studies (Bergeron 1991; Dansereau and Bergeron 1993). Each stand was sampled with a 30 m-sided equilateral triangle (McRae *et al*. 1979). The triangular layout was used to minimize bias in situations where logs are not randomly oriented, and to cover the variation in CWD distribution (Van Wagner 1980). This delineated sampling area was used to evaluate CWD composition and loads, and to analyze the living canopy composition (Appendix 2.1).

Canopy composition

We sampled canopy characteristics (tree species, diameter at breast height (DBH), total height of trees, and status in the canopy as dominant or dominated tree) using the point centered quadrant method (Mueller-Dombois and Ellenberg 1974; McRae *et al*. 1979). Six points were set up along triangle edges and forty-eight trees were recorded per stand: twenty-four dominating trees and twenty-four suppressed trees. The forty-eight distances between trees and the quadrant centers were measured to calculate densities and basal areas per stand and per species (McRae *et al*. 1979).

Coarse woody debris

In the analyses, logs represent boles and branches, while snags represent standing dead trees. The line intersect method (Van Wagner 1968, 1980) was used along each triangle side to sample logs. All surficial woody pieces crossing the line were recorded into six diameter size classes. Pieces less than 7 cm in diameter were recorded within the five classes recommended by McRae *et al*. (1979) using different intersect line lengths for each diameter class: class I = 0-0.49 cm on 15m long; class II = 0.5-0.99 cm on 30m; class III = 1-2.99 cm on 45m; class IV = 3-4.99 cm on 60m; class V = 5-6.99 cm on 75m long. For these classes, total number of woody pieces and species proportions were measured (Appendixes 2.2 and 2.3).

We recorded the species and diameter size (to the nearest 0.1 cm) of each item greater than 7 cm in diameter. These large logs constitute the sixth class. We did not use decay classes because only logs on the litter (upper duff logs) were measured, and few showed different decay states. Moreover, the moss cover was very infrequent in all stands, as reported previously by De Grandpré *et al*. (1993). The equation to calculate corrected fuel loading per species and diameter size class (Van Wagner 1980) is:

$$W = \frac{(Gk)}{L} * \sum d^{2} * a * c \qquad (2.1)$$

where W is weight per unit ground area (t/ha), G is specific gravity (g/cm^3), k is a constant corresponding to 1.234 (see Van Wagner 1980, p3.), L is the length of sample line (m), d is piece diameter at intersection (cm), a is a correction factor for nonhorizontal angle of fuel pieces and it corresponds to 1.13 (see Brown 1974), and c is the slope correction factor (see McRae *et al*. 1979, Table 4) computed from the tilt angle of each triangle side. Moreover,

$$\sum d^{2} = n * \left(dq^{2}\right) \qquad (2.2)$$

where n is number of intercepts per species and diameter size class, and dq^2 is the squared quadratic-mean diameter (cm^2). As no previous study focusing on CWD has been done in Quebec, we used specific multiplication factors Z from Ontario (McRae *et al*. 1979, Tables 2 and 3) for balsam fir, white birch, white spruce and trembling aspen, with the Z factor corresponding to:

$$Z = \frac{G * k * (dq)^{2} * a}{L} \qquad (2.3)$$

For white cedar, we used original quadratic-mean diameters and particle specific gravities in Minnesota from Roussopoulos (1978). As quadratic-mean diameters and specific gravities were provided for the American diameter size classes, we used simple linear interpolations to calculate the corresponding quadratic diameters and gravities in our CWD diameter size classes. We then calculated Z multiplication factors for white cedar (see footnote of Appendix 2.3) to compute the cedar log loads per diameter size class. Both Ontario and Minnesota are the nearest geographically and ecologically relevant data sources for our study area.

Snags were sampled within six one-meter radius circles centered on the point-centered quadrants. Snag species, height and diameter (at the top) were recorded (DBH if taller than 1.3 m). Densities, basal areas, and volumes of snags were calculated, but decay class was not recorded as all snags were in an intermediate decay class that no longer had bark.

2.4.3. Analyses

Canonical Correspondence Analysis

Canonical Correspondence Analysis (CCA) (Ter Braak 1987-1992) was used to analyze the distribution of the 48 stands according to species log loads and species snag basal areas. Basal area of living trees (dominant and dominated trees) are the active descriptive variables while time since fire was included in the analysis as a passive variable (Jongman *et al*. 1987) to evaluate the degree of association between this variable and the two first canonical axes. Further, Monte Carlo permutation tests (Ter Braak 1987-1992) were performed on the first two canonical axes to investigate the statistical significance of the impact variables on axes.

Covariance analysis

The pathway types found in the CCA have been analyzed through a covariance analysis (SAS Institute Inc. 1985). This analysis was conducted on total log loads with time since fire as the continuous variable, and the pathway type (birch or aspen) as a classification variable. Because a positive effect for both variables was found, the covariance analysis was extended to all CWD components. The 236-year-old coniferous stands were not included as they were the same for both pathways.

Effect of time since the last fire

Nonlinear regressions (SAS Institute Inc. 1985) were computed to study the potential effect of time since fire on log loads (total per stand, per diameter size classes, and per species) and on snag volumes (total per stand, and per species).

Effects of time since the last fire and stand composition.

We used multiple linear regressions with the forward stepwise procedure (SAS Institute Inc. 1985) to test the significant effect of time since fire and basal areas of living tree species on log and snag components. For each species we distinguished two canopy layers (BA and ba for basal area of dominant and dominated trees, respectively). This partitioning of the living basal areas is used to represent the canopy composition and its variability. If time is the only significant descriptor, it will then correspond to the real change in CWD accumulation with time. Conversely, if living basal areas are also descriptors of CWD accumulation changes, then changes are the result of canopy composition changes through the successional process.

2.5. RESULTS

2.5.1. CWD characteristics for the 48 stands sampled

Appendix 2.1 presents the characteristics of stands sampled for CWD components, while Appendixes 2.2 and 2.3 report on logs in terms of counts, diameters, and species composition. Density of living trees per stand varied from 437 stems/ha in a 236-year old white cedar stand to 3422 stems/ha in a pure 80-year old aspen stand. The aspen stand also had the largest living tree basal area (125.6 m^2/ha); the old white cedar stand had the lowest living tree basal area (15.1 m^2/ha). Total log load per stand varied from 17.8 to 111.5 tons/ha (51.1 tons/ha in average). Snag density varied from 0 to 7000 stems/ha, corresponding with 0 to 108 m^2/ha in basal area and from 0 to 700 m^3/ha in snag volume.

2.5.2. Analysis for multivariate potential factors

Results of the CCA, reported in Table 2.1 and Figure 2.1, show that 42.9% of the initial variance in the CWD data set can be explained by the basal area of living trees in the canopy (33.8% on the first two axes; see Table 2.1). Monte Carlo tests show that the first two canonical axes are good linear combinations of these descriptive variables. The upper graph shows the descriptive basal areas and time since fire as passive variable (Figure 2.1). On the main graph, birch and aspen stands are segregated according to the first axis and they present two successional pathways (the two arrow directions). These two patterns converge at the end of the succession (after 200 years since fire) towards coniferous stands. The upper graph shows that all trembling aspen, tall white birch and white cedar are the best represented variables defining the two axes. These living basal areas are well correlated with both the basal area of snags and the log loads of their respective species (see scores in Table 2.1). Finally, time since fire is well correlated with canonical axes and it is related positively with the dominating coniferous basal areas, and negatively with the trembling aspen basal areas and the dominated conifer basal areas (Table 2.1).

Table 2.1 Canonical Correspondance Analysis results for species log loads and snag basal areas in relation to basal areas of living canopy trees

	Axis 1	Axis 2	Axis 3	Axis 4	Total variance in the CWD a data
Eigenvalues	0.303	0.164	0.052	0.036	1.38
Variance in the CWD data					
% explained per axis	22	11.9	3.8	2.6	
cumulative % explained	22	33.9	37.7	40.3	42.90
Pearson correlation between CWD and basal area of the living trees	0.846	0.828	0.527	0.512	
Monte Carlo test on correlation (p)	0.01	0.01			
CWD scores					
Log load of balsam fir	-0.275	-0.362			
Log load of white birch	-0.283	0.71			
Log load of white spruce	-0.386	-0.305			
Log load of trembling aspen	0.857	0.092			
Log load of white cedar	-0.918	-0.13			
BA of balsam fir snags	-0.294	-0.197			
BA of white birch snags	-0.792	1.37			
BA of white spruce snags	-0.103	-1.181			
BA of trembling aspen snags	0.888	-0.211			
Correlation of time since fire with axes	-0.672	-0.596			

[a] : CWD for Coarse Woody Debris

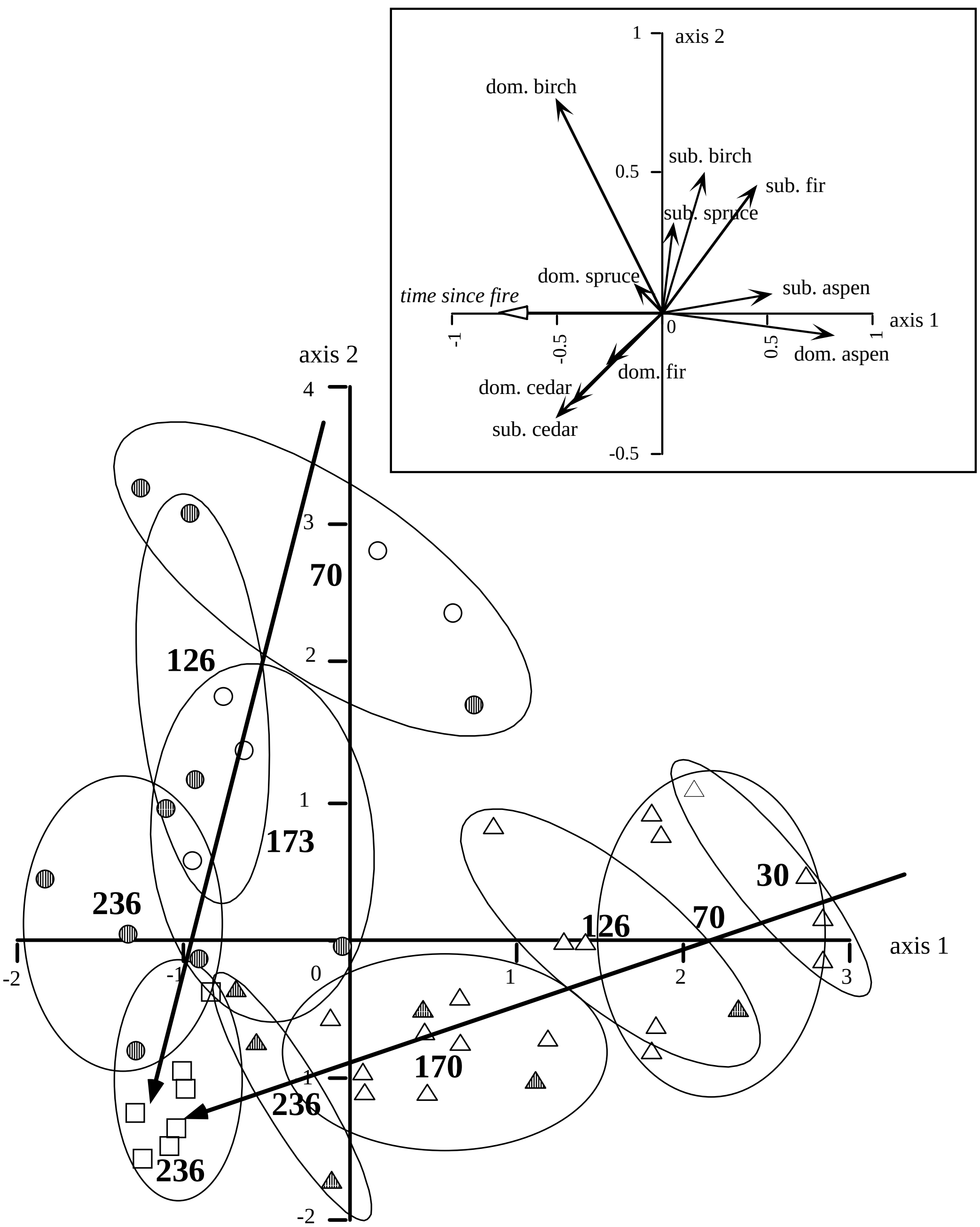

Figure 2.1 Canonical Correspondence Analysis on Coarse woody debris characteristics (species log loads and snag basal areas) in relation to basal areas of living canopy species.
Arrows on the upper graph represent vectors for living basal areas of species (dom. for dominant trees and sub. for sub-dominant trees). On the main graph the numbers in the ellipses are the mean stand age. The symbols of stand scores are : Δ = aspen stands, O = birch stands, and □ = for conifer stands. Shaded symbols represent mixed stands (based on coniferous basal area between 25 and 74% of total stand). Arrow directions on the main graph symbolize the successional pathway trends.

The covariance analysis on total log load (Figure 2.2) shows that birch and aspen stands have the same slope parameter, but significantly different intercepts. Log loads in birch stands tend to be heavier than in aspen stands. This analysis supports the existence of two successional pathways shown by the CCA. However, covariance analyses carried out on each CWD component showed (not presented) that time since fire was the most significant factor. For these reasons, we have chosen to only consider time since fire as a significant factor, and we have analyzed the 48 stands together.

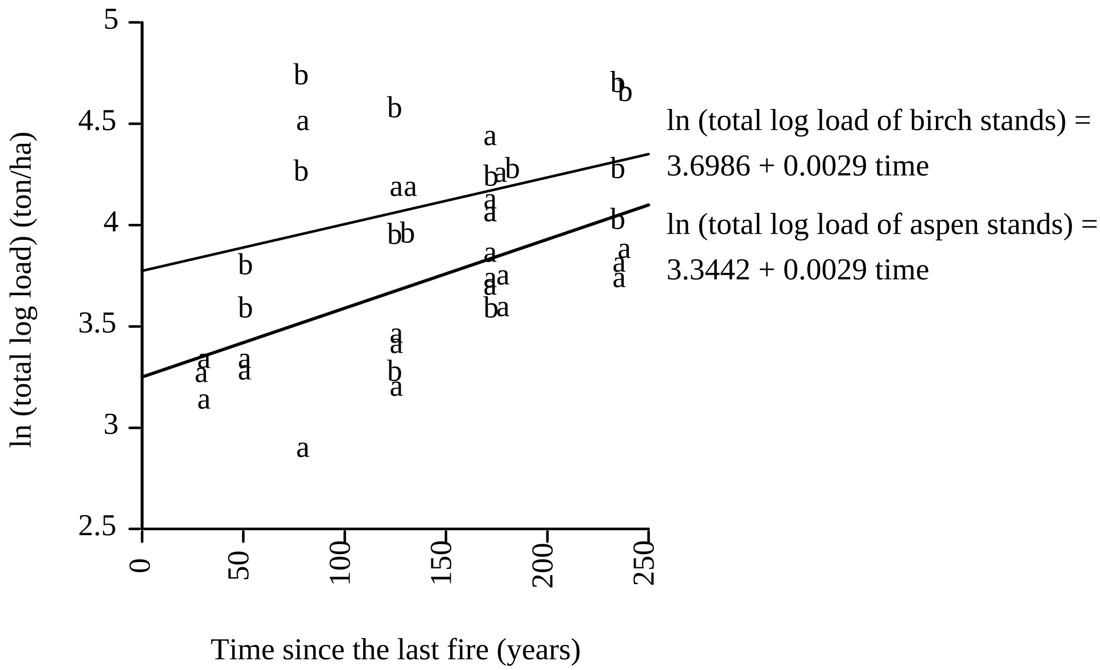

Figure 2.2 Results of the covariance analysis conducted on the natural logarithms of total log loads with time since the last fire as continuous variable and successional pathway (a for trembling aspen and b for white birch) as classification variable.

2.5.3. Effects of time since the last fire

The first five diameter size classes of logs present the same general trend in relation to time since fire (Figure 2.3), with a U-shape curve prolonged by a plateau or a decrease in the very late successional stages. All these fine debris levels tend to be variable but quite high 30 years following the fire. Log loads decrease with stand age to a low at 80-120 years. An accumulation period follows then for a 100-year period, and then plateaus or decreases again at late stages. However, the cumulated five class loads represent on average 43% of total log load (standard deviation = 18%). The class VI load presents a significantly different pattern through time since fire (Figure 2.3): coarse debris levels are the lowest 30 years after fire, but they increase up to 100 years. In stands older than 120 years, log loads tend to slightly decrease before increasing again in late stages. These two different patterns among log class loads associated to the proportions of fine and coarse debris loads imply that total log loads (Figure 2.4) fit a different pattern from the above ones, with a general load increasing with time. Among the species, only balsam fir and trembling aspen log loads had significant trends through time (Figure 2.4). Aspen log loads decrease with time whereas fir log loads increase through time. White birch log loads remain high over the entire period, while white spruce log loads are consistently the lowest. White cedar log loads increase significantly in stands older than 200 years.

Among snags, the only significant relationships between volumes and time since fire are for the total snag, and balsam fir snag volumes (Figure 2.5). In both cases, volumes increase with time. Volumes of trembling aspen snags present a U-shaped pattern from 30 to 170 years, and then disappear after 200 years. Some birch snags are present in young and very old stands and show no particular trend. White spruce and white cedar did not have a sufficient number of stands with snags.

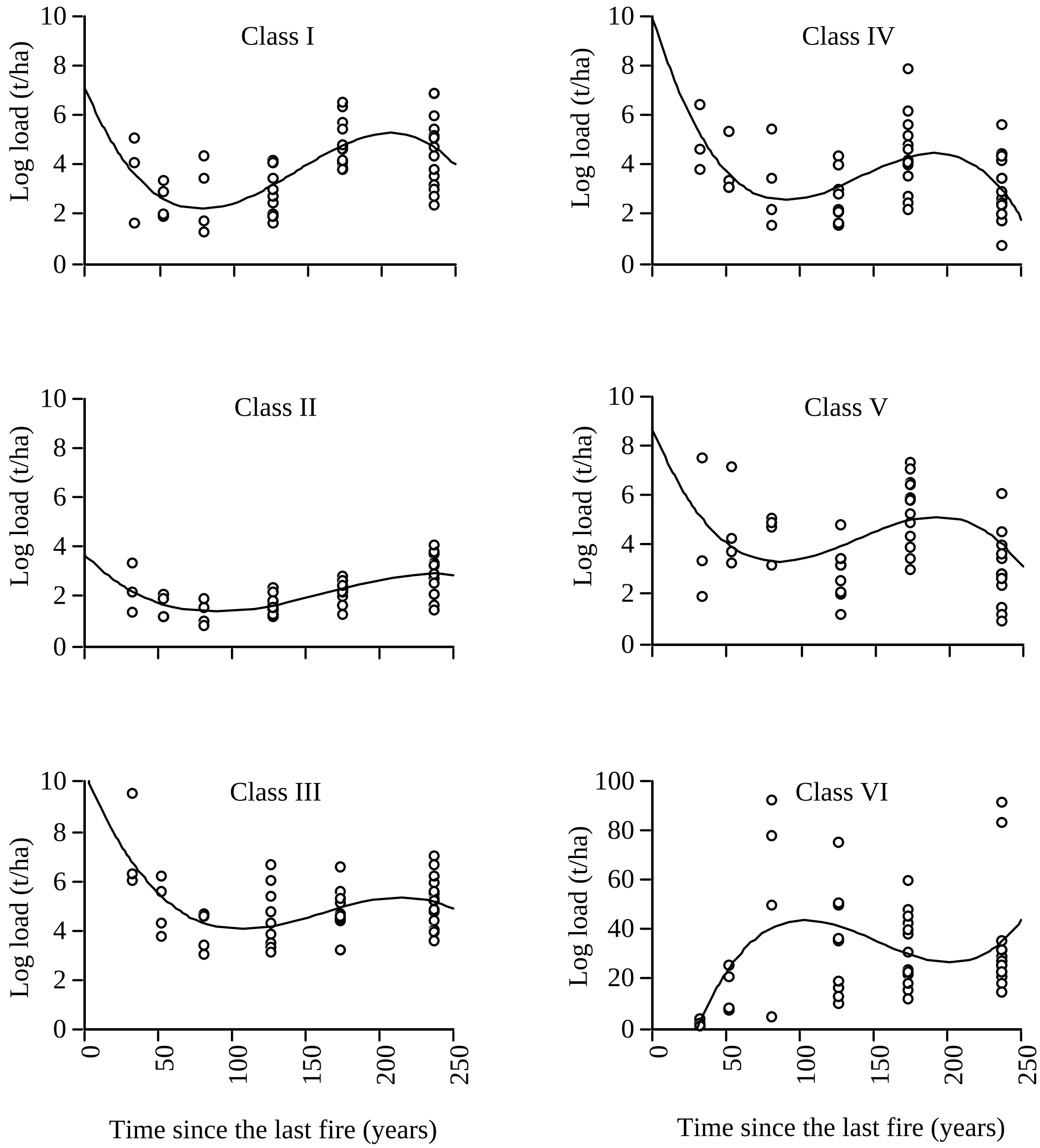

Figure 2.3 Changes in log loads by diameter size class with time since the last fire n = 48; critical $R^2_{(47\,;\,0.05)} = 0.081$. Equations for log load through time are:

ClassI = $-0.000003 * t^3 + 0.0026 * t^2 - 0.1442 * t + 7.1656$ ($R^2 = 0.387$; $p = 0.0001$)
ClassII = $-0.000001 * t^3 + 0.0005 * t^2 - 0.0598 * t + 3.7074$ ($R^2 = 0.484$; $p = 0.0001$)
ClassIII = $-0.000002 * t^3 + 0.0010 * t^2 - 0.1444 * t + 10.5044$ ($R^2 = 0.271$; $p = 0.0028$)
ClassIV = $-0.000004 * t^3 + 0.0016 * t^2 - 0.1969 * t + 10.1109$ ($R^2 = 0.245$; $p = 0.0058$)
ClassV = $-0.000004 * t^3 + 0.0016 * t^2 - 0.1893 * t + 9.8683$ ($R^2 = 0.219$; $p = 0.0119$)
ClassVI = $-0.000037 * t^3 - 0.0170 * t^2 + 2.3330 * t$ ($R^2 = 0.175$; $p = 0.0360$)

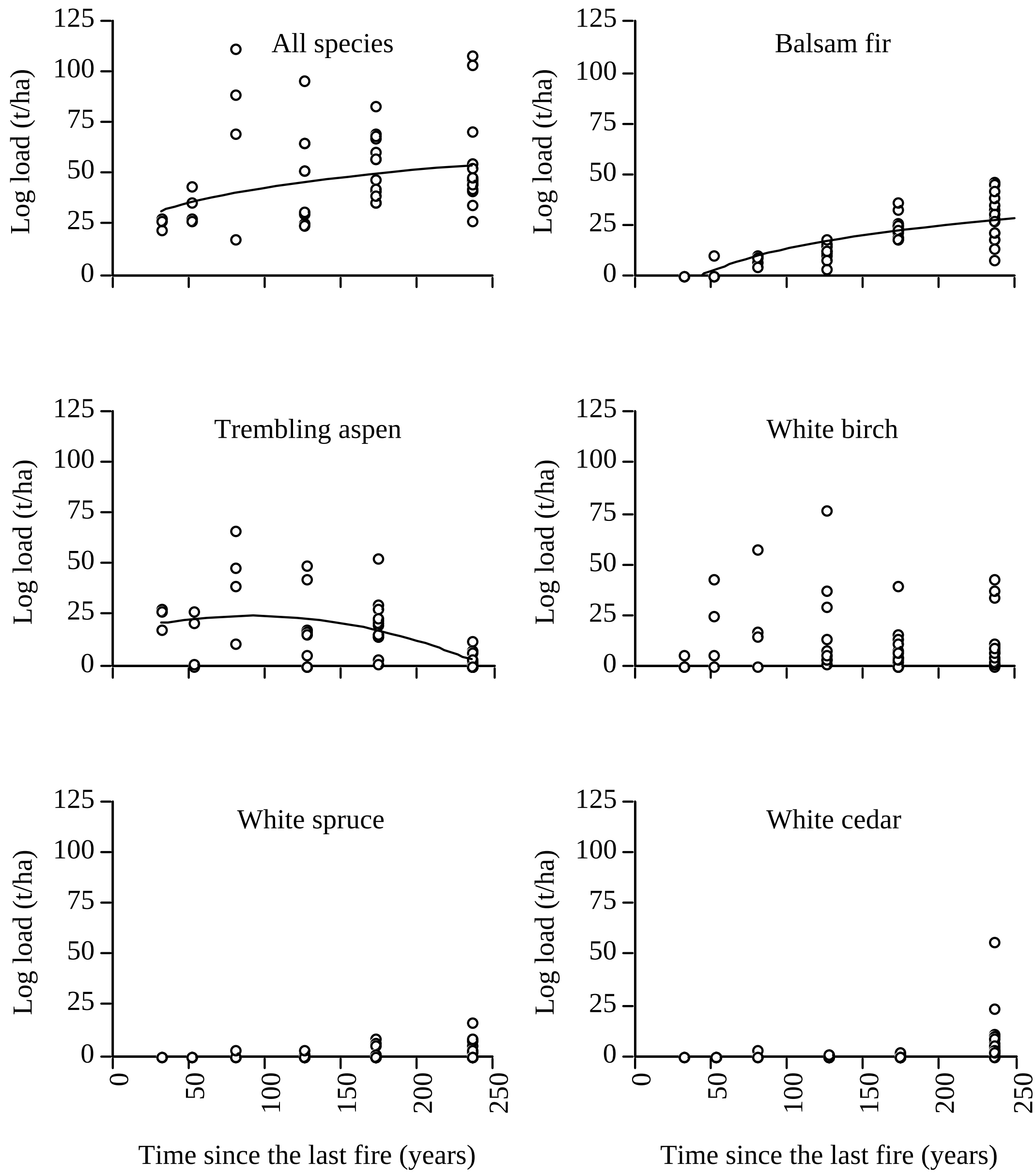

Figure 2.4 Changes in the total and species log loads with time since the last fire. n = 48; critical $R^2_{(47\,;\,0.05)} = 0.081$. Residual homoscedasticity was not accepted for white birch, white spruce, and white cedar log loads. Equations for log load through time are:

All species	$y = 11.96 * t^{0.276}$	($R^2 = 0.139$; $p = 0.0091$)
Balsam fir	$y = 0.2725 * t - 2.565 * \mathrm{sqrt}(t) + 6.0221$	($R^2 = 0.636$; $p = 0.0001$)
Trembling aspen	$y = 0.0003 * t^{2} - 0.2080 * t + 33.3853$	($R^2 = 0.241$; $p = 0.0004$)

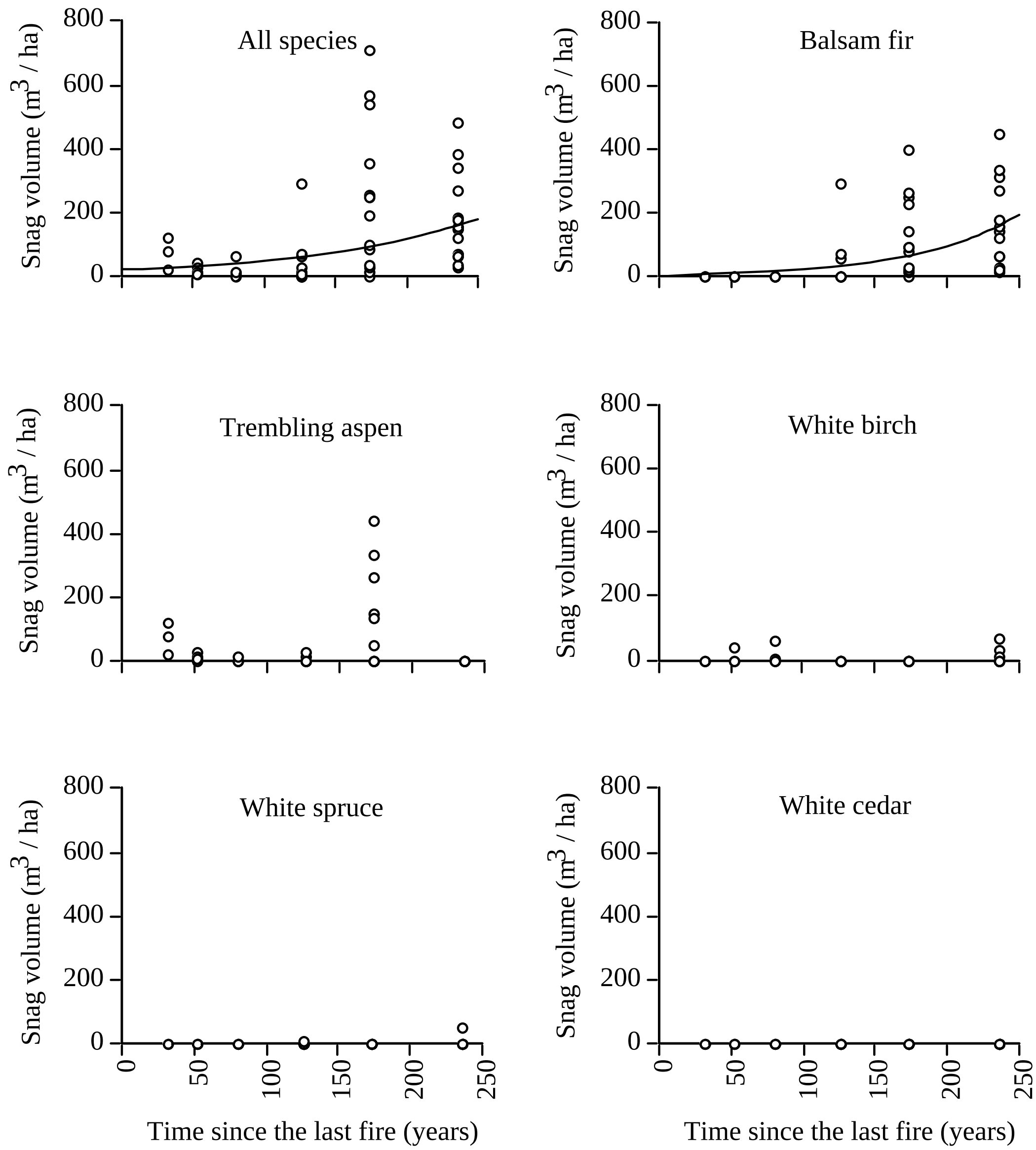

Figure 2.5 Changes in snag characteristics with time since the last fire.
n = 48 when not mentioned; critical $R^2_{(47\ ;\ 0.05)} = 0.081$. Residual homoscedasticity was not accepted for trembling aspen, white birch, white spruce, and white cedar snag volumes. Equations for snag volume through time are:

All species	$y = 20.184 * 10^{0.004*t}$	($R^2 = 0.253$; p = 0.0006 ; n = 43)
Balsam fir	$y = 0.004 * t^2 - 0.419 * t + 9.778$	($R^2 = 0.461$; p = 0.0001)

2.5.4. Effects of time since the last fire and stand composition

Results for logs and snags (Tables 2.2 and 2.3, respectively) show different behaviours of CWD in relation to time since fire and canopy basal areas. Log loads of classes V and VI, white spruce, and white cedar are not reported in Table 2.2 as they did not show significant linear relationships with time or basal areas. However, basal areas and time since fire are significant descriptors in four of the eight significant models (Table 2.2). Basal areas are the only significant descriptors in three models, whereas time since fire is the only significant descriptor for the class I log load. This implies that changes in the log accumulation (total and species log loads) are influenced both by the effect of time since fire and by the change in canopy species dominance, reflecting successional process. Moreover, the species composition effect is as important as time (see Table 2 for outcome order entries from the forward stepwise selection).

Conversely, time is always the first variable selected for snags (Table 2.3), with four cases where it is the only significant variable. Even in models where basal areas are part of significant descriptors, only one species basal area is included. This implies that changes in snag components (species composition, density, basal area and volume) are almost only influenced by time since fire.

Table 2.2 Significant multiple regressions for the log load components

Variable	Total	Class I	Class II	Class III	Class IV	ln(fir)	sqrt(birch)	sqrt(aspen)
p	0.0004	0.0003	0.0001	0.0010	0.0280	0.0001	0.0001	0.0001
R multiple	0.5430	0.5030	0.6850	0.5140	0.3170	0.8820	0.7078	0.8370
R^2	0.2640	0.2360	0.4690	0.2320	0.1010	0.7770	0.5010	0.7010
intercept	0.0000	2.2370	1.6110	5.2040	3.3691	0.0000	1.3071	3.6940
time	0.1303 [2]	0.0110 [1]	0.0061 [1]			0.0151 [1]		-0.0099 [2]
BA fir								
BA birch							0.2103 [1]	-0.1267 [4]
BA spruce			-0.0827 [2]					
BA aspen				-0.0197 [2]				0.0413 [1]
BA cedar								
ba fir	5.3814 [1]		-0.0869 [3]			0.1674 [2]		0.3716 [3]
ba birch							0.4195 [2]	
ba spruce								
ba aspen				0.1875 [1]	0.1608 [1]			
ba cedar								

Note: BA = basal area of dominant trees ; ba = basal area of sub-dominant trees. Only the significant parameters are given; p is the overall model probability; small numbers refer to entering order.

Table 2.3 Significant multiple regressions for the snag components

Variable	sqrt(stand vol)	sqrt(stand BA)	% conifer snags	ln(fir dens.)	ln(fir BA)	sqrt(fir vol.)	ln(aspen dens.)
p	0.0015	0.0068	0.0001	0.0001	0.0001	0.0001	0.0001
R multiple	0.4447	0.3858	0.6162	0.6756	0.7080	0.7114	0.6940
R^2	0.1977	0.1489	0.3797	0.4564	0.5018	0.5061	0.4816
intercept	0.0000	1.8548	0.0000	0.0000	0.0000	-4.297	7.8627
time	0.0445 ^{1}	0.0136 ^{1}	0.3005 ^{1}	0.0333 ^{1}	0.0156 ^{1}	0.7990 ^{1}	-0.0259 ^{1}
BA fir							
BA birch			-2.3327 ^{2}				-0.2468 ^{2}
BA spruce							
BA aspen							
BA cedar							
ba fir							
ba birch							
ba spruce							
ba aspen							
ba cedar						-0.5436 ^{2}	

Note: BA = basal area of dominant trees ; ba = basal area of sub-dominant trees ; vol = volume (m^3 / ha) ; dens = density (stems / ha).
Only the significant parameters are given ; p is the overall model probability ; small numbers refer to entering order.

2.6. DISCUSSION

2.6.1. Comparisons of CWD characteristics with other boreal ecosystems

The presented snag components (mean volume = 140.5 m^3/ha, mean density = 1570 stems/ha) are at the upper limit of ranges found to date in boreal forests (Lee *et al*. 1997; Linder *et al*. 1997; Schimmel and Granström 1997; Sturtevant *et al*. 1997). If we only focus on snags larger than 10 cm in DBH, the mean snag volume per stand increases from 140 to 143 m^3/ha, corresponding with a slight decrease of mean snag basal area from 22 to 19 m^2/ha and a mean density dropping from 1570 to 830 snags/ha. These values fit better with those of other studies. Moreover, snag density and basal area may reflect some processes such as self-thinning and natural disturbance effects. Indeed, the maximum density of 6900 snags/ha (all diameters included) was recorded in the youngest aspen stands, where trees are undergoing self-thinning (snag basal area is only 17 m^2/ha). However, only one of these stands recorded a snag larger than 10 cm in DBH. In mixed and conifer stands, snag densities higher than 3000 stems/ha represent mostly balsam firs killed during the last and most severe spruce budworm outbreak (Bergeron *et al*. 1995; Morin *et al*. 1993). Insects may have also fed on sub-dominant trees (from 5 to 10 cm in DBH) during the maximum defoliation period.

Comparisons between log characteristics found in our study area (mean log load = 51 t/ha) and other boreal ecosystems showed (Hély *et al*. n.d.) differences with deciduous stands in Alberta (109-124 m^3/ha, Lee *et al*. 1997) and mixedwood boreal stands in New Brunswick (7 – 20 t/ha, Freedman *et al*. 1996). It is likely that the drier climate in Alberta creates a less favourable environment for decomposer fungi. This would slow down bole decay rate and result in a greater log accumulation. Conversely, in New Brunswick, the moister climate has inverse effects, and it favours log decay. Moreover, Freedman *et al*. (1996), by sampling only logs greater than 5 cm in diameter, have not taken into account small diameter log loads that comprise up to 43% of the total load in our study area.

2.6.2. Stand composition and time since the last fire as descriptors of the change in the CWD accumulation

The analyses have shown that time since fire and stand canopy composition are two significant descriptive factors in CWD changes. However, while logs are influenced by both factors, snags are mainly influenced by time since fire.

The high correlation between species log loads and living basal areas show that logs reflect the stand dynamics through the successional process: the dominant deciduous species die and deciduous stands are replaced by mixed and conifer stands in later successional stages (Bergeron and Dubuc 1989). Other studies have recorded poor correlation between the stand basal area and log composition (Muller and Liu 1991; Sturtevant *et al*. 1997), but species composition of CWD reflected the species composition of the standing forest (Muller and Liu 1991). In our study, log loads respond almost immediately to the canopy species replacement by a rapid change in their species composition. This rapid change can be explained by the rapid replacement of the dominant canopy species that is also the largest log producer at that time. Paré and Bergeron (1995) have shown that total aboveground biomass closely corresponds to the trembling aspen biomass, and when this species is replaced by conifer species, the total aboveground biomass decreases in the same way. Moreover, the involved species are known to have rapid decay rates (Alban and Pastor 1993; Lambert *et al*. 1980). The log load at a given time corresponds closely to the present load (input and decay likely in equilibrium) and also to the actual canopy composition. It seems that there is no long-term accumulation process.

Conversely, the snags component changes are, when they exist, mainly influenced by time since fire, and to a lesser extent by the canopy composition changes. Snags, because they are standing, are drier and less exposed to decomposers, would have a slower decay rate than logs. Thus snags can then remain standing for a long time, even though their species no longer dominates the living stand canopy. This explains why snag composition does not fit the actual living tree canopy composition, but that it is instead related to stand age and to its history. Snags reflect the transition from deciduous towards conifer dominance occurring during succession (Bergeron and Dubuc 1989); the percentage of conifer snags (mainly represented by balsam fir) increases with time, whereas trembling aspen, which is the most prolific deciduous snag producer, has a snag density that varies with time. After fire, shade-intolerant species such as trembling aspen dominate the stand canopy and the snag compartment from self-thinning. From 50 to 120 years after fire, snag volume is low but still dominated by deciduous species. After 150 years, overmature aspen die and constitute high snag inputs. Conifer species, mostly balsam fir, finally replace deciduous species in the canopy, and produce at that time, the largest snag volumes particularly from spruce budworm outbreaks. These intrinsic and extrinsic events created, within a short time period, large amounts of snags, standing for a long time after the disturbance event. Moreover, we also have to consider the relationship between the timing of the sampling period and spruce budworm outbreak events. If CWD were sampled 25 to 30 years ago, the last spruce budworm outbreak would not have occurred yet, and the balsam fir CWD (logs and snags) would have been less important. Additionally, if we were to sample CWD 20 years from now with no spruce budworm outbreaks occurring, the balsam fir snags would have fallen and would therefore be less important.

2.6.3. CWD accumulation models in the southern boreal forest

Coarse woody debris accumulation models in the mixedwood boreal forest (Figures 2.3, 2.4, and 2.5) are different from other CWD accumulation models (Agee and Huff 1987; Bormann and Likens 1994; Harmon *et al*. 1986; Spies *et al*. 1988; Sturtevant *et al*. 1997). As we have seen above, the key factors explaining these differences (for total or species loads) are the species replacement occurring during succession (Bergeron and Dubuc 1989), differences in species productivity (Paré and Bergeron 1995), rapid decay rates (Lambert *et al*. 1980, Harmon *et al*. 1986), and disturbances such as cyclic spruce budworm outbreaks killing mature balsam firs and favouring white birch and/or balsam fir regeneration (Kneeshaw and Bergeron 1998). The CWD models in our study do not show the high initial post-fire CWD load resulting from the fire event, as in the traditional U-shaped model (Agee and Huff 1987; Bormann and Likens 1994; Spies *et al*. 1988; Sturtevant *et al*. 1997). The initial load at 30 years is composed of small diameter pieces (that would not have withstood the fire event), and not from pre-fire or fire residual debris. The absence of large pre-fire CWD can be explained by the faster decay rate of burned pieces a few years after fire (Harmon *et al*. 1986), and by the absence of large diameter trees in pre-fire stands. Indeed, even in old stands, conifers that dominate stands are much smaller trees and snag producers than deciduous species (Paré and Bergeron 1995). The log pattern of trembling aspen through time found and discussed in our study shows the same trend as the aboveground biomass found by Paré and Bergeron (1995) in the same study area, and also as the trembling aspen snags in western Canada found by Lee *et al*. (1997). This pattern is characterized by a first load increase until stand maturity, followed by a period of load decrease during the late successional stages. It resembles the aboveground living biomass of forest stands presented by Peet (1981), who assigned this accumulation pattern to boreal stands such as feathermoss-black spruce stands with very thick organic layer. In our case, the period of load decrease is not a decrease in the successive cohort productivity due to less favourable abiotic conditions, but to the species replacement with a resulting decrease in the abundance of trembling aspen (Bergeron and Dubuc 1989; Paré and Bergeron 1995). Aspen and birch have no difference in decay rates, but white birch log accumulation is significantly higher than trembling aspen (same trend in Figure 2.2). Moreover, the birch trend through time is more variable (non significant trend in Figure 2.4) than for aspen logs. This variability difference may come from the distinct life history patterns of these species: white birch is a long-lived species with one postfire cohort that can stand more than 200 years (Frelich and Lorimer 1991; Dansereau and Bergeron 1993) and create logs and snags along the entire succession, whereas the mean life span of trembling aspen is 80 years in eastern North America (Hosie 1979); aspen regenerates through successive cohorts over a comparable period. Thus aspen produces high episodic amounts of CWD (Figure 2.5 for snags). The birch pattern variability could also be explained by periodic openings of stand canopy from balsam fir death during spruce budworm outbreaks. Large gaps are favourable to white birch regeneration (Kneeshaw and Bergeron 1998). For white cedar, this study would have required a longer time window to record a significant trend, as this shade-tolerant species only begins to regenerate in the late stages (Bergeron and Dubuc 1989). A recently discovered mesic forest dominated by white cedar, and regenerated at least 400 years ago after a fire, had a total log load of 29 tons/ha. Balsam fir, white cedar, and white birch were responsible for 18, 8, and 3 tons/ha, respectively. This suggests that mixed stands can be maintained for a long time (high variability of old stand composition also shown by the CCA), and that high balsam fir log load was probably created by the last spruce budworm outbreak. However, because of the longevity of white cedar, it produces only small log loads even in 236-year old stands. White spruce seems the least active species in the successional replacement process, with a low abundance of living trees and few logs and snags. This could explain why there was no significant trend over time for this species. Finally, balsam fir accumulates CWD during the entire succession. The trend seems to be dependent on both disturbance and decay rate. Because the outbreak periods seem to be as long as the decay rate, the log pattern should stay stable over the long-term, until another fire occurs.

2.7. ACKNOWLEDGMENTS

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2.8. REFERENCES

Agee, J.K., and Huff, M.H. 1987. Fuel succession in a western hemlock/Douglas-fir forest. Can. J. For. Res. **17**: 697 - 704.

Alban, D.H., and Pastor, J. 1993. Decomposition of aspen, spruce, and pine boles on two sites in Minnesota. Can. J. For. Res. **23**: 1744 - 1749.

Bergeron, Y. 1991. The influence of island and mainland lakeshore landscapes on boreal forest fire regimes. Ecology **72**: 1980 - 1992.

Bergeron, Y. and Dubuc, M. 1989. Succession in the southern part of the Canadian boreal forest. Vegetation **79**: 51 -63.

Bergeron, Y., Leduc, A., Morin, H., and Joyal, C. 1995. Balsam fir mortality following the last spruce budworm outbreak in northwestern Quebec. Can. J. For. Res. **25**: 1375-1384.

Bormann, F.H., and Likens, G.E. 1994. Pattern and process in a forested ecosystem. Springer - Verlag, New York : 124p.

Brown, J.K. 1974. Handbook for inventorying downed woody material. U.S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT 16 : 24p.

Dansereau, P., and Bergeron, Y. 1993. Fire history in the southern boreal forest of northern Québec. Can. J. For. Res. **23**: 25 - 32.

De Grandpré, L., Gagnon, D., and Bergeron, Y. 1993. Changes in the understory of Canadian southern boreal forest after fire. J. Veg. Sci. **4**: 803-810

Environment Canada 1993. Canadian climate normals 1961-90. Can. Clim. Prog. Environ. Can. Atmos. Environ. Serv. Downsview, Ontario.

Freedman, B., Zelazny, V., Beaudette, D., Fleming, T., Flemming, S., Forbes, G., Gerrow, J.S., Johnson, G., and Woodley, S. 1996. Biodiversity implications of changes in the quantity of dead organic matter in managed forests. Environ. Rev. **4**: 238-265.

Frelich, L.E., and Lorimer, C.G. 1991. Natural disturbance regimes in hemlock-hardwood forests of the Upper Great Lakes region. Ecol. Monogr. **61**: 145-164.

Harmon, M.E., Franklin, J.F., Swanson, F.J., Sollins, P., Gregory, S.V., Lattin, J.D., Anderson, N.H., Cline, S.P., Aumen, N.G., Sedell, J.R., Lienkaemper, G.W., Cromack, K., Jr., and Cummins, K.W. 1986. Ecology of coarse woody debris in temperate ecosystems. *In* Advances in ecological research. Vol. 15. Edited by A. MacFadyen and E. D. Ford. Academic Press, London: 133 - 302.

Harmon, M.E., Cromack, H., Jr. and Smith, B. 1987. Coarse woody debris in mixed-conifer forest, Sequoia National Park, California. Can. J. For. Res. **17**: 1265 - 1272.

Hély, C., Bergeron, Y. and Flannigan, M.D. n.d. Fire susceptibility in the southeastern Canadian boreal forest: effects of stand composition. J. Veg. Sci. (submitted).

Hosie, R.C. 1979. Native trees of Canada. Eighth edition. Fitzhenry and Whiteside Ltd., Toronto, 380p.

Jongman, R.H.G., Ter Braak, C.J.F., and Can Tongeren, O.F.R. 1987. Data analysis in community and landscape ecology. Rodoc, Wageningen.

Keddy, P.A. and Drummond, C.G. 1996. Ecological properties for the evaluation, management, and restoration of temperate deciduous forest ecosystems. Ecol. Appl. **6**: 748 - 762.

Kneeshaw, D. and Bergeron, Y. 1998. Canopy gap characteristics and tree replacement in the southeastern boreal forest. Ecology **79**: 783-794.

Lambert, R.L., Lang, G.E., and Reiners, W.A. 1980. Loss of mass and chemical change in decaying boles of a subalpine balsam fir forest. Ecology **61**: 1460 - 1473.

Lee, P.C., Crites, S., Nietfeld, M., Van Nguyen, H., and Stelfox, J.B. 1997. Characteristics and origins of deadwood material in aspen-dominated boreal forests. Ecol. Appl. **7**: 691 - 701.

Linder, P., Elfving, B., and Zackrisson, O. 1997. Stand structure and successional trends in virgin boreal forest reserves in Sweden. For. Ecol. Manage. **98**: 17 - 33.

McRae, D.J., Alexander, M.E., and Stocks, B.J. 1979. Measurement and description of fuels and fire behaviour on prescribed burns: a handbook. Can. For. Serv., Report O-X-287.

Morin, H., Laprise, D., and Bergeron, Y. 1993. Chronology of spruce budworm outbreaks near Lake Duparquet, Abitibi region, Quebec. Can. J. For. Res. **23**: 1497-1506.

Mueller-Dombois, D. and Ellenberg, H. 1974. Aims and methods of vegetation ecology. Wiley, New York.

Muller, R.N., and Liu, Y. 1991. Coarse woody debris in an old-growth deciduous forest on the Cumberland plateau, southeastern Kentucky. Can. J. For. Res. **21**: 1567-1572.

Paré, D., and Bergeron, Y. 1995. Above-ground biomass accumulation along a 230-year chronosequence in the southern portion of the Canadian boreal forest. J. Ecol. **83**: 1001-1007.

Payette, S. 1992. Fire as controlling process in the Northern American boreal forest. *In* A systems analysis of the global boreal forest. Edited by H.H. Shugart, R. Leemans, and G.B. Bonan. Cambridge Univ. Press, New York: 144-169.

Peet, R.K. 1981. Changes in biomass and production during secondary forest succession. *In* Forest succession : concepts and application. Edited by D. C. West, H. H. Shugart, and D. B. Botkin. Springer-Verlag : 325 - 338.

Roussopoulos, P.J. 1978. An appraisal of upland forest fuels and potential fire behavior for a portion of the boundary waters canoe area. Ph.D. Thesis, Michigan State University, Department of Forestry : 166p.

Rowe, J.S. 1972. Forest regions of Canada. Environment Canada, Ottawa, Ontario. Public. 1300.

SAS Institute Inc. 1985. SAS user's guide: statistics, version 6 edition. SAS Institute Inc., Cary, N.C.

Schimmel, J., and Granström, A. 1997. Fuel succession and fire behavior in the Swedish boreal forest. Can. J. For. Res. **27**: 1207-1216.

Spies, T.A., Franklin, J.F., and Thomas, T.B. 1988. Coarse woody debris in Douglas-fir forests of western Oregon and Washington. Ecology **69**: 1689-1702.

Stauffer, D.F., and Best, L.B. 1980. Habitat selection by birds of riparian communities: evaluating effect of habitat alteration. J. Wildlife Manage. **44**: 1-15.

Sturtevant, B.R., Bissonette, J.A., Long, J.M. and Roberts, D.W. 1997. Coarse woody debris as a function of age, stand structure, and disturbance in boreal Newfoundland. Ecol. Appl. **7**: 702-712.

Ter Braak, C.J.F. 1987-1992. CANOCO - a FORTRAN program for Canonical Community Ordination. Microcomputer Power, Ithaca, New York.

Tyrrell, L.E., and Crow, T.R. 1994. Dynamics of dead wood in old-growth hemlock-hardwood forests of northern Wisconsin and northern Michigan. Can. J. For. Res. **24**: 1672-1683.

Van Wagner, C.E. 1968. The line intersect method in forest fuel sampling. For. Sci. **14**: 20-26.

Van Wagner, C.E. 1980. Practical aspects of the line intersect method. Can. For. Serv. Inf. Rep. PI-X-12.

Vincent, J.S., and Hardy, L. 1977. L'évolution et l'extension des lacs glaciaires Barlow et Ojibway en territoire québecois. Géographie Physique et Quaternaire **31**: 357-372.

CHAPITRE III

EFFECTS OF STAND COMPOSITION ON FIRE HAZARD IN THE MIXEDWOOD CANADIAN BOREAL FOREST

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3.1. RÉSUMÉ

Les combustibles de surface ont été analysés au sein de 48 peuplements de la forêt boréale mixte Canadienne, regroupés en quatre types de peuplements. La canopée a été caractérisée selon la méthode du quadrant centré sur le point, et les peuplements ont alors été caractérisés comme feuillus, mixtes-feuillus, mixtes-conifères, ou conifères selon le pourcentage de leur surface terrière coniférienne. Les charges de débris ligneux ont été estimées d'après la ligne d'intersection, et les charges et épaisseurs de la litière, de l'humus, et des arbustes ont été calculées sur la base d'échantillons récoltés dans différents quadrats. Les résultats montrent qu'il n'y a pas de différence significative entre les quatre types de peuplements concernant les charges totales des débris ligneux, les charges des arbustes de larges diamètres, ni la charge ou l'épaisseur de la litière ou de l'humus. Cependant, les peuplements conifères ont des charges significativement plus importantes d'élements fins (branches et arbustes), et ils présentent aussi plus de particules conifériennes au sol que les peuplements feuillus. Ces changements qualitatifs et quantitatifs, résultant du remplacement des espèces dominantes au cours de la succession et des taux de décompostion rapide, ont des répercussions sur la susceptibilité au feu. Le système de prédiction BEHAVE a été utilisé pour évaluer l'impact de ces différences sur le comportement du feu potentiel pour des situations où la topographie et le climat ont été fixés constants. Les simulations du comportement du feu montrent qu'en forêt boréale mixte l'intensité du feu dans les peuplements feuillus est moins importante et le feu se propage plus lentement que dans les peuplements mixtes ou conifères. De plus, les feux de printemps sont plus intenses que ceux d'été, et les différences entre les saisons augmentent avec la proportion de surface terrière des feuillus.

3.2. ABSTRACT

Surface fuels were examined according to four stand types in 48 stands of the Canadian mixedwood boreal forest. Tree canopy was characterized with the point centered quadrant method and stands were characterized as either being deciduous, mixed-deciduous, mixed-coniferous, or coniferous stands according to their percentage of conifer basal area. Woody debris loadings were measured with the line intersect method, and the litter, duff, and shrub loads and depths or heights were sampled with various quadrats. No significant difference was found among stand types for total woody debris load, large basal diameter shrub loads, and load or depth of litter and duff. However, conifer stands had significantly heavier loads of small diameter elements (twigs and shrubs), and conifer pieces were more numerous within these stands than in deciduous stands. These qualitative and quantitative changes, resulting from species replacement and fast decay rates, had impacts on fire hazard. The BEHAVE prediction system was used to evaluate the impact of these differences on the potential of fire ignition and surface fire propagation (early fire behaviour) in situations where topography and weather were held constant. Simulations of fire behaviour show that in the mixedwood boreal forest fire intensity in deciduous stands was less high and the fire spreads slower than in mixed or coniferous stands. Moreover, spring fires were more intense than summer fires, and differences between seasons increased with the increase of deciduous basal area.

3.3. INTRODUCTION

Wildfires are one of the most important disturbances in the boreal forest (Heinselman 1981; Wein & McLean 1983; Johnson 1992; Engelmark et al. 1993) shaping forest mosaics of patches, stand age, and stand composition (Shugart et al. 1992; Bergeron & Dansereau 1993; Turner & Romme 1994). At the landscape level, the mosaic of the southeastern Canadian boreal forest is composed of deciduous, mixed, and conifer stands of different stages of the post-fire secondary succession (Bergeron & Dubuc 1989; Bergeron & Dansereau 1993). Indeed, numerous studies have demonstrated within different regions of the boreal forest that these successional changes are in fact examples of species dominance replacement in the canopy (Carleton & Maycock 1978; Wein & McLean 1983; Bergeron 2000). In young mixedwood boreal forest stands, shade-intolerant species such as trembling aspen (*Populus tremuloides* Michx.) dominate. With time, these stands develop into mixed stands as more shade-tolerant conifers eventually replace deciduous species. Stand conversion to conifers occurs gradually if the inter-fire period is sufficiently long. If the fire interval is short, shade-intolerent species will always dominate. Recent research in the southeastern Canadian boreal forest suggests that the fire return interval is being extended (Bergeron 1991; Johnson 1992; Bergeron & Archambault 1993; Flannigan et al. 1998). If so, an increasingly larger proportion of the forest will attain old-growth status with balsam fir (*Abies balsamea* (L.) Mill.) and white cedar (*Thuja occidentalis* L.) as the main species (Bergeron & Dubuc 1989).

Most of the crown fires start from surface fires where surface fuels release a head fire intensity greater than the critical surface intensity needed to initiate crowning (Van Wagner 1977). This study will then focus on surface fire behaviour with woody debris (i.e. twigs and branches less than 7.6 cm in diameter), litter layer, dead herbs and small basal diameter shrubs layers constituting fuels for fire ignition and surface fire propagation (Johnson 1992). Previous studies, conducted within different boreal forest regions, have demonstrated that fuel changes are responsible for different fire behaviours (Taylor & Fonda 1989; Schimmel & Granström 1997). However, because no recent wildfires or experimental burns have been measured in the mixedwood boreal forest, an alternative approach to assess fire hazard and initial surface fire behaviour is to evaluate the surface fuel loads for fire hazard and then, to input these fuel components into a fire behaviour prediction system that models the surface fire behaviour. The BEHAVE prediction system (Burgan & Rothermel 1984; Andrews 1986; Andrews & Chase 1989) has been selected to conduct this study because it predicts surface fire behaviour from the stand characteristics and the different fuel type loads collected within the stand, whereas a fire behaviour prediction system such as the Canadian Fire Behaviour Prediction (FBP) system would only require the general stand type description (Forestry Canada Fire Danger Group 1992; Hirsch 1996).

The hypothesis is that stands of varying composition (deciduous, mixed, and coniferous) have differences in loads and quality of surface fuels. This variability can then be related to stand fire hazard defined in the present study in terms of the characteristics (chemical and physical) of fuel elements that would favour flame propagation if ignition occurred (Montgomery & Cheo 1971). A stand with a high fire hazard possesses elements containing flammable products or products that sustain the combustion such as low moisture and lignin contents and / or high resins or essential oil contents, (Pompe & Vines 1966; Philpot 1970; Susott et al. 1975; Susott 1980). Conifer fuel particles contain many of these products (Van Wagner 1977). Moreover, fuel elements may have a spatial distribution that can also favour fire propagation such as high surface / volume ratio, aerated particle bed, or ladder fuels like the basal conifer branches (Montgomery & Cheo 1971; Taylor & Fonda 1989). Finally, different surface fuel characteristics are hypothesized to lead to different fire behaviours (rate of spread of the fire front (m / min.) and head fire intensity (kW / m)) during the early phase of fire.

The objective of this study is to describe how changes in surface fuel characteristics (species composition, particle sizes, loads, spatial arrangement) during the species replacement succession in the mixedwood boreal forest affect potential fire ignition and surface fire behaviour (mainly during the surface fire phase). First, we will analyse the surface fuel components (woody debris, litter, duff, and shrub layers) across 48 stands that mostly differ in their canopy composition (surficial deposits, drainage, elevation, slope, and natural disturbance types all being nearly identical). This will lead us to evaluate the stand fire hazard. Secondly, the initial surface fire behaviour will be modelled from these fuel types and loads for each stand using the BEHAVE prediction system.

3.4. METHODS

3.4.1. Study area

The study area is located around Lake Duparquet (see Harvey 1999 for detailed map), in the Clay Belt of northwestern Quebec (48°30'N, 79°20'W), a large physiographic region characterized by lacustrine clay deposits left by the proglacial lakes Barlow and Ojibway (Vincent and Hardy 1977). The area surrounding Lake Duparquet has forests that have never been commercially harvested. Lake Duparquet is situated at the southern limit of the boreal forest in the Missinaibi-Cabonga section (Rowe 1972), which is characterized by the presence of balsam fir (*Abies balsamea* (L.) Mill.), black spruce (*Picea mariana* (Mill.). B.S.P.), paper birch (*Betula papyrifera* Marsh.), white spruce (*Picea glauca* (Moench) Voss) and trembling aspen (*Populus tremuloides* Michx.). The mean annual temperature is 0.6° C; the mean annual precipitation is 822.7 mm; and the mean annual frost-free period is 64 days. However, freezing temperatures may occur throughout the year (Environment Canada 1993).

3.4.2. Field sampling

Forty-eight stands were sampled on gently sloping mesic clay deposits (Appendix 3.1). All were regenerated from stand-replacing fires dating from 32 to 236 years ago. Year of stand initiation has been determined in previous dendrochronological studies (Bergeron 1991; Dansereau & Bergeron 1993). Each stand was sampled with a 30 m-sided equilateral triangle (McRae et al. 1979). Canopy characteristics (tree species and diameter at breast height (DBH) in centimeters) were sampled using the point-centered quadrant method (Mueller-Dombois & Ellenberg 1974; McRae et al. 1979). Six points were set up along triangle sides and forty-eight trees were recorded per stand: twenty-four dominate trees and twenty-four suppressed trees. The forty-eight distances between trees and the quadrant centers were measured to calculate tree densities per stand and per species (McRae et al. 1979). Stand and species basal areas were calculated from the DBH. The forty-eight stands were then characterized as being either deciduous, mixed-deciduous, mixed-coniferous, or coniferous stands, depending on the conifer basal area (< 25%, 25-50%, 50-75%, or > 75% of stand basal area, respectively; Appendix 3.1). This stand type classification was chosen to reconstruct the main successional stages because all stands do not present the same rate of changes after the fire. For example, nine stands older than 170 years old and mainly dominated by trembling aspen have less than 25 % conifer in their canopy basal area, whereas other stands of the same age are coniferous.

To evaluate the stand fire hazard, all surface fuel types have been first measured within each stand. Woody debris were measured by the line intersect method for each stand (Van Wagner 1968, 1980) along the sides of the equilateral triangle (McRae et al. 1979). The triangular layout was used to minimize bias in situations where woody debris are not randomly oriented, and to cover the variation in woody debris distribution (Van Wagner 1980). The six classes recommended by McRae et al. (1979) were first used. All surficial woody pieces crossing the sampling plane were recorded in the six diameter size classes: class I = 0 - 0.49 cm on 15m line intersect; class II = 0.5 - 0.99 cm on 30m; class III = 1 - 2.99 cm on 45m; class IV = 3 - 4.99 cm on 60m; class V = 5 - 6.99 cm on 75m. For these classes, the total number of woody pieces and species proportions were measured (all species can be differentiated; Appendix 3.2). For each item greater than 7 cm in diameter we recorded the species and diameter size (to the nearest 0.1 cm). These large dead woody pieces constitute the sixth class. We did not use decay classes because only dead woody material on the litter (upper duff logs) were measured, and few showed different decay states. The equation to calculate dead wood loading (Van Wagner 1980) is:

$$W = \frac{(Gk)}{L} * \sum d^{2} * a * c \qquad (3.1)$$

where W is weight per unit ground area (t / ha), G is specific gravity (g / cm ^{3}), k is a constant corresponding to 1.234 (see Van Wagner 1980, p3.), L is the length of sample intersect line (m), d is the piece diameter at intersection (cm), a is a correction factor for nonhorizontal angle of woody pieces and corresponds to 1.13 (see Brown 1974), and c is the slope correction computed from the tilt angle of each triangle side factor (see Table 4 in McRae et al. 1979). The term,

$$\sum d^{2} = n * \left(dq^{2}\right) \qquad (3.2)$$

where n is number of intercepts per species and diameter size class, and dq^2 is specific squared quadratic-mean diameter (cm^2). As no previous study focusing on dead woody material has been done in Quebec, we used specific multiplication factors Z from Ontario

(Tables 2 and 3 in McRae et al. 1979) for balsam fir, white birch, white spruce and trembling aspen, with Z factor corresponding to:

$$Z = \frac{G * k * (dq)^2 * a}{L} \tag{3.3}$$

So, the dead wood loading equation was:

$$W = Z * n * c \tag{3.4}$$

For white cedar, original quadratic-mean diameters and particle specific gravities from Minnesota were used (Roussopoulos 1978). As quadratic-mean diameters and specific gravities were provided for the American diameter size classes, linear interpolations were computed to calculate corresponding quadratic diameters and gravities for the Canadian diameter size classes. The Z multiplication factors were then calculated for white cedar to compute the cedar log loads per diameter size class. Both Ontario and Minnesota are the nearest geographically and the best ecologically relevant data sources for the study area.

As the BEHAVE system inputs FWD from three time lag classes (1-h, 10-h, and 100-h time lags) corresponding to pieces < 0.62 cm, 0.63 - 2.54 cm, and 2.55 - 7.62 cm in diameter, respectively, we used a linear interpolation to split loads of diameter size classes from the six classes seen above into the three American time lag classes for dead woody fuels (Bradshaw et al. 1983). This procedure was used to remove woody pieces greater than 7.62 cm in diameter from the simulation as they are not relevant to the flame ignition, nor in the fire front rate of spread prediction.

Shrub and litter materials (Brown et al. 1982) were measured in quadrats evenly spaced along the 90-m triangle transect. Shrubs basal diameters were measured by species in nine quadrats ($1m^2$ each) at 10-m intervals, and shrub loads were calculated from equations determined from shrub samples collected in the Duparquet area (Aubin 1998 unpubl.). Shrub height and percentage of dead shrubs were visually estimated. Litter (L layer) and duff (F+H layers) depths were measured in twelve quadrats (25cm * 25cm each), and total materials from litter and duff were separately collected to obtain litter and duff oven-dried weights.

To simulate the fire ignition and early fire behaviour under different weather conditions, three fire weather indices from the Canadian Fire weather (Van Wagner 1987) were selected. The three indices represent low, moderate and extreme fire danger conditions

(FWI=5: low; FWI=15: moderate; and FWI=25: extreme, as used by the SOPFEU (Society for the protection of forests against fire of Quebec)). In order to apply the simulation on real weather data from the region, the 1200-hour local standard time weather data (temperature, precipitation for the previous 24-h period, wind speed, and relative humidity) were extracted for the 1991-1997 period from four local weather stations set up around Lake Duparquet and FWI were computed for every day of these fire seasons. Several days corresponded to each set of fire danger conditions. As several combinations of intermediate FWI indices (Build-Up Index, Initial Spread Index) can result in the same final FWI, therefore two days per fire danger conditions were selected, one with the minimum BUI and maximum ISI and one with the maximum BUI and minimum ISI (Table 3.1).

Table 3.1 Six weather conditions and fire weather indices from local meteorological stations around Lake Duparquet.

Replicate #	Day	Temp. (°C)	Relat. Hum. (%)	Wind speed (km/h)	FFMC	ISI	BUI	FWI
1	D-1	30	18	21				
	D	30	24	9	87.4	4.6	11.5	5
2	D-1	19	76	6				
	D	26	53	9	72.7	1.1	76.7	5
1	D-1	14	36	3				
	D	16	35	22	89.0	11.5	15.1	15
2	D-1	28	59	5				
	D	29	65	5	86.8	3.4	92.5	15
1	D-1	22	14	7				
	D	23	14	9	95.4	14.1	40.4	25
2	D-1	31	15	4				
	D	25	29	7	91.8	8.0	85.5	25

Note: FFMC = Fine Fuel Moisture Content, is a numerical rating of the moisture content of litter and other cured fie fuels. This code is an indicator of the relative ease of ignition and flammability of fine fuel. ISI = Initial Spread Index, is a rating of the expected rate of fire spread. It combines the effects of wind and FFMC on rate of spread without the influence of variable quantities of fuel. BUI = Buildup index, is a numerical rating of the total amount of fuel available for combustion. FWI = Fire Weather Index, a rating of fire intensity that combines ISI and BUI. It is suitable as a general index of fire danger throughout the forested areas of Canada (Canadian Forestry Service 1987). D = Weather conditions used in BEHAVE; D-1 = Weather conditions the day previous the simulated day.

3.4.3. Simulations

Using simulations we wanted to evaluate the effect of the different fuels within the 48 stands on the ignition potential and the fire behaviour in the first stages of the fire propagation using the rate of spread (ROS) and the head fire intensity (HFI). The BEHAVE System is composed of two subsystems: the fuel modeling system referred to as FUEL (Burgan and Rothermel 1984) and the fire behaviour prediction subsystem named BURN with FIRE1 and FIRE2 programs (Andrews 1986; Andrews and Chase 1989). The FUEL subsystem provides 13 standard existing fuel models that can be used unaltered or modified to create new fuel models based on the measured loading data for each fuel component. The SITE module in the FIRE1 program predicts rate of spread and frontal fire intensity, whereas the SIZE module in the FIRE1 program calculates the area burned from a point source that results in a rough elliptical shape. The weather and topography conditions are fully described inputs in the SITE module, which uses information included in the FUEL model file to provide the fire behaviour prediction outputs.

First the FUEL subsystem was used to create 48 new fuel models based on combinations of measured loading for the following fuel types: litter, live shrubs, 1-h time lag fuels (1-h time lag woody debris + dead shrub wood fitting in the 1-h time lag class), 10-h time lag fuels (10-h time lag woody debris + dead shrub wood fitting in the 10-h time lag class), and 100-h fuels (100-h time lag woody debris). Each fuel type was assigned a standard surface-area-to-volume ratio following the suggestions provided by Burgan and Rothermel (1984).

In the second step, weather conditions for the previous day and the selected day (D-1 and D) were used to calculate the moisture content of the three time lag fuel types (Table 3.1). The 1-h time lag fuel moisture content was calculated from the MOISTURE module of the FIRE2 BURN subsystem (Andrews & Chase 1989). The 10-h time lag fuel moisture content was predicted from the equilibrium moisture content equation of the National Fire Danger Rating System (Bradshaw et al. 1983). This equation calculates the 10-h time lag fuel moisture content from the 1-h time lag fuel type. Finally, the 100-h time lag fuel moisture content was directly calculated in the SITE module of the FIRE1 BURN subsystem

(Andrews 1986). The moisture content of living herbs and shrubs was fixed at 100% according to the suggestion of Burgan and Rothermel (1984).

In the third step, the SITE module in the FIRE1 program was used to predict rate of spread (ROS) in m/min. and frontal fire intensity (HFI) in kW/m. The simulation was completed by using the SIZE module, also part of the FIRE1 program, to calculate the area burned in hectares. This last fire behaviour variable was selected to compare the potential fire sizes in different fuel complexes two hours after the fire ignition. Indeed, in terms of fire protection in Quebec, a fire has to be detected when its size is less than 0.5 - 1 ha in order to be successfully contained. To calculate the area burned, the elapsed time since the ignition was fixed at two hours. This elapsed time was selected to take also into account the necessary acceleration time required to reach the equilibrium state that has been previously calculated by the Canadian Forest Fire Behavior Prediction (FBP) system (Forestry Canada Fire Danger Group 1992) with the selection of the point source ignition pattern. Without taking into account such a delay to reach the equilibrium state, the ROS would have no meaning.

Each simulation (one stand for one weather condition) was run twice; the first simulation for spring fires when the deciduous and shrub foliage are absent and the second simulation with summer fires when all the foliage is in place. This implied a total of 576 simulations (48 stands * 6 weather conditions * 2 fire seasons). With respect to topography, a zero slope effect and an elevation of 300-m were used to represent conditions on the study area. Finally, a factor for the crown closure (90% on D, 70% in MD, 50% in MC, and 30% in C stands) and a factor for the crown height / crown diameter ratio (1 for D, 1.5 for MD, 2 for MC, 2.5 for C stands, see Andrews (1986)) were associated with the wind speed to calculate the midflame wind speed that is an intermediate index used in the ROS calculation.

3.4.4. Data analysis

Comparison of surface material characteristics among the four stand types

Multiple mean comparison tests (Kruskal-Wallis non-parametric test and Hsu's MCB test from the software JUMP (1989)) were computed to compare dead wood loads, depth and load for litter and duff, and total loads of shrubs (dead and live) distributed into three basal diameter size classes (class I: < 0.62 cm, class II: 0.63 - 2.54 cm, and class III: 2.55 - 7.62 cm in diameter) among the four stand types.

Fire behaviour simulation predictions and comparisons among the four stand types

The same non-parametric Kruskal-Wallis test and Hsu's MCB multiple mean comparison tests were used to compare the four stand types according to ROS, HFI, and burned area. For a given fire season (spring or summer) and a given stand, the simulations from the six weather conditions were combined so that the weather variability is removed.

3.5. RESULTS

3.5.1. Fire hazard analysis

If we take into account the 48 stands *per se*, all time lag classes do not show significant trends of loads with time since the last fire, but an increase can, however, be seen (Figure 3.1). Total mean dead wood load per stand is 51.1 ton/ha (Table 3.2). Woody pieces with a diameter greater than 7 cm, or the cumulative load of the three main species (balsam fir, trembling aspen, and white birch), represent about 63 % of the total dead wood load. The mean duff load was ten times heavier than the mean litter load, and the duff depth was three times greater than litter depth (6.4 and 2.2 cm for duff and litter, respectively). Finally, although shrub loads were highly heterogeneous, shrub load of the diameter size class I was eight times lighter than the size class II load and twelve times lighter than the size class III load. For more details, the individual stand values are provided in the Appendix 3.1.

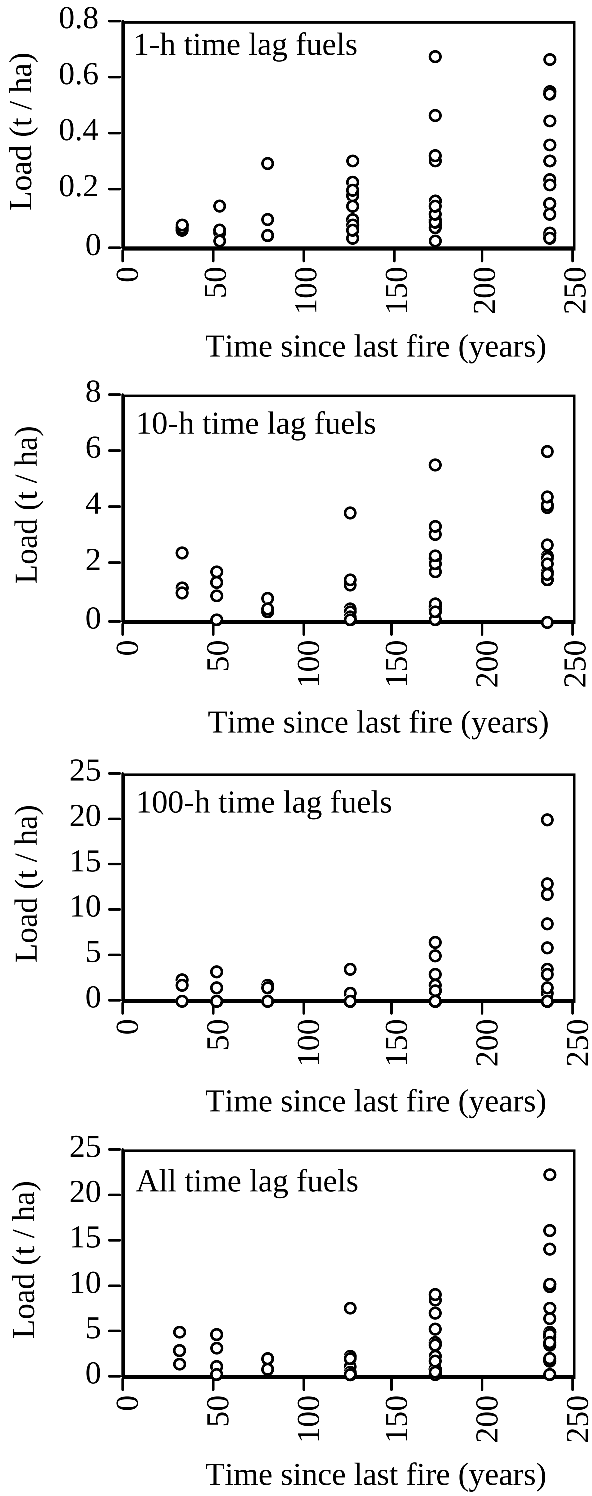

Figure 3.1 Changes within the different time lag fuel loads through time since the last fire.

Table 3.2 Comparison of four stand types for surface fuel characteristics.

	$p > \chi^2$	Stand type mean	Deciduous 24 stands		Mixed-deciduous 13 stands		Mixed-coniferous 4 stands		Coniferous 7 stands	
Stand age	0.0001		117 ± 11	c	175 ± 15	b	205 ± 26	ab	236 ± 20	a
DEAD WOOD										
Total load	0.5776	51.1 ± 8.1	49.5 ± 4.7		55. ± 6.4		63.8 ± 11.5		42.2 ± 8.7	
Total load of class I	0.2314	4.0 ± 0.5	3.6 ± 0.3		4.2 ± 0.4		4.6 ± 0.7		4.7 ± 0.6	
Total load of class II	0.0147	2.2 ± 0.1	1.9 ± 0.1	b	2.2 ± 0.2	b	2.5 ± 0.3	ab	3.1 ± 0.3	a
Total load of class III	0.7513	5.0 ± 0.2	4.9 ± 0.3		5.0 ± 0.4		5.3 ± 0.6		5.1 ± 0.5	
Total load of class IV	0.5008	3.5 ± 0.5	3.9 ± 0.3		3.4 ± 0.4		3.1 ± 0.7		2.8 ± 0.6	
Total load of class V	0.1681	4.3 ± 0.9	4.6 ± 0.6		4.4 ± 0.7		4.3 ± 1.3		3.1 ± 1.0	
Total load of class VI	0.6052	32.1 ± 8.0	30.7 ± 4.6		35.8± 6.3		44.0 ± 11.4		23.4 ± 8.6	
Total load for balsam fir	0.0038	18.5 ± 1.8	12.7 ± 2.4	c	19.0 ± 2.9	bc	28.0 ± 5.2	ab	31.8 ± 4.0	a
Total load for white spruce	0.1762	1.5 ± 1.1	0.6 ± 0.6		2.2 ± 0.8		6.15 ± 1.5		0.4 ± 1.1	
Total load for white cedar	0.0006	3.0 ± 1.3	0.2 ± 1.8	b	6.2 ± 2.4	a	8.3 ± 4.3	a	4.3 ± 3.2	ab
Total load for aspen	0.0001	15.0 ± 2.4	24.3 ± 2.8	a	8.4 ± 3.8	b	4.9 ± 6.9	b	0.8 ± 5.2	b
Total load for white birch	0.1091	13.2 ± 6.0	11.7 ± 3.4		19.5 ± 4.7		16.6 ± 8.4		4.9 ± 6.3	
LITTER AND DUFF										
Litter load	0.2388	4.4 ± 0.4	4.3 ± 0.2		4.8 ± 0.3		4.0 ± 0.5		4.5 ± 0.4	
Litter depth	0.8037	2.2 ± 0.1	2.2 ± 0.1		2.3 ± 0.2		2.5 ± 0.3		2.0 ± 0.2	
Litter density	0.2027	21.8 ± 1.4	22.0 ± 2.0		22.5 ± 2.8		16.0 ± 5.0		23.5 ± 3.8	
Duff load	0.2813	46.7 ± 5.4	44.3 ± 3.1		48.4 ± 4.2		54.4 ± 7.7		47.7 ± 5.8	
Duff depth	0.1761	6.4 ± 0.3	5.9 ± 0.3		6.8 ± 0.5		7.4 ± 0.9		6.6 ± 0.6	
Duff density	0.9633	73.5 ± 2.7	74.3 ± 4.0		73.1 ± 5.4		73.9 ± 9.7		71.1 ± 7.3	
SHRUBS										
Total load in class I	0.0167	0.2 ± 0.0	0.1 ± 0.0	b	0.2 ± 0.0	ab	0.2 ± 0.0	ab	0.4 ± 0.1	a
Total load in class II	0.0036	1.7 ± 0.2	1.0 ± 0.3	b	2.3 ± 0.4	a	1.9 ± 0.7	ab	2.8 ± 0.5	a
Total load in class III	0.2053	2.5 ± 1.8	1.6 ± 1.0		2.6 ± 1.3		1.1 ± 2.4		4.5± 1.8	

Note: p results from Kruskal-Wallis' test. Loads are in ton/ha, depths in cm, and densities in kg/m^3. For dead wood: class I<0.5 cm in diameter; 0.5<classII<1 cm; 1<class III<3 cm; 3<class IV<5 cm; 5<class V<7 cm; class VI=>7 cm. For shrubs: class I<0.62 cm in diameter; 0.63<class II<2.54 cm 2.55<class III<7.62 cm. Values are means ± standard error. Values in a row followed by the same letter are not significantly different at $\alpha = 0.05$ for Hsu's test.

If we look at the 48 stands classified through the four stand types, different significant trends in the mean stand characteristics for each surface fuel type do appear (Table 3.2). Deciduous stands are significantly younger than mixed and conifer stands, even though these deciduous stands include nine 173-year old stands. Mixed-deciduous stands are not significantly younger than mixed-conifer stands, but they are significantly younger than conifer stands. There is also no significant difference between mixed-conifer and conifer stands ages. Concerning the different fuel types, the total dead wood load does not show a significant difference among the four stand types. However, several significant differences occur once the total woody debris load is broken down into six diameter size classes, or into the different dead wood producing species. The load of size class II as well as the woody debris loads of balsam fir and white cedar increase significantly with the increase of conifer basal area, whereas the trembling aspen dead wood load decreases significantly at the same time. There were no significant differences among the four stand types for the white spruce and white birch loads or for any other diameter size class loads. These different changes in the species composition of dead woody material among the different stand types are illustrated in Figure 3.2 in terms of species percentages across diameter size classes. It is interesting to note that litter and duff components present no difference in relation to the change of stand composition (Table 3.2). Finally, the first two shrub load classes (less than 2.54 cm in diameter) are significantly lower in the pure deciduous stands.

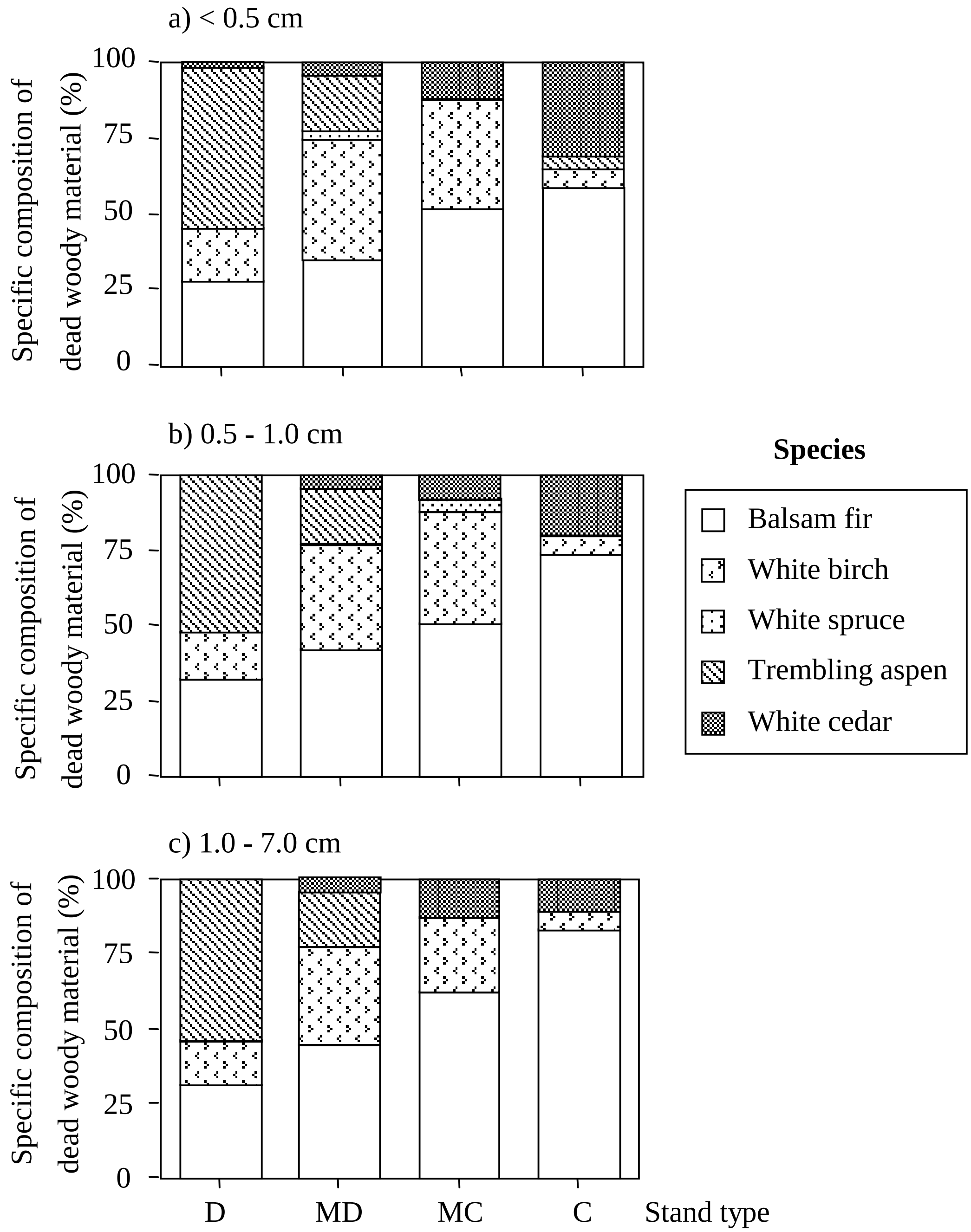

Figure 3.2 Mean specific composition (%) of dead woody pieces within each stand type (D: deciduous, MD: mixed-deciduous, MC: mixed-coniferous, and C: coniferous) for three diameter size classes.

3.5.2. Potential fire behaviour among the four stand types

The fire behaviour predictions from the BEHAVE System for spring and summer and the results of the comparisons among the four stand types are reported in Table 3.3. The six tests are all significant. In each stand type, spring fire behaviour values are higher than summer values, but the differences seem to decrease as the percentage of conifer basal area increases. For all fire behaviour variables, and for the two fire seasons, deciduous stands yield the lowest significant predicted values, while most of the time coniferous stands show the highest values for the same variables. Mixed stands have intermediate predicted values, with no significant differences between mixed-deciduous and mixed-coniferous stands for the two fire seasons.

Table 3.3 Results of Kruskal-Wallis and multiple comparison tests on ranks for simulated rate of spread, head fire intensity, and area burned for spring and summer seasons using the BEHAVE system

Season		Stand type	Rate of spread Score mean		Head fire intensity Score mean		Area burned 2 h since ignition Score mean	
Spring	$\chi2$		12.723		47.860		18.629	
	p		0.0053		0.0001		0.0003	
		D	130.872	b	114.837	c	126.219	b
		MD	161.038	a	169.154	ab	165.538	a
		MC	134.104	ab	135.708	bc	130.604	ab
		C	166.452	ab	205.440	a	176.048	ab
Summer	$\chi2$		20.216		68.253		33.212	
	p		0.0002		0.0001		0.0001	
		D	127.545	b	109.483	c	119.580	b
		MD	152.737	a	162.615	b	158.333	ab
		MC	151.104	ab	155.917	b	154.208	a
		C	183.560	a	224.393	a	198.702	a

Note: Stand type is D for deciduous, MD for mixed-deciduous, MC for mixed-coniferous, and C for coniferous stands. Score means from Kruskal-Wallis tests are given for each combination for a given season; results with the same letters are not significantly different at $\alpha = 0.05$ from the Hsu's MCB and the Kruskal-Wallis tests.

3.6. DISCUSSION

3.6.1. Comparisons of surface fuels with other boreal ecosystems

Table 3.4 presents the mean surface fuel values obtained for the four stand types in this study and for other sites in the boreal forest. Woody debris volumes are a more practical unit for comparison with other studies. Linder et al. (1997) observed fuel loads that ranged from 16 to 80 m^3/ha for deciduous stands and from 31 to 168 m^3/ha for coniferous stands. For the mixed-coniferous stands, Sturtevant et al. (1997) found values that ranged from 15 to 80 m^3/ha. Noticable differences in log volumes were found by Lee et al. (1997) for deciduous stands (109-124 m^3/ha) in Alberta and by Freedman et al. (1996) for deciduous and mixed stands (13-20 and 33 m^3/ha, respectively) in New Brunswick. It is likely that the drier climate in Alberta creates a less favourable environment for decomposing fungi. This would slow down bole decay rate and result in a greater log accumulation. Conversely, in New Brunswick, the moister climate has inverse effects, and it favours log decay. Moreover, Freedman et al. (1996), by sampling only logs greater than 5 cm in diameter, have not taken into account small diameter log loads that can comprise up to 43% of the total load in our study area. In boreal forest studies dealing with organic matter characteristics, data for duff are more numerous than those for litter. Even though the fuelbeds have variable composition, total loads from this study were in the same value ranges as those observed in other studies (Vogt et al. 1986; Barney & Van Cleve 1973).

3.6.2. Surface fuel heterogeneity and changes in fire susceptibility

The southeastern Canadian mixedwood boreal forest appears as a relatively homogeneous forest mosaic at the landscape level when total amounts of surface fuels (Appendix 3.1) or different time lag fuel loads are analyzed (Figure 3.1). Even though the trends with time are not significant, the visual increase of loads with time for all time lag classes (Figure 3.1) confirms partially the accumulation patterns of fuels through time found in previous studies in the boreal forest (Wein & McLean 1983; Lee et al. 1997; Sturtevant et al. 1997).

Table 3.4 Surface fuel components characteristics for comparisons with other boreal forests.

Stand type	BA of living trees (m^2/ha)	Stand age (years)	Total log volume (m^3/ha)	Duff load (t/ha)	Duff depth (cm)	References
Deciduous	45.9 ± 3.8	117 ± 11	40.1 ± 3.8	44.3 ± 3.1	5.9 ± 0.3	this study
Deciduous		88 - 108	16 - 80©			1
Deciduous (aspen)		23 - 146	108.8 - 124.3			2
Deciduous (aspen)				38.3 - 60.0 @		3
Deciduous (birch)				24.8 - 68.8 @		3
Deciduous	29.3		13.0 - 19.9 #			4
Mixed (deciduous >)	37.4 ± 5.2	175 ± 15	44.6 ± 5.2	48.4 ± 4.2	6.8 ± 0.5	this study
Mixed (coniferous >)	26.1 ± 9.4	205 ± 26	51.7 ± 9.4	54.4 ± 7.7	7.4 ± 0.9	this study
Mixed	28.0 - 33.1		32.7 #			4
Mixed (coniferous >)	22.1 ± 50.5	33 -110	15 - 80			5
Coniferous	35.3 ± 7.1	236 ± 20	34.2 ± 7.1	47.7 ± 5.8	6.6 ± 0.6	this study
Coniferous (spruce)		133 - 245	34 - 166©			1
Coniferous (pines)		117 - 270	31 - 68©			1
Coniferous (spruce)				63.7 - 128.9 @		3
Coniferous (fir)				65 - 117 @		3
Coniferous	25.4 - 30.5		41.6 - 56.5 #			4
Coniferous (spruce-fir)		51 - 55		17	6.6 - 7.9	6

Note: # = only logs greater than 5 cm in diameter; ©= only logs greater than 10 cm in diameter ; @ = litter and duff included; references : 1 = Linder et al. 1997 ; 2 = Lee et al. 1997 ; 3 = Vogt et al. 1986 ; 4 = Freedman et al. 1996 ; 5 = Sturtevant et al. 1997 ; 6 = Barney and Van Cleve 1973.

In a previous study (Hély et al. 2000), we showed in the mixedwood boreal forest that coarse woody debris do not present the common U-shape accumulation with time found in other boreal forest types (Lambert et al. 1980; Lee et al. 1997; Sturtevant et al. 1997). When the four different stand types are used as successive stages (D, MD, MC, and C, respectively) to reconstruct the post-fire succession pathway (Bergeron & Dubuc 1989), the homogeneity still exists (Table 3.2), but it is explained by the high variability of species replacement rates. Low accumulation in the mixedwood boreal forest is the result of species replacement in the canopy through time and the rate of changes (Appendix 3.1, and Bergeron & Dubuc 1989), fast decay rates observed for the involved species (Alban & Pastor 1993) and their different elements (wood, leaves, and needles). However, composition and quality of dead wood fuels do change considerably from deciduous towards coniferous elements (Figure 3.2, Table 3.2, and Appendix 3.2). Moreover, this replacement is correlated to the basal areas of the dominant trees of the canopy (Hely et al. 2000). This change in fuel elements will lead to an increase in the amount of conifer pieces that typically are more flammable than deciduous pieces (Brown & Davis 1973; Rowe & Scotter 1973), an increase in small dead wood loads, and the associated development of basal conifer branches that can act as vertical fuel ladders. These changes will increase the stand fire hazard. Furthermore, the vertical ladders also constitute aerial fuels (Heinselman 1973) that are involved in surface and crown fires.

For the litter and the duff layers, Paré et al. (1993) found similar results for the litter layer with no difference in load along the chronosequence, whereas they found a significant increase in the ash-free dry-weight of the duff layer with time (not significant in the present study, see Table 3.2). However, the fire hazard of the litter compartment should change from deciduous to coniferous stands. As the structure of the litter changes from deciduous leaves towards needles, litter should decrease the compactness of the litter bed and create a more aerated fuelbed that could more easily propagate the flame and lead it to reach shrubs and ladder fuels.

The increase of the small diameter shrub loads from the deciduous to coniferous stands is the result of a change in shrub composition with the canopy species replacement (De Grandpré et al. 1993). *Taxus canadensis* is a low evergreen shrub with small diameter that dominates the shrub layer in the mixed-conifer and coniferous stands, whereas the deciduous and mixed-deciduous stands are frequently dominated by herb species such as *Aster macrophyllus* and *Aralia naudicaulis* or tall shrubs with large diameters such as *Acer* *spicatum* and *Corylus cornuta* (De Grandpré et al. 1993). These changes in the shrub layer from deciduous to coniferous dominated stands are expected to increase its fire hazard as the load of small diameter fuels will increase and given that *Taxus canadensis* contains flammable essential oils.

3.6.3. Measured fire susceptibility and predicted fire behaviour

The increase in fire hazard along the tree species replacement in the canopy was partially confirmed by comparisons of simulations from the BEHAVE System. Deciduous stands, when they do burn, sustain a fire behaviour with relatively low fire intensity and rate of spread as compared to conifer stands that can sustain relatively high fire intensity and rate of spread. These differences in fire behaviour will imply smaller burn areas in deciduous stands than in conifer stands for a given period of time. However, the prediction values of fire intensity, rate of spread, and burned areas from BEHAVE should be used with caution because no comparison can be done with real fire behaviour data recorded in the mixedwood boreal forest. Indeed, there is a lack of data from wildfires or prescribed burns that would have occurred in this forest ecosystem type. To develop the FBP system that is used across Canada, the Canadian Forest Service had to create mixedwood boreal forest equations for fire behaviour components from fire behaviour data available recorded for black spruce stands and pure aspen stands (Forestry Canada Fire Danger Group 1992; Hirsch 1996). This system predicts fire behaviour from the FWI and the canopy composition in terms of percentage of conifer for the mixedwood stand types. Some simulation tests done with the FBP would show higher predicted values than those predicted by BEHAVE. Nevertheless, using the FBP system, deciduous stands would always record the less intense fire behaviour, mixed stands would present moderate fire intensities, while conifer stands would record the extreme fire behaviour.

Season is also an influencing factor on fire hazard because spring simulations yielded higher values of fire behaviour variables than those observed in summer. The higher the deciduous percentage, the higher spring and summer differences; however, the only difference recorded between spring and summer is exclusive to the vegetation phenology with the presence or absence of the deciduous leaves. In spring, the total absence of deciduous foliage does not interfere with the direct ground and surface fuel bed warming from the direct sunlight (Furyaev et al. 1983), whereas in summer deciduous leaves intercept the sunlight and create a cool and moist understory and ground environments (Van Wagner 1983). The BEHAVE system takes this season change into account with the date, the main species (deciduous or conifer species), and the foliage presence requirements. It results in a slower rate of spread in summer associated with less intensity released and smaller burned areas. Conifer stands in this region are generally older, and they present a more open canopy than deciduous stands. Because these conifer species are evergreen, new foliage represents only a small proportion of their total foliage load (Brown & Davis 1973). The difference of environment under conifer stands between spring and summer will then be small and spring and summer fire behaviour will be quite equivalent. When a difference exists between spring and summer fire behaviour in conifer stands, it comes from an herb cover presence in summer that retains higher moisture content in the surface fuels and decreases the propagation rate. This ground cover does not take into account a moss layer since this fuel type is almost non-existent in these stand types (De Grandpré et al. 1993). However these smaller values in fire behaviour components for summer simulations are not visible for conifer dominated stands in Table 3.3 as only the relative differences between stands are reported. In this case, the differences between deciduous and coniferous dominated stands overwhelm the season effect within a given stand.

3.7. CONCLUSION

The analysis of the different surface fuel types in the mixedwood boreal forest suggests that stand fire hazard increases through the canopy tree species replacement sequence in the southeastern Canadian boreal forest. This increase in fire hazard does not result from a heavy fuel load accumulation, but rather from many changes in the quality of surface fuels. Flammable materials (small woody particles, conifer pieces, aerated organic matter bed) are more important in mixed or coniferous stands than in the deciduous ones. The modeling aspect of fire behavior suggests an increase in fire rate of spread and head fire intensity with increasing conifer density in the late successional stands.

Fuel load accumulation through time exists in the mixedwood boreal forests, but it is less important to fire behavior than in other forest ecosystems. Nevertheless, the fire suppression policies in place in the region and the global warming trend decreasing the fire frequency will be responsible for an aging of the forest mosaic whereby the proportion of coniferous stands has a chance to increase with extended fire exclusion. The stand fire hazard will then increase not from a long-term heavy fuel accumulation but from a more flammable and susceptible fuel composition.

3.8. ACKNOWLEDGMENTS

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3.9. REFERENCES

Alban, D.H. & Pastor, J. 1993. Decomposition of aspen, spruce, and pine boles on two sites in Minnesota. *Can. J. For. Res*. 23 (9): 1744 - 1749.

Andrews, P.L. 1986. *BEHAVE: fire behavior prediction and fuel modeling system--BURN subsystem, PART 1*. Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT-194: 130p.

Andrews, P.L. & Chase, C.H. 1989. *BEHAVE: fire behavior prediction and fuel modeling system--BURN subsystem, PART 2*. Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT-260: 93p.

Barney, R.J. & Van Cleve, K. 1973. Black spruce fuel weights and biomass in two interior Alaska stands. *Can. J. For. Res.* 3: 304-311.

Bergeron, Y. 1991. The influence of island and mainland lakeshore landscapes on boreal forest fire regimes. *Ecology* 72 (6): 1980-1992.

Bergeron, Y. 2000. Species and stand dynamics in the mixed-woods of Quebec Southern Boreal Forest. *Ecology*. In-press.

Bergeron, Y. & Dubuc, M. 1989. Succession in the southern part of the Canadian boreal forest. *Vegetatio* 79: 51-63.

Bergeron, Y. & Archambault, S. 1993. Decreasing frequency of forest fires in the southern boreal zone of Quebec and its relation to global warming since the end of the "Little Ice Age". *The Holocene* 3 (3): 255-259.

Bergeron, Y. & Dansereau, P.R. 1993. Predicting the composition of Canadian southern boreal forest in different fire cycles. *J. Veg. Sci.* 4: 827-832.

Bradshaw, L.S., Deeming J.E., Burgan, R.E. & Cohen, J.D. 1983. *The 1978 National Fire Danger Rating System: technical documentation*. Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT-169: 41p.

Brown, A.A. & Davis, K.P. 1973. *Forest fire: control and use.* McGraw-Hill, New York.

Brown, J.K. 1974. *Handbook for inventorying downed woody material*. Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT 16 : 24p.

Brown, J.K., Oberheu, R.D. & Johnston, C.M. 1982. *Handbook for inventoring surface fuels and biomass in the Interior West*, Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT-129: 48p.

Burgan, R.E. & Rothermel, R.C. 1984. *BEHAVE: fire behavior prediction and fuel modeling system--FUEL subsystem*. Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT-167: 126p.

Burns, R.H. & Honkala, B.H. 1990. *Silvics of North America.* USDA Agricultural Handbook 654. Washington, DC.

Carleton, T.J. & Maycock, P.J. 1978. Dynamics of the boreal forest south of James Bay. *Can. J. Bot*. 56: 1157-1173.

Dansereau, P.R. & Bergeron, Y. 1993. Fire history in the southern boreal forest of northwestern Quebec. *Can. J. For. Res.* 23: 25-32.

De Grandpré, L., Gagnon, D. & Bergeron, Y. 1993. Changes in the understory of Canadian southern boreal forest after fire. *J. Veg. Sci.* 4: 803-810.

Engelmark, O., Bradshaw, R. & Bergeron, Y. 1993. Disturbance dynamics in boreal forest: introduction. *J. Veg. Sci.* 4: 730-732.

Environment Canada. 1993. *Canadian climate normals 1961-90*. Canadian Climate Program. Environment Canada. Atmospheric Environment Sevice. Downsview, Ontario.

Flannigan, M.D., Bergeron, Y., Engelmark, O. & Wotton, M. 1998. Future wildfire in circumboreal forests in relation to global warming. *J. Veg. Sci.* 9: 469-476.

Forestry Canada Fire Danger Group. 1992. *Development and structure of the Canadian Forest Fire Behavior Prediction System*. For. Can., Ottawa, Ontario, Inf. Rep. ST-X-3: 63p.

Freedman, B., Zelazny, V., Beaudette, D., Fleming, T., Flemming, S., Forbes, G., Gerrow, J.S., Johnson, G., & Woodley, S. 1996. Biodiversity implications of changes in the quantity of dead organic matter in managed forests. *Environ. Rev.* 4: 238-265.

Furyaev, V.V., Wein, R.W. & MacLean, D.A. 1983. Fire influences in *Abies*-dominated forests. In: R. W. Wein & D.A. MacLean (eds.) *The role of fire in Northern circumpolar ecosystems*; pp. 221-234. Scope 18: John Wiley & Sons, Chichester, England.

Harvey, B. 1999. The Lake Duparquet Research and Teaching Forest: Building a foundation for ecosystem management. *For. Chron.* 75 (3): 389-393.

Heinselman, M.L. 1973. Fire in the virgin forest of the Boundary Waters Canoe Area, Minnesota. *Quat. Res.* 3: 329-382.

Heinselman, M.L. 1981. Fire and succession in the conifer forests of northern North America. In: D.C. West, H.H. Shugart & D.B. Botkin (eds.) *Forest succession: concepts and application*, pp. 374-405. Springer-Verlag, New York.

Hély, C., Bergeron, Y. & Flannigan, M.D. 2000. Coarse woody debris in the southeastern Canadian boreal forest: composition and load variations in relation to stand replacement. *Can. J. For. Res.* 30: 674-687.

Hirsch, K.G. 1996. *Canadian Forest Fire Behavior Prediction (FBP) System: user's guide*. Nat. Res. Can., Can. For. Serv., Northwest Reg., North. For. Cent., Edmonton, AB. Spec. Rep. 7: 121p.

JUMP. 1989. Statistical visualization for the Macintosh. Version 2.0.4. SAS Institute, Inc., Cary, NC.

Johnson, E.A. 1992. *Fire vegetation dynamics: Studies from the North American boreal forest.* R.S.K. Barnes, H.J.B. Birks, E.F. Connor & R.T. Paine (eds.) Cambridge studies in ecology, Cambridge Univ. Press, Cambridge.

Lambert, R.L., Lang, G.E. & Reiners, W.A. 1980. Loss of mass and chemical change in decaying boles of a subalpine balsam fir forest. *Ecology* 61 (6): 1460-1473.

Lee, P.C., Crites, S., Nietfeld, M., Van Nguyen, H. & Stelfox, J.B. 1997. Characteristics and origins of deadwood material in aspen-dominated boreal forests. *Ecol. Applic.* 7 (2): 691-701.

Linder, P., Elfving, B. & Zackrisson, O. 1997. Stand structure and successional trends in virgin boreal forest reserves in Sweden. *For. Ecol. Manage.* 98: 17-33.

McRae, D.J., Alexander, M.E. & Stocks, B.J. 1979. *Measurement and description of fuels and fire behavior on prescribed burns: a handbook*. Can. For. Serv., Great Lakes For. Res. Cent. Sault Ste. Marie, Ont. Rep. O-X-287: 44p.

Montgomery, K.R. & Cheo, P.C. 1971. Effect of leaf thickness on ignitability. *For. Sci.* 17 (4): 475-478.

Mueller-Dombois, D. & Ellenberg, H. 1974. *Aims and Methods of Vegetation Ecology*. Wiley, New York.

Paré, D., Bergeron, Y. & Camiré, C. 1993. Changes in the forest floor of Canadian southern boreal forest after disturbance. *J. Veg. Sci.* 4: 811-818.

Philpot, C.W. 1970. Influence of mineral content on the pyrolysis of plant materials. *For. Sci.* 16 (4): 461-471.

Pompe, A. & Vines, R.G. 1966. The influence of moisture content on the combustion of leaves. *Aust. For.* 30 (3): 231-241.

Roussopoulos, P.J. 1978. *An appraisal of upland forest fuels and potential fire behavior for a portion of the boundary waters canoe area*. Ph.D. Dissertation, Michigan State University, Department of Forestry : 166p.

Rowe, J.S. 1972. *Forest regions of Canada*. Environment Canada, Ottawa, Ontario. Publ. 1300.

Rowe, J.S. & Scotter, G.W. 1973. Fire in the boreal forest. *Quat. Res.* 3: 444-464.

Schimmel, J. & Granström, A. 1997. Fuel succession and fire behavior in the Swedish boreal forest. *Can. J. For. Res.* 27: 1207-1216.

Shugart, H.H., Leemans, R. & Bonan, G.B. 1992. *A systems analysis of the global boreal forest.* Cambridge Univ. Press: 563p.

Simard, A.J. 1973. *Forest fire weather zones of Canada*. Environ. Can., Can. For. Serv., Ottawa, Ont. map with text.

Sprugel, D.G. 1985. Changes in biomass components through stand development in wave-regenerated balsam fir forests. *Can. J. For. Res.* 15: 269-278.

Stocks, B.J. 1987. Fire behavior in immature jack pine. *Can. J. For. Res*. 17: 80-86.

Stocks, B.J., Lawson, B.D., Alexander, M.E., Van Wagner, C.E., McAlpine, R.S., Lynham, T.J., & Dubé, D.E. 1989. Canadian Forest Fire Danger Rating System: an overview. *For. Chron*. 65 (4): 258-265.

Sturtevant, B.R., Bissonette, J.A., Long, J.N. & Roberts, D.W. 1997. Coarse woody debris as a function of age, stand structure and disturbance in boreal Newfoundland. *Ecol. Applic*. 7 (2): 702-712.

Susott, R.A. 1980. Thermal behavior of conifer needle extractives. *For. Sci.* 26 (3): 347-360.

Susott, R.A., De Groot, W.H. & Shafizadef, F. 1975. Heat content of natural fuels. *Journal of Fire and Flammability* 6: 311-325.

Taylor, K.L. & Fonda, R.W. 1989. Woody fuel structure and fire in subalpine fir forests, Olympic National Park, Washington. *Can. J. For. Res.* 20: 193-199.

Turner, M.G. & Romme, W.H. 1994. Landscape dynamics in crown fire ecosystems. *Landscape Ecology* 9 (1): 59-77.

Van Wagner, C.E. 1968. The line intersect method in forest fuel sampling. *For. Sci*. 14 (1) : 20-26.

Van Wagner, C.E. 1977. Conditions for the start and spread of crown fire. *Can. J. For. Res.* 7: 23-34.

Van Wagner, C.E. 1980. *Practical aspects of the line intersect method*. Can. For. Serv., Petawawa For. Exp. Stn., Chalk River, Ont., Info. Rep. PI-X-12.

Van Wagner, C.E. 1983. Fire behaviour in northern conifer forests and shrublands. In: R.W. Wein & D.A. Maclean (eds.). *The role of fire in Northern Circumpolar Ecosystems,* pp. 65-81. Scope 18, John Wiley & Sons, Chichester, England.

Van Wagner, C.E. 1987. *Development and structure of the Canadian Forest Fire Weather Index System*. Can. For. Serv., Ottawa, Ont., Tech. Rep. 35: 36p.

Vincent, J.S. & Hardy, L. 1977. L'évolution et l'extension des lacs glaciaires Barlow et Ojibway en territoire québécois. *Géogr. Phys. Quat*. 31: 357-372.

Vogt, K.A., Grier, C.C. & Vogt, D.J. 1986. Production, turnover, and nutrient dynamics of above- and belowground detritus of world forests. *Adv. Ecol. Res.* 15: 303-377.

Wein, R.S. & McLean, D.A. 1983. Fire and succession in the conifer forests of Northern North America. In: R.W. Wein & D.A. Maclean (eds.). *The role of fire in Northern Circumpolar Ecosystems,* pp. 374-405. Scope 18, John Wiley & Sons, Chichester, England.

CHAPITRE IV

ROLE OF VEGETATION AND WEATHER ON FIRE BEHAVIOUR IN THE CANADIAN MIXEDWOOD BOREAL FOREST USING TWO FIRE BEHAVIOUR PREDICTION SYSTEMS

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4.1. RÉSUMÉ

Des simulations de feux printaniers et estivaux ont été réalisées à l'aide du système de prédiction du comportement des feux de forêts (FBP) et du système BEHAVE pour étudier le rôle de la végétation et du climat dans le comportement des feux en forêt boréale mixte. Les peuplements autour du lac Duparquet (Québec, Canada) ont été caractérisés comme feuillus, mixtes-feuillus, mixtes-conifères, ou conifères selon le pourcentage de surface terrière coniférienne. Les charges des différents types de combustibles (litière, humus, débris ligneux, herbacées, et arbustes), et différentes conditions climatiques locales (trois niveaux de risque de feu) ont été entrées dans les modèles. Les variables prédites du comportement du feu sont la vitesse de propagation du front de flamme (ROS), l'intensité dégagée (HFI), et la surface brûlée. Les résultats des ANOVA montrent que les deux facteurs ne sont pas toujours significatifs et que qualitativement les systèmes attribuent différemment la variance expliquée par les deux facteurs: le climat est le principal facteur intervenant dans le comportement du feu pour le FBP, alors que la végétation prédomine pour BEHAVE. Cependant, trois feux prescrits réalisés en Ontario ont révélé que le système BEHAVE est inadéquat pour cette région de la forêt boréale, alors que le FBP propose des prédictions quantitatives proches des valeurs réelles obtenues. Un Indice Forêt Météo (IFM) extrême crée effectivement des feux très vastes et intenses, cependant des différences de comportement du feu selon les types de peuplements existent quel que soit l’IFM atteint. Les impacts des changements climatiques, de la composition de la végétation, ainsi que la saison sur le comportement du feu et sur la mosaïque forestière sont discutés.

4.2. ABSTRACT

Spring and summer simulations were carried out using the Fire Behavior Prediction (FBP) and BEHAVE systems to study the role of vegetation and weather on fire behaviour in the mixedwood boreal forest. Stands at Lake Duparquet (Quebec, Canada) were characterized as either being deciduous, mixed-deciduous, mixed-coniferous, or coniferous according to their percentage of conifer basal area. Sampled fuel loads (litter, duff, woody debris, herbs, and shrubs), and local weather conditions (three different fire risk levels) were used as inputs in the simulations. The predicted fire behaviour variables were rate of spread (ROS), head fire intensity (HFI), and area burned. Results from ANOVAs showed that both factors are not always significant, and that the two systems qualitatively attribute the explained variance to those factors differently. The FBP selects the weather as the most important factor for all fire behaviour variables, whereas BEHAVE selects the vegetation factor. However, three research burns located in Ontario revealed that BEHAVE was not well adapted to the mixedwood boreal region, whereas FBP predictions were quantitatively close to observed experimental burn values. Extreme Fire Weather Index (FWI) is confirmed as being responsible for large and intense fires, but differences in fire behaviour among stand types exist whatever the FWI. Implications of climate change, vegetation, and seasonal effects on fire behaviour and the forest mosaic are discussed.

4.3. INTRODUCTION

The boreal forest stretches across Canada and is a natural ecosystem affected by large-scale natural disturbances (Shugart et al. 1992) such as insect outbreaks (Blais 1983; Holling 1992; Morin 1995) and fires (Cogbill 1984; Johnson 1992; Levine et al. 1993; Pickett and White 1985; Turner and Romme 1994; Schimmel and Granström 1997). The fire environment is composed of weather, fuels, and topography, and these three factors constantly interact (Agee 1997). However, the respective importance of the role of these factors may vary according to the region, the ecosystem type and its historical events (Fryer and Johnson 1988; Harrington et al. 1991; Johnson 1992), through factors such as ignition probability (depending upon lightning and human population density), species composition, stand structure, and the climate conditions associated with each site (Harrington et al. 1991; Heinselman 1973; Pickett and White 1985; Shugart et al. 1992; Wein and Moore 1977). For example, Bessie and Johnson (1995) have shown that weather was the most important factor for fire occurrence in the western part of Canada. Bergeron and Archambault (1993) have shown that high fire frequency in the eastern Canadian mixedwood boreal forest was associated with longer drought periods in summer during the " Little Ice Age", whereas low fire frequency was associated with moister summer periods. Different studies have dealt with increasing risk of fire ignition and fire propagation as a result of long-term fuel accumulation in several ecosystem types (Aber and Melillo 1991; Dodge 1972; Schimmel and Granström 1997; Wright and Bailey 1982). The composition differences among studied forest types may explain the differences of interpretations. The weather may become the most important factor where the forest mosaic composition is homogeneous, or when the fire frequency is low and fires occur under extreme weather conditions such as the blocking high-pressure anomaly events (Flannigan and Harrington 1988). On the other hand, the vegetation composition may be the driving factor when the fire frequency is high within a forest mosaic where the stand composition is quite variable. The mixedwood boreal forest is an interesting ecosystem in which to analyze the respective effects of meteorological conditions and vegetation characteristics, because the fire frequency is relatively high and the vegetation composition highly variable. Furthermore, the respective role of these two factors is required in several applications such as determinig if fire control policies have to be based on vegetation composition, and if fire cycles differ according to stand type.

The first objective of this study is to evaluate the respective effects of vegetation characteristics and weather conditions on the fire behaviour in the mixedwood boreal forest. Both factors should be significant, but the vegetation (fuel) factor is most important. This hypothesis is based on the species composition (particularly with the presence of numerous deciduous and mixed stands) and the differential flammability to the seasonal phenology changes (Van Wagner 1983) that exist in the mixedwood boreal forest. In this study, flammability is defined as the time spent in pyrolysis (Johnson 1992) and corresponds with the delay required for a particle exposed to a source of heat to be chemically decomposed before the ignition occurs. The delay ends with the occurrence of the flaming stage of combustion. The climatic conditions during the fire season also vary, but it may not be as important as the vegetation variability.

Among the tools available to assess respective effects of meteorological conditions and vegetation characteristics on fire behaviour, there are two fire behaviour prediction systems used across North America. The first one is the Canadian Forest Fire Behavior Prediction (FBP) System (Forestry Canada Fire Danger Group 1992), an empirical model based on wildfire and prescribed burn data. This model is used to determine surface and crown fire behaviour. The second one is the BEHAVE System developed for the United States (Andrews 1986; Andrews and Chase 1989; Burgan and Rothermel 1984), a deterministic model that is based on the physical properties of fuels studied in the laboratory rather than on field data. The BEHAVE System is only used to determine surface fire behaviour. Both models were built to predict the fire behaviour and to help understand the fire effects on the different ecosystem compartments. The systems give as primary outputs the rate of spread of the fire front (ROS) and the head fire intensity (HFI), which combined together, can determine the fire severity. In this study, the fire severity refers to the fire impact on the ecosystem through the fuel consumption and the fire-caused vegetation mortality. A secondary output variable is the burned area for an elapsed time since fire ignition, which forest managers can use to estimate potential patch sizes. It should be noted, however, that these two systems use several inputs with varying definitions within the fire environment, and this could result in different fire behaviour predictions.

The second objective of this study is to compare the two national fire behaviour prediction systems (FBP and BEHAVE). First, we will compare the two systems using the results obtained for spring and summer from simulations using fuel inventories of 48 sampled stands and 6 weather conditions corresponding to 3 fire risk levels. Secondly, we will use independent data sets obtained from three actual research burns conducted in the mixedwood boreal forest of Ontario. We will compare the observed surface fire behaviour components recorded during the research burns with predicted fire behaviour sets from the two systems using fuel inventories and weather conditions that were observed during these fires. The BEHAVE system should give more realistic predictions because it is based on measured fuel loading, whereas the FBP system defines the fuel type according to the stand type, which is a general qualification.

4.4. MATERIALS AND METHODS

4.4.1. Fire behaviour prediction systems background

The FBP System

Detailed information about the Canadian Forest Fire Danger Rating System (CFFDRS) and its subsystems, the Canadian Forest Fire Weather Index (FWI) and the FBP Systems can be found in Canadian Forestry Service (1987), Forestry Canada Fire Danger Group (1992), and Hirsch (1996). The FWI and FBP subsystems relate to the relative wildland fire potential and the actual fire behaviour, respectively. The FBP system has sixteen general fuel types, which represent many, but not all, of the major fuel types found in Canada (Hirsch 1996). For the weather inputs, the FBP System uses the Fine Fuel Moisture Code (FFMC), the Initial Spread Index (ISI), and the Build-up Index (BUI) from the FWI System. These indices are considered as fuel moisture codes and fire behaviour indices, and they are calculated from 1200-hour local standard time observations of temperature, relative humidity, wind speed, and precipitation for the previous 24 hours. The third fire environmental factor is topography, and it can be characterized using percent slope and aspect.

The BEHAVE System

The BEHAVE System is made up of two subsystems: the fuel modeling system referred to as FUEL (Burgan and Rothermel 1984) and the fire behavior prediction subsystem named BURN with FIRE1 and FIRE2 programs (Andrews 1986; Andrews and Chase 1989). The FUEL subsystem provides 13 standard existing fuel models that can be used unaltered or modified to create new fuel models based on the measured loading data for each fuel component. The SITE module in the FIRE1 program predicts rate of spread and frontal fire intensity, whereas the SIZE module in the FIRE1 program calculates the area burned from a point source that results in a rough elliptical shape. The weather and topography conditions are fully described inputs in the SITE module, which uses information included in the FUEL model file to provide the fire behaviour prediction outputs.

4.4.2. Relative importance of vegetation and weather on fire behaviour

Study area description

The study area is located on the Lake Duparquet Research and Teaching Forest, located in the Clay Belt of northwestern Quebec (48°30'N, 79°20'W), a large physiographic region characterized by lacustrine clay deposits left by the proglacial lakes Barlow and Ojibway (Vincent and Hardy 1977). The area surrounding Lake Duparquet has forests that have never been commercially harvested. Lake Duparquet is situated at the southern limit of the boreal forest in the Missinaibi-Cabonga section (Rowe 1972), which is characterized by balsam fir (*Abies balsamea* (L.) Mill.) as the dominant species. Black spruce (*Picea mariana* (Mill). BSP), paper birch (*Betula papyrifera* Marsh.), white spruce (*Picea glauca* (Moench) Voss) and trembling aspen (*Populus tremuloides* Michx.) also represent important components in the forest. The mean annual temperature is 0.6°C, the mean annual precipitation is 822.7 mm, and the mean annual frost-free period is 64 days. However, freezing temperatures may occur throughout the year (Environment Canada 1993).

Data collection

Stands selection

Forty-eight stands were selected on mesic clay deposits (with gentle slope) around Lake Duparquet. All stands have lethal fires dating from 32 to 236 years ago (Bergeron 1991; Dansereau and Bergeron 1993). Each stand was inventoried using a 30-m sided equilateral sample triangle (McRae et al. 1979) to evaluate all downed woody fuels. We sampled the stand structure (tree species, tree densities) using the point centered quadrant method (McRae et al. 1979) with six sample points located along the triangle, and the equations from Mueller-Dombois and Ellenberg (1974) to calculate tree densities. The forty-eight stands were characterized as either being deciduous, mixed-deciduous, mixed-coniferous, or coniferous stands, if their conifer basal area was less than 25%, between 25 and 50%, between 50 and 75%, or greater than 75% of stand basal area, respectively.

Fuel inventory

For each stand, downed woody fuels were measured by the line intersect method (Van Wagner 1968) along the equilateral triangle (McRae et al. 1979) to measure all pieces from diameters less than 0.5 cm to big branches and boles with diameters greater than 7 cm. We first used the five classes recommended by McRae et al. (1979). We then used a linear interpolation to split these diameter class loads from the five classes seen above to the three American classes 1-h, 10-h, and 100-h time lag dead woody loads (Bradshaw et al. 1983). Shrub, herbs, and litter fuels (Brown et al. 1982) were measured in quadrats evenly spaced along the 90-m triangle transect. Shrubs (by species) had basal diameter measured in nine quadrats (1 m^2) at 10-m intervals. Loads were calculated from equations determined from shrub samples collected in the Duparquet area (Aubin, unplublished data). Height and percentage of dead material were also measured. Litter (L layer) and duff depths (F+H layers) were measured in twelve quadrats (0.0625 m^2 each) and their material was separated and collected to obtain their oven-dried weight. The surface fuel components to be used in the BEHAVE prediction System were divided into the following classes: litter, live shrubs and herbs, 1-, 10- and 100-h time lag fuels (dead wood and dead shrubs when appropriate). Finally, each fuel class was assigned a standard surface-area-to-volume ratio according to Burgan and Rothermel (1984).

Weather data source

All 1200-hour local standard time weather data (temperature, precipitation for the previous 24-h, wind speed, and relative humidity) for the 1991-1997 period were obtained from four local weather stations set up around Lake Duparquet. From these data sets, we selected three fire weather index levels (Table 4.1). To compare with other Canadian references, the low fire weather risk selected (FWI = 5) corresponds with the average fire weather of the Canadian zones 3 and 4 where the study area is located (Simard 1973). The moderate (FWI = 15) and extreme (FWI = 25) fire weather indices selected have been previously used for experimental burns in Ontario (Stocks 1987; Stocks et al. 1989). Because several combinations of intermediate FWI indices (Build-Up Index, Initial Spread Index) can result on the same final FWI, weather conditions from two days (D and D-1) were selected from the observed data, corresponding with simultaneously minimum BUI and maximum ISI or the inverse (Table 4.1).

Simulation characteristics

With respect to topography, a zero slope effect and an elevation of 300-m were used to represent conditions in the study area. The point-source ignition pattern was chosen to emulate natural fire ignitions. This pattern is automatically given for BEHAVE whereas the FBP system provides options for using either a point source or line as ignition source pattern. The elapsed time since ignition for this study was fixed at two hours to calculate the burned area (in hectares). Two hours was selected as the necessary acceleration time required to reach the equilibrium state, which has been previously calculated. Indeed, among the forty-eight stands, the slowest acceleration took more than one hour to reach the equilibrium state.

Table 4.1 Six weather conditions and fire weather indices from local meteorological stations around Lake Duparquet.

Replicate #	Day	Temp. (°C)	Relat. Hum. (%)	Wind speed (km/h)	FFMC	ISI	BUI	FWI
1	D-1	30	18	21				
	D	30	24	9	87.4	4.6	11.5	5
2	D-1	19	76	6				
	D	26	53	9	72.7	1.1	76.7	5
1	D-1	14	36	3				
	D	16	35	22	89.0	11.5	15.1	15
2	D-1	28	59	5				
	D	29	65	5	86.8	3.4	92.5	15
1	D-1	22	14	7				
	D	23	14	9	95.4	14.1	40.4	25
2	D-1	31	15	4				
	D	25	29	7	91.8	8.0	85.5	25

Note: FFMC = Fine Fuel Moisture Content, is a numerical rating of the moisture content of litter and other cured fie fuels. This code is an indicator of the relative ease of ignition and flammability of fine fuel. ISI = Initial Spread Index, is a rating of the expected rate of fire spread. It combines the effects of wind and FFMC on rate of spread without the influence of variable quantities of fuel. BUI = Buildup index, is a numerical rating of the total amount of fuel available for combustion. FWI = Fire Weather Index, a rating of fire intensity that combines ISI and BUI. It is suitable as a general index of fire danger throughout the forested areas of Canada (Canadian Forestry Service 1987). D = Weather conditions used in BEHAVE; D-1 = Weather conditions the day previous the simulated day.

Simulations using the two fire behaviour prediction systems (FBP and BEHAVE) were run separately for all forty-eight stands using the six different weather condition days. Simulations were based on spring conditions when deciduous foliage is absent (M1 in the FBP System) and on summer conditions when deciduous foliage is in place (M2 in the FBP System). The foliage presence is requested in the BEHAVE System. This design led us to do 576 simulations for each system. We recorded the fire front rate of spread (ROS) in meters per minute, the frontal fire intensity at the fire's head (HFI) in kilowatts per meter, and the area burned in hectares two hours after fire ignition to have a complete understanding of the fire behaviour characteristics.

For the FBP simulations, the only variation among stands is the relative coniferous basal area, which must be estimated when using the FBP System. We used the ROS variable from the FBP System instead of direct ISI from the FWI System. Indeed, the ROS calculation in the FBP system takes into account the specific fuel type and the topography, whereas the ISI takes into account the "relative potential spread in a standard mature pine fuel type " (Hirsch 1996).

For the BEHAVE simulations (Andrews 1986), three moisture contents of the time lag fuels, related to same daily weather in the FWI-FPB Systems, are needed. The 1-h time lag fuel moisture content was calculated from the MOISTURE module of the FIRE2 BURN subsystem (Andrews and Chase 1989) using the same weather data (D-1 and D days) that were selected for the FBP System. The 10-h time lag fuel moisture content was predicted from the equilibrium moisture content equation of the National Fire Danger Rating System (Bradshaw et al. 1983). This equation calculates the 10-h time lag fuel moisture content from the 1-h time lag fuel type. Finally, the 100-h time lag fuel moisture content was directly calculated in the SITE module of the FIRE1 BURN subsystem (Andrews 1986). The moisture content of living herb and shrubs was fixed at 100% according to the suggestion of Burgan and Rothermel (1984).

Analyses

We analyzed the respective role of stand types and weather conditions using a two-way ANOVA design on rank scores (Conover and Iman 1981) with the General Linearized Model procedure (SAS Institute Inc. 1985). We tested both factor effects (fuel and weather) and their potential interaction. The explained variance was partitioned using the type III sum of squares. We also looked at the differences in the fire behaviour variables among the four stand types and the three FWI values using a Kruskal-Wallis test followed by a Hsu's MCB test (JUMP 1989). Finally, using these last two tests we analyzed the FBP outputs between the four stand types for each FWI taken individually.

4.4.3. Fire behaviour comparisons between research burns and predictions from simulators

Research burn data

The research burns used in this study were conducted in the mixedwood boreal forest of Ontario (46°38'N, 83°25'W) between 1992 and 1998. They represent three of the ten plots that have been selected to support fire behaviour experiments. Plot size is 1 ha and stand structure characteristics are synthesized in Appendix 4.1, as there is quite a difference between plots even though they are located on the same site. According to the weather conditions, the two first plots were burned under an FWI of 23 and 21, respectively, whereas the plot #3 burned with an FWI of 16.

Fire behaviour simulations were conducted using the two systems to compare the prescribed and predicted results, and to enable us to determine if a system is realistic in simulating fire behaviour for this region of the boreal forest. According to Appendix 4.1, plots #1 and #3 are considered as mixed-conifer stands while plot #2 is a coniferous stand. We conducted only spring fire simulations because all three burns occurred during the spring season.

Analyses

We used the total time spent by the fire front to burn the entire plot area to calculate the mean ROS. Secondly, total fuel consumption (McRae et al. 1979) was calculated from the depth of burn measurements and fuel loading. The total heat release was then calculated from the fuel consumption and the low heat of fuel combustion. Finally, fuel consumption, low heat of fuel combustion, and ROS were used to calculate the frontal fire intensity. We then compared the observed results from the research burns with the predicted results from the two simulation systems.

4.5. RESULTS

4.5.1. Relative importance of vegetation and weather on fire behaviour

The results from the two-way ANOVA, done on the simulated output variables (rate of spread, head fire intensity, and burned area) for spring and summer simulated fires with both the FBP and the BEHAVE Systems, are presented in Table 4.2. All overall tests are highly significant, but vegetation and weather are not always significant factors. The partitioning of the explained variance presents differences between the FBP and BEHAVE systems, and between spring and summer fire behaviour. The FBP system takes into account only one significant interaction between the two factors but this interaction accounts for less than 3% of the total explained variance. Both factors are significant from the FBP system, but the maximum explained variance is always attributed to the weather, and the importance of weather decreases from spring to summer fire simulations. For spring ROS and burned area variables, the weather importance is about nine times higher than that of the vegetation factor, whereas in summer simulations the weather importance is only four to five times higher than for the vegetation factor. For the HFI variable, weather importance is four times higher than the vegetation factor for both seasons. From the BEHAVE system simulations, vegetation is the only significant factor for HFI and burned area for the entire fire season. For ROS, both factors are significant but the vegetation type is the most important factor, and its associated explained variance increases from spring to summer fires.

4.5.2. Differences in the fire behaviour variables among the four stand types and the three FWI values

Differences for ROS, HFI, and burned area among stand types and FWI are presented in Figures 4.1, 4.2, and 4.3, respectively. These figures present the spring and summer fire simulations for both systems. The difference in the ranges of values between both systems for a given variable follows always the same pattern: the quantitative outputs from the FBP system are always higher than those from the BEHAVE System. The three figures will be analyzed together because they show the same trends for each season and each system.

Table 4.2 Results of the two-way ANOVA done on ranks for the simulated rate of spread, head fir intensity, and area burned for spring and summer fires on two simulators

Simulator	**Variable**	Spring fire			Summer fire		
		F	p	Variance partition	F	p	Variance partition
	Rate of spread						
FBP System	Overall model	171.25	0.0001		126.37	0.0001	
	Stand type	27.86	0.0001	9.76%	47.82	0.0001	22.71%
	FWI	286.34	0.0001	90.24%	244.19	0.0001	77.29%
BEHAVE System	Overall model	4.77	0.0003		5.87	0.0001	
	Stand type	4.42	0.0041	56.87%	7.31	0.0001	74.73%
	FWI	5.14	0.0064	41.13%	3.71	0.0258	25.27%
	Head fire intensity						
FBP System	Overall model	64.68	0.0001		106.63	0.0001	
	Stand type	41.25	0.0001	24.04%	46.84	0.0001	26.36%
	FWI	188.78	0.0001	73.37%	196.31	0.0001	73.64%
	Stand type * FWI	2.22	0.0413	2.59%			
BEHAVE System	Overall model	11.8	0.0001		18.54	0.0001	
	Stand type	18.96	0.0001	100%	29.7	0.0001	100%
	FWI	1.07	0.3445	0%	1.8	0.1669	0%
	Burned area						
FBP System	Overall model	208.9	0.0001		132.31	0.0001	
	Stand type	24.7	0.0001	7.09%	43.74	0.0001	19.83%
	FWI	485.19	0.0001	92.91%	265.17	0.0001	80.17%
BEHAVE System	Overall model	4.5	0.0006		8.45	0.0001	
	Stand type	6.59	0.0003	100%	12.51	0.0001	100%
	FWI	1.37	0.2553	0%	2.37	0.0954	0%

Note: F tests and associated probabilities are given. The explained variance was partitioned between factors using the type III sum of squares. Interactions are indicated by *.

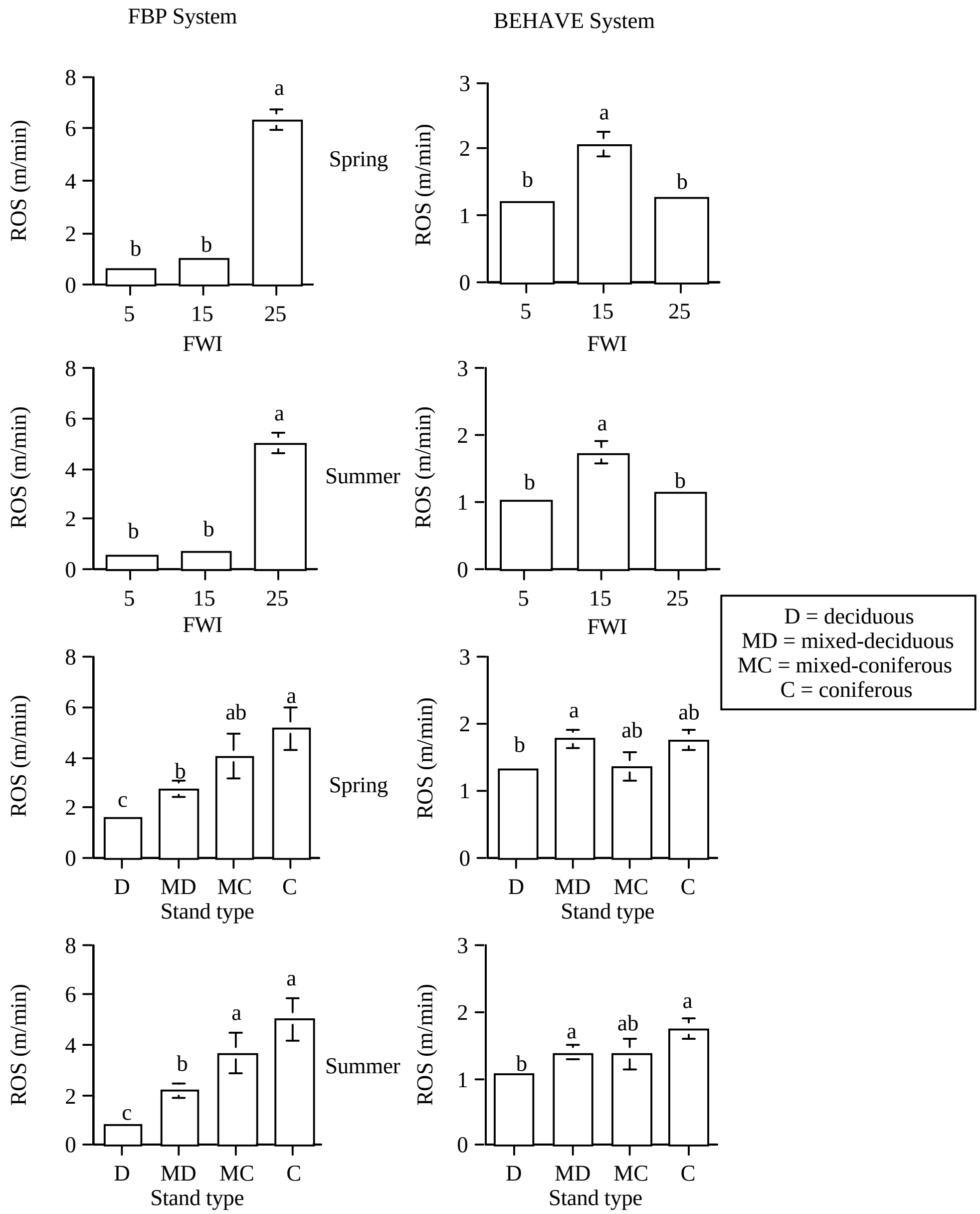

Figure 4.1 Differences in the simulated rates of spread (ROS) between the FBP and BEHAVE systems for spring and summer fires according to stand types and fire weather indices (FWI). Kruskal-Wallis tests followed by Hsu's MCB tests (JUMP 1989) have been performed to analyse the differences. For each graph, rates of spread with the same letter are not significantly different at $\alpha = 0.05$.

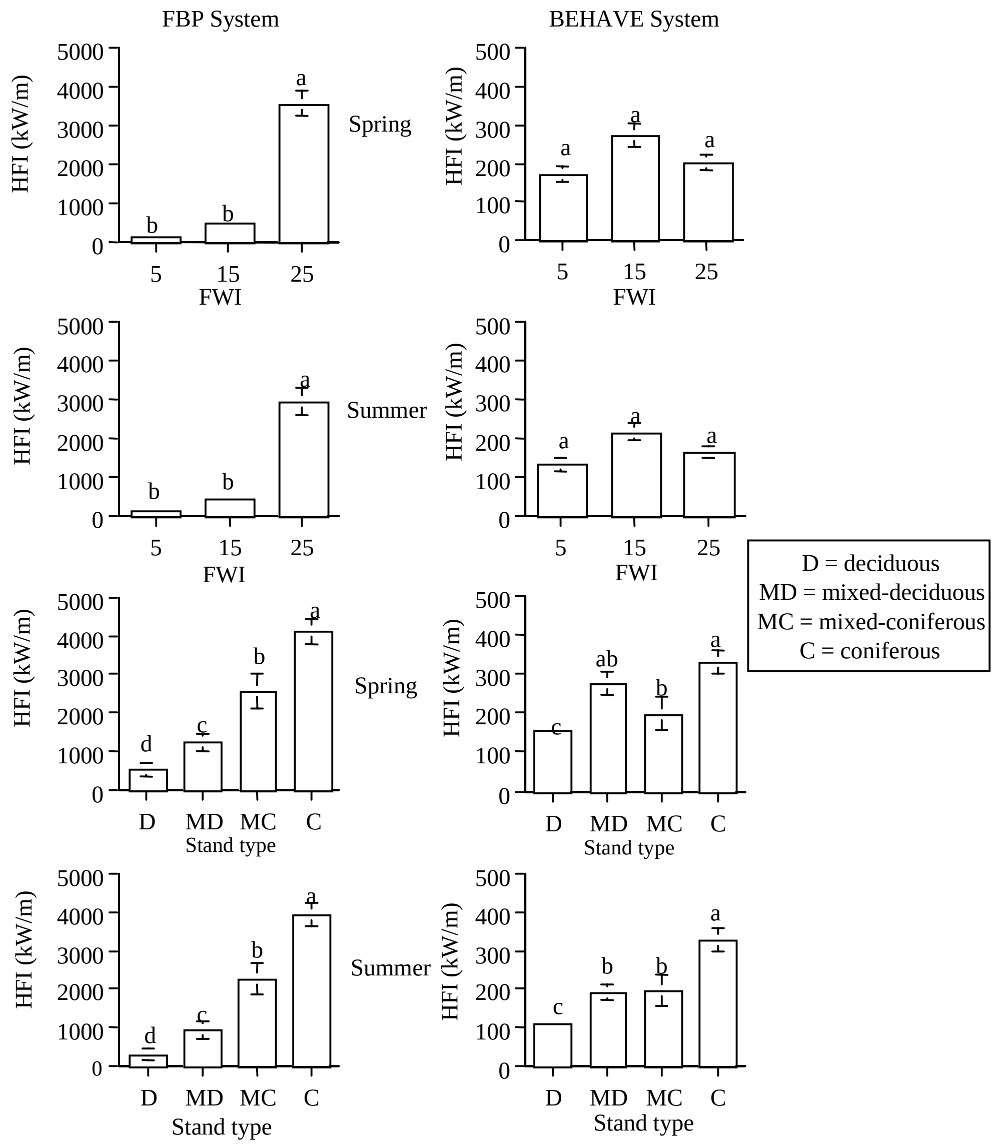

Figure 4.2 Differences in the simulated head fire intensities (HFI) between the FBP and BEHAVE systems for spring and summer fires according to stand types and fire weather indices (FWI).

Kruskal-Wallis tests followed by Hsu's MCB tests (JUMP 1989) have been performed to analyse the differences. For each graph, head fire intensities with the same letter are not significantly different at $\alpha = 0.05$.

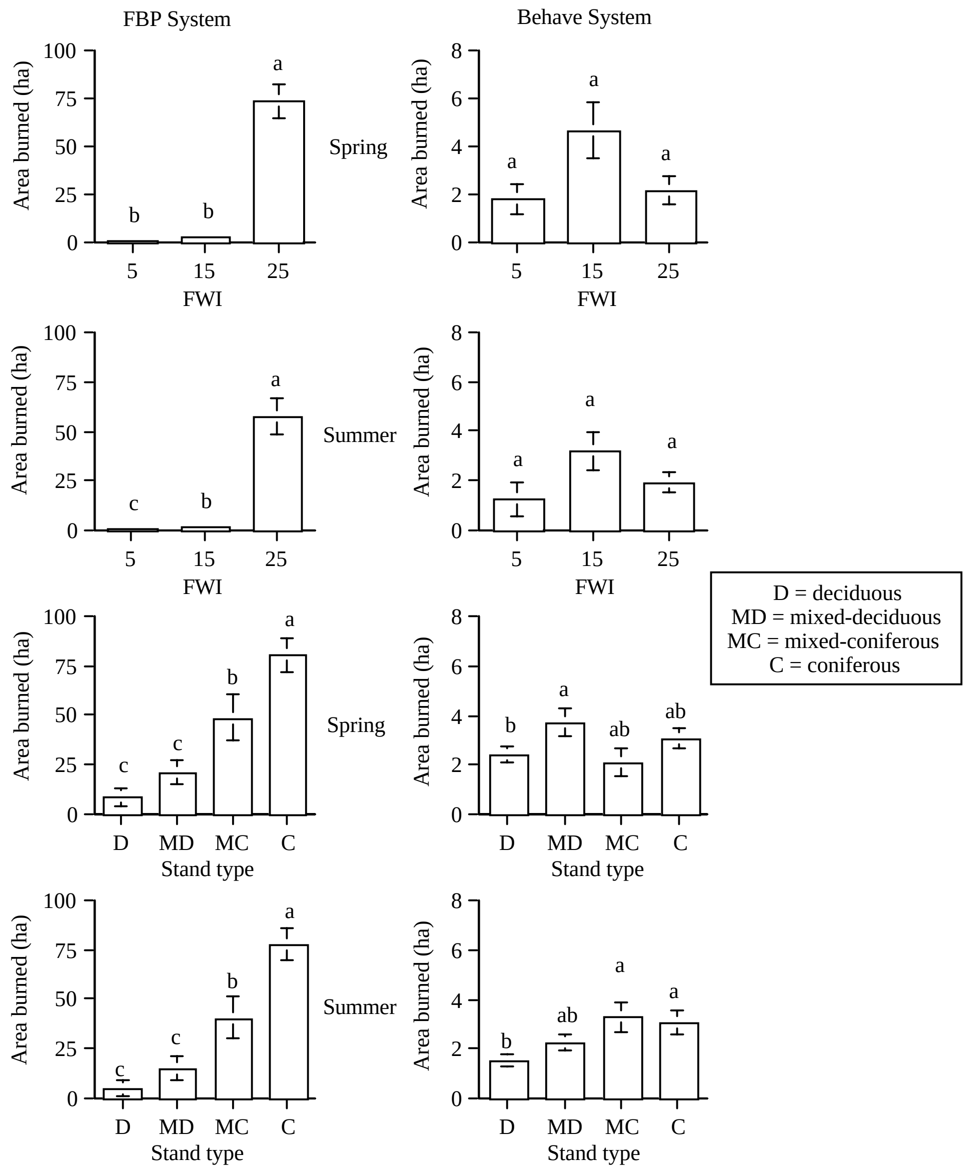

Figure 4.3 Differences in the simulated areas burned with the FBP and BEHAVE systems for spring and summer fires according to stand types and fire weather indices (FWI).

Kruskal-Wallis tests followed by Hsu's MCB tests (JUMP 1989) have been performed to analyse the differences. For each graph, areas burned with the same letter are not significantly different at $\alpha = 0.05$.

For the weather factor, the FBP system shows that the three fire behaviour variables have the highest significant values with extreme FWI for both seasons. Significant differences between low and intermediate FWI (5 and 15, respectively) only exist for the summer burned area. For the BEHAVE system, predicted ROS is the only variable that shows significant difference among FWI, with the moderate FWI creating the fastest ROS (twice as high as for other FWI) for both seasons. For both systems, spring fire behaviour always has faster ROS, higher HFI, and larger areas burned than with summer simulations.

For the vegetation factor (stand types), the FBP presents significant differences among the four stand types, for the HFI variable. There is no significant difference between mixed-coniferous (MC) and MD or C stand types for ROS, nor between deciduous (D) and mixed-deciduous (MD) for the burned area variable. In all cases, the highest values for the three variables are recorded for coniferous stands. The values decrease then from the coniferous to the deciduous stands with mixed stands at an intermediate position. The BEHAVE System gives significant differences between D and C stands for spring HFI and all summer predictions, whereas there is no significant difference between the two mixed stand types. In all cases, and for both systems, summer rate of spread and fire intensity are less important than rate of spread and fire intensity.

The comparisons of the fire behaviour components (ROS, HFI, and burned area) predicted by the FBP System between the four stand types for each FWI and two seasons are reported in Table 4.3. These comparisons show that only three of the eighteen models are not able to differentiate the four stand types (i.e., spring ROS for FWI = 5, spring and summer HFI for FWI = 15, respectively). For the fifteen significant models, there is always a decreasing response to the fire behaviour component from coniferous stands (highest values) to deciduous stands (lowest values). Several models show differences between deciduous, mixed, and coniferous stands, without showing differences between the two mixed types. Moreover, eight models (the best ones) significantly differentiate the four stand types, with two models dealing with a FWI of 5 (spring and summer HFI) and six models dealing with a FWI of 25 (ROS, HFI, and burned area both in spring and summer). According to these results, the idea that when extreme FWI values develop all vegetation types would burn comparably is not applicable to mixedwood boreal forests.

Table 4.3 Comparisons between the different stand types for each FWI and each fire behavior component for spring and summer FBP simulations

		FWI	Deciduous	Mixed-deciduous	Mixed-coniferous	Coniferous
Spring fires	ROS	5	0.38 ± 0.04 a	0.67 ± 0.08 b	1.03 ± 0.23 ab	1.31 ±0.21 a
		15	0.70 ± 0.04 a	1.01 ± 0.11 a	1.40 ± 0.36 a	1.77 ± 0.35 a
		25	3.88 ± 0.25 d	6.53 ± 0.34 c	9.73 ± 0.62 b	12.45 ± 0.47 a
	HFI	5	63.00 ± 3.00 d	143.31 ± 6.03 c	271.45 ±17.19 b	413.43 ± 17.28 a
		15	227.12 ± 29.32 a	486.22 ± 90.06 a	911.20 ± 332.15 a	1451.00 ± 393.34 a
		25	1320.20 ± 112.50 d	3064 ± 152.86 c	6513.90 ± 275.57 b	10512.60 ± 208.31 a
	Burned area	5	0.23 ± 0.06 c	0.69 ± 0.15 bc	1.50 ±0.60 ab	2.36 ± 0.67 a
		15	1.27 ± 0.11 b	2.81 ± 0.46 b	5.75 ± 2.05 a	8.21 ± 2.04 a
		25	23.30 ± 7.09 d	59.85 ± 9.63 c	138.9 ± 17.36 b	231.3 ± 13.13 a
Summer fires	ROS	5	0.22 ± 0.03 c	0.56 ± 0.07 b	0.94 ± 0.22 a	1.29 ± 0.21 a
		15	0.34 ± 0.04 c	0.73 ± 0.11 b	1.16 ± 0.39 ab	1.58 ± 0.38 a
		25	2.09 ± 0.24 d	5.29 ± 0.33 c	8.96 ± 0.60 b	12.21 ± 0.45 a
	HFI	5	37.34 ± 3.23 d	119.46 ± 6.13 c	253.43 ± 17.01 b	406.11 ± 18.04 a
		15	127.12 ± 21.06 a	388.32 ± 79.27 a	833.25 ± 317.13 a	1357.29 ± 376.03 a
		25	733.60 ± 116.79 d	2332.51 ± 158.72 c	5724.57 ± 286.08 b	10137.17 ± 216.26 a
	Burned area	5	0.00 ± 0.00 c	0.50 ± 0.11 bc	1.38 ± 0.53 b	2.36 ± 0.67 a
		15	0.27 ± 0.08 c	1.65 ± 0.38 bc	4.75 ± 1.89 b	9.00 ± 2.56 a
		25	8.10 ± 6.62 d	42.75 ± 8.99 c	115.9 ± 16.21 b	221.54 ± 12.26 a

Note: ROS = rate of spread (m/min); HFI = head fire intensity (kW/m); area burned in ha; Values in a row followed by the same letter are not significantly different at $\alpha = 0.05$ for the Hsu's test.

4.5.3. Simulations and experimental burns comparisons

The fire behaviour characteristics from the research burns and from the two simulation systems are reported in Table 4.4. The limited number of plots prevents us from conducting statistical tests to make comparisons. However, the values show that for the three fire behaviour variables, the FBP predictions are closer to the observed fire behaviour (and in the same order) than the BEHAVE predictions. In fact, the BEHAVE quantitative predictions are so low for all components that a minimum threshold that would sustain the fire seems to not have been reached. The FBP System seems to overestimate the head fire intensity component, but research burns show that within the mixed-coniferous stand type (plots #1 and #3) fire behaviour can be quite variable and likely explained by the FWI differences.

Table 4.4 Fire behaviour characteristics from the research burns and from the two simulation systems.

		Plot #1	Plot #2	Plot #3
Research burns				
	ROS (m/min)	8.84	12.86	3.72
	HFI (kW/m)	2236	3420	789
	Area burned(ha)	1	1	1
FBP predictions				
	ROS (m/min)	8.51	11.65	7.17
	HFI (kW/m)	3538	7522	1939
	Area burned(ha)	1.04	0.99	4.94
Behave predictions				
	ROS (m/min)	1.00	<0.1	<0.1
	HFI (kW/m)	121	15	12
	Area burned(ha)	<0.01	<0.01	<0.01

4.6. DISCUSSION

4.6.1. Fire behaviour prediction systems comparisons

By using two fire behaviour prediction systems we have found that there are two kinds of responses: qualitative and quantitative responses. The qualitative response of the systems deals with the respective effects of the involved factors in the fire behaviour predictions. For this qualitative aspect, the vegetation factor is significant in all simulations from BEHAVE and the FBP Systems, whereas the weather factor is not significant in the HFI and burned area predicted from the BEHAVE system. Moreover, the two systems assign different roles to these factors: the FBP System presents the weather as always the most important factor, whereas the BEHAVE System attributes most of the time the explained variance to the vegetation factor. The quantitative response of the systems deals with the predictions of fire behaviour components such as ROS, HFI, and burned area. The comparisons between the observed fire behaviour during the research burns and the quantitative predicted fire behaviour from the two systems revealed that the FBP predictions were very close to the observed values, whereas the BEHAVE predictions were so low (HFI less than 10 kW/m for two plots) that they would correspond to smoldering fires in deep organic layers (Van Wagner 1983). In fact, the BEHAVE System seems to not have achieved, in these simulations, a minimum threshold beyond which fire propagation could be sustained.

Among possible causes explaining differences between the two systems, and the fact that BEHAVE is not well adapted for quantitative fire behaviour predictions in the mixedwood boreal forest, are some weather and vegetation aspects.

Even though the precipitation is included differently in both systems, it cannot discriminate between them. Indeed, while the FBP System records the amount of rain for the last 24 hours and distributes it within different compartments through indices, BEHAVE records only the duration of rain and allocates it only to the top of the exposed aboveground fuels for which moisture content react faster than would the DMC and DC codes from the FWI (Van Wagner 1975). Moreover, rain effects in the BEHAVE System cannot readily wet the fuels to near saturation as in the FWI and FBP Systems (Van Wagner 1975). The FWI has then a longer record of precipitation through the DMC and the DC indices than the BEHAVE system does. Nevertheless, these facts would support higher fire behaviour for the BEHAVE system because fuels would dry faster than for the FBP, but this is not the case. Wind speed could be a discriminant weather component between both systems, as the moderate FWI seems to reveal. Indeed, the BEHAVE System shows consistently higher values of rate of spread, head fire intensity, and area burned at FWI 15 than at FWI 25. Nevertheless, the only significant difference between those two risks is the presence of a windy day (22 km/h) for FWI 15, whereas the extreme FWI has only light winds (9 and 7 km/h, respectively). The most likely explanation for the differences between the two systems and the quantitatively low predictions from BEHAVE could come from the vegetation factor, and more specifically from the fact that the BEHAVE system does not take into account the deeper duff layer. Indeed, the research burns have shown that the average depth of burn was deeper than the 0.5-cm litter layer. This could support the idea that the fuel amounts in the BEHAVE system would be underestimated and not recognized in supporting fire propagation. This would lead to low or unsustainable ROS, HFI, and burn areas, as shown by the results.

4.6.2. Role of vegetation and weather factors

The relative importance of the vegetation and weather factors depends on the fire behaviour prediction system used (as shown in this study), and on the studied region. Fires, in general, are more frequent and more severe in the western part of the Canadian boreal forest than in the eastern part, where the climate is moister and less favourable to wildfire propagation. Moreover, while boreal forest types are more segregated in western Canada, where aspen dominates the plains and conifers dominate the foothills, mixedwood forest types containing both conifer and hardwood species are more common in eastern Canada (Bergeron and Dubuc 1989; Bergeron and Dansereau 1993). Studying the relative importance of vegetation composition and weather conditions on fire behaviour in the western Canadian subalpine forest, Bessie and Johnson (1995) have found by using only the BEHAVE System that the weather was the most important factor explaining fire behaviour. In their study, the vegetation is only composed of conifer species, all being good fuels. Therefore, the warmer and drier western climate could be the real driving factor because the weather variability has always been higher than the vegetation variability. Conversely, our study shows that in the eastern Canadian mixedwood boreal forest the use of the BEHAVE System would lead us to present a different conclusion with the vegetation factor being the most important one for ROS, and the only significant factor for HFI and area burned. The presence of deciduous species, less prone to burn as compared to conifer species (Brown and Davis 1973), would induce higher vegetation variability than weather variability; therefore, vegetation composition could be the most important factor in fire behaviour for eastern Canadian mixedwood boreal forests. At best, future fire behaviour studies should acknowledge the regional natural variability that exists all over the continent in terms of climate and vegetation composition (Forestry Canada Fire Danger Group 1992; Harrington et al. 1991; Shugart et al. 1992). Different vegetation types and weather conditions interact together with the topography to create unique fire environments. It is important, then, to consider all environmental factors that interact with fire behaviour (Agee 1997) before regionally adapting the relevant model (i.e., the FBP for the mixedwood boreal forest of eastern Canada) to find out the potential predominant factors. In mountainous regions, topography could be as much of a driving factor as fuel type or weather because slope is known to have important effects on ROS and on burned patches, sizes and shapes (Andrews 1986; Forestry Canada Fire Danger Group 1992; Fryer and Johnson 1988; Jean 1992; Johnson 1992).

If the FBP System is adapted as the relevant model for the mixedwood boreal forest of eastern Canada, both vegetation and weather factors have a significant effect, but the weather is the most important factor because it explains the maximum of the variance (Table 4.2). Nevertheless, the vegetation factor has a significant influence on fire behaviour components whatever the FWI conditions (Table 4.3). This shows that the small amount of explained variance attributed to the vegetation factor could, however, correspond to a more important effect of stand types on fire behaviour. Moreover, the natural stand variability is higher in the studied forest mosaic than shown in the analyses, and it could have a greater influence on fire behaviour. Indeed, stands with *Picea mariana* or *Pinus banksiana* on exposed bedrock or with *Larix laricina* or *Fraxinus nigra* on silted lowland sites subject to flooding and boggy habitats, respectively (Bergeron and Dubuc 1989), are present in the patchiness of the forest mosaic but were not included in the study. The idea that slight differences between forest stands in fuel characteristics are insignificant when compared with the large short-term variations in the weather (Johnson 1992) should not be applied at least to the mixedwood boreal forest. The same is true of the idea that above a certain extreme fire weather risk all vegetation types will burn similarly. Our results show that deciduous tree dominated stands would support less intense fires, which would burn smaller areas, than fires in conifer tree dominated stands. Several studies dealing with large burned areas (Johnson 1992; Payette 1992; Van Wagner 1983) have reported conifer forest mosaics, such as the black spruce-feathermoss forest, where the deciduous species and stands are poorly represented. However, a recent study (Bergeron et al. 1999) has shown that fires burning large areas were proportionately more numerous in the black spruce-feathermoss forest than in the mixedwood boreal forest, and this despite similar regional climatic conditions (Hofgaard et al. 1999). These results are consistent with ours, and they show that the conifer composition may have a direct effect on the area burned at the stand and the landscape level. Also, the head fire intensity increases with the increase of the conifer proportion in a stand. Therefore, fires in conifer stands or in landscape mosaics dominated by conifer stands should also be more severe.

The fire season is another important factor to take into account. The ANOVA results have shown that season does not change the order of importance of the two factors, but it generally increases the vegetation influence during summer simulations. Moreover, the season is responsible for lower fire behaviour values recorded in the summer simulations than in spring simulations. The higher the deciduous tree percentage in a stand, the higher spring and summer differences. However, the only difference recorded between spring and summer is exclusive to the vegetation factor with the presence or absence of the deciduous fuel type (Forestry Canada Fire Danger Group 1992). In spring, the absence of deciduous tree leaves and herbaceous foliage don’t interfere with the direct ground and surface fuel bed warming from the direct sunlight (Furyaev et al. 1983), whereas in summer deciduous leaves intercept the sunlight and create a cooler and moister understory and ground environment (Van Wagner 1983). This explains the slower ROS in summer associated with less intensity released and smaller burned areas. Generally, conifer stands in this region are older and present a more open canopy than deciduous stands. Because these conifer species are evergreen, there will be little or no difference between spring and summer foliage or between the different seasonal fire behaviour as shown by the mixed-coniferous and coniferous stands. When a difference exists, it is the result of an herb cover presence in summer that retains higher moisture content in the surface fuels and decreases the propagation rate. This ground cover does not take into account a moss layer since this fuel type does not exist in these stand types (De Grandpré et al. 1993). In this way, deciduous phenological changes explain the increased importance of the vegetation factor for summer fires through the indirect season factor. Wotton and Flannigan (1993) have shown that global warming may induce an increase in the length of the fire season, and this may have some consequences on the number and size of fires during the fire season. Indeed, the fire season would start earlier because many stands would still be leafless, and it would end later because stands with senescent foliage provide good conditions for fire propagation.

4.7. CONCLUSION

This research has shown the importance of considering all the environmental factors of fire before selecting the appropriate fire behaviour prediction system for any given area. The use of experimental burns, when available, can be helpful to choose the best system. The results of this study have denied the hypothesis related to the greater accuracy of the BEHAVE System. In fact, the BEHAVE System is not well adapted to the mixedwood boreal forest to predict realistic quantitative fire behaviour, whereas the FBP system seems to be an efficient fire behaviour prediction system for this boreal ecosystem. The slight overestimation of the head fire intensity prediction from the FBP System should be improved by adjusting the model through the use of future prescribed burn data. Differences in fire behaviour in relation to stand composition are significant for all fire behaviour variables (ROS, HFI, and burned area). This implies that the mixedwood boreal forest, with its natural stand variability in terms of its proportion of conifers, is a complex ecosystem with respect to fire disturbance. This ecosystem can indeed present several fire behaviours at the local scale, but the idea that when extreme FWI values develop all vegetation types would burn comparably is not applicable to mixedwood boreal forests. This inherent variability needs to be integrated in any fire behaviour prediction system that aims to predict final size and shape of wildfires at the landscape level.

4.8. ACKNOWLEDGMENTS

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4.9. REFERENCES

Aber, J.D. and Melillo, J.M. 1991. *Terrestrial ecosystems.* ed. by Saunders College Publishing, New York.

Agee, J.K. 1997. The severe weather wildfire - too hot to handle? Northwest Science **71** (1): 153-156.

Andrews, P.L. 1986. *BEHAVE: fire behavior prediction and fuel modeling system--BURN subsystem, PART 1*. Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT-194: 130p.

Andrews, P.L. and Chase, C.H. 1989. *BEHAVE: fire behavior prediction and fuel modeling system--BURN subsystem, PART 2*. Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT-260: 93p.

Bergeron, Y. 1991. The influence of island and mainland lakeshore landscapes on boreal forest fire regimes. Ecology **72** (6): 1980-1992.

Bergeron, Y. and Archambault, S. 1993. Decreasing frequency of forest fires in the southern boreal zone of Quebec and its relation to global warming since the end of the "Little Ice Age". The Holocene **3** (3): 255-259.

Bergeron, Y. and Dansereau, P.R. 1993. Predicting the composition of Canadian southern boreal forest in different fire cycles. J. Veg. Sci. **4** : 827-832.

Bergeron, Y. and Dubuc, M. 1989. Succession in the southern part of the Canadian boreal forest. Vegetatio **79**: 51-63.

Bergeron, Y., Gauthier, S., Carcaillet, C., Flannigan, M.D., Prairie, Y. and Richard, P.J.H. 1999. Variability in fire frequency and forest composition in Canada's southeastern boreal forest: a challenge for sustainable forest management. *Proceedings from* The sustainable forest management network. Science and practice: sustaining the boreal forest. Edmonton, Alberta, Canada: 74-80.

Bessie, W.C. and Johnson, E.A. 1995. The relative importance of fuels and weather on fire behavior in subalpine forest. Ecology **76** (3): 747-762.

Blais, J.R. 1983. Trends in the frequency, extent and severity of spruce budworm outbreaks in eastern Canada. Can. J. For. Res. **13** : 539-547.

Bradshaw, L.S., Deeming J.E., Burgan, R.E. and Cohen, J.D. 1983. *The 1978 National Fire Danger Rating System: technical documentation*. Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT-169: 41p.

Brown, A.A, and Davis, K.P. 1973. Forest fire: control and use. Second edition. McGraw-Hill New York: 686p.

Brown, J.K., Oberheu, R.D. and Johnston, C.M. 1982. *Handbook for inventoring surface fuels and biomass in the Interior West*, Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT-129: 48p.

Burgan, R.E. and Rothermel, R.C. 1984. *BEHAVE: fire behavior prediction and fuel modeling system--FUEL subsystem*. Ogden, UT. Interm. For. Range Exp. Stn., U. S. Dep. Agric., For. Serv. Gen. Tech. Rep. INT-167: 126p.

Canadian Forestry Service. 1987. Canadian forest fire danger rating system - user’s guide. Produced by the Canadian Forestry Service fire danger group. Three-ring binder (unnumbered publication).

Cogbill, C.V. 1984. Dynamics of the boreal forest of the Laurentian highlands, Canada. Can. J. For. Res. **15** : 252-261.

Conover, W.J. and Iman, R.L. 1981. Rank transformations as a bridge between parametric and nonparametric statistics. The American Statistician **35** (3): 124-129.

Dansereau, P.R. and Bergeron, Y. 1993. Fire history in the southern boreal forest of northwestern Quebec. Can. J. For. Res. **23** : 25-32.

De Grandpré, L., Gagnon, D. and Bergeron, Y. 1993. Changes in the understory of Canadian southern boreal forest after fire. J. Veg. Sci. **4** : 803-810.

Dodge, M. 1972. Forest fuel accumulation - a growing problem. Science **177** : 139-142.

Environment Canada. 1993. Canadian climate normals 1961-90. Canadian Climate Program. Environ. Can. Atmos. Environ. Serv. Downsview, Ontario.

Flannigan, M.D. and Harrington, J.B. 1988. A study of the relation of meteorological variables to monthly provincial area burned by wildfire in Canada (1953-80). J. Appl. Meteorol. **27** : 441-452.

Forestry Canada Fire Danger Group. 1992. *Development and structure of the Canadian Forest Fire Behavior Prediction System*. Ottawa, Ontario, Forestry Canada.

Fryer, G.I. and Johnson, E.A. 1988. Reconstructing fire behavior and effects in a subalpine forest. J. Appl. Ecol. **25** : 1063-1072.

Furyaev, V.V., Wein, R.W. and MacLean, D.A. 1983. Fire influences in Abies-dominated forests. In: Wein R. W., MacLean, D. A. (eds). *The role of fire in Northern circumpolar ecosystems*; Scope 18: 221-234. Wiley, New York.

Harrington, J., Kimmins, J., Lavender, D., Zoltai, S. and Payette, S. 1991. The effect of climate change on forest ecology in Canada. 10ème congrès forestier mondial, Paris.

Heinselman, M.L. 1973. Fire in the virgin forest of the boundary waters Canoe Area, Minnesota. Quaternary Research **3** : 329-382.

Hirsch, K.G. 1996. *Canadian Forest Fire Behavior Prediction (FBP) System: user's guide*. Nat. Resour. Can., Can. For. Serv., North. For. Cent., Edmonton, AB. Spec. Rep. 7.

Hofgaard, A., Tardif, J. and Bergeron, Y. 1999. Dendrometric response of *Picea mariana* and *Pinus banksiana* along a latitudinal gradient in the eadtern Canadian boreal forest. Can. J. For. Res. **29** (9): 1333-1346.

Holling, C.S. 1992. The role of forest insects in structuring the boreal landscape. In: H. H. Shugart, R. Leemans, and G.B. Bonan (eds). *A systems analysis of the global boreal forest*. Cambridge Univ. Press, Cambridge: 171-191.

Jean, F. 1992. Modélisation du comportement du feu. Influence de la pente et de la charge d'une litière d'aiguilles de pin maritime. Avignon, Institut National de la Recherche Agronomique. PIF 9205.

Johnson, E.A. 1992. *Fire vegetation dynamics: Studies from the North American boreal forest.* Cambridge.

JUMP 1989. Statistical visualization for the Macintosh. Version 2.0.4. SAS Institute, Inc., Cary, NC.

Levine, E.R., Ranson, K.J., Smith, J.A., Williams, D.L., Knox, R.G., Shugart, H.H., Urban, D.L. and Lawrence, W.T. 1993. Forest ecosystem dynamics : linking forest succession, soil process and radiation models. Ecol. Model. **65** : 199-219.

McRae, D.J., Alexander, M.E. and Stocks, B.J. 1979. *Measurement and description of fuels and fire behavior on prescribed burns: a handbook*. Can. For. Serv., Great Lakes For. Res. Cent. Sault Ste. Marie, Ont. Rep. O-X-287: 44p.

Morin, H. 1995. Importance et évolution des épidémies de la tordeuse des bourgeons de l'épinette dans la zone boréale au vingtième siècle. In: *En savons nous assez pour lutter efficacement contre la tordeuse?* : 92-95.

Mueller-Dombois, D. and Ellenberg, H. 1974. *Aims and methods of vegetation ecology*. Wiley, New York.

Payette, S. 1992. Fire as controlling process in the Northern American boreal forest. In: H. H. Shugart, R. Leemans, G.B. Bonan (eds). *A systems analysis of the global boreal forest.* Cambridge Univ. Press: 144-169.

Pickett, S.T.A. and White, P.S. 1985. *The ecology of natural disturbance and patch dynamics*. Orlando, California.

Rowe, J.S. 1972. *Forest regions of Canada*. Environment Canada, Ottawa, Ontario. Publ. 1300.

SAS Institute Inc. 1985. SAS user's guide: statistics, version 6 edition. SAS Institute Inc., Cary, N.C.

Schimmel, J. and Granström, A. 1997. Fuel succession and fire behavior in the Swedish boreal forest. Can. J. For. Res. **27** : 1207-1216.

Shugart, H.H., Leemans, R. and Bonan, G.B. 1992. *A systems analysis of the global boreal forest.* Cambridge Univ. Press, Cambridge.

Simard, A.J. 1973. *Forest fire weather zones of Canada*. Environ. Can., Can. For. Serv., Ottawa, Ont. map with text.

Stocks, B.J. 1987. Fire behavior in immature jack pine. Can. J. For. Res. **17**: 80-86.

Stocks, B.J., Lawson, B.D., Alexander, M.E., Van Wagner, C.E., McAlpine, R.S., Lynham, T.J., and Dubé, D.E. 1989. Canadian Forest Fire Danger Rating System: an overview. For. Chron. **65** (4): 258-265.

Turner, M.G. and Romme, W.H. 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology **9** (1): 59-77.

Van Wagner, C.E. 1968. The line intersect method in forest fuel sampling. For. Sci. **14**: 20-26.

Van Wagner, C.E. 1975. A comparison of the Canadian and the American forest fire danger rating systems. Environ. Can., Can. For. Serv., Petawawa For. Exp. Stn., Chalk River, Ontario, Inf. Rep. PS-X-59: 22p.

Van Wagner, C.E. 1983. Fire behaviour in northern conifer forests and shrublands. In: R.W. Wein and D.A. Maclean (eds). *The role of fire in Northern Circumpolar Ecosystems.* SCOPE 18, John Wiley and Sons, Chichester, England: 65-81.

Vincent, J.S., and Hardy, L. 1977. L'évolution et l'extension des lacs glaciaires Barlow et Ojibway en territoire québecois. Géographie Physique et Quaternaire **31**: 357-372.

Wein, R.W. and Moore, J.M. 1977. Fire history and rotations in the New Brunswick Acadian forest. Can. J. For. Res. **7** : 285-294.

Wotton, B.M. and Flannigan, M.D. 1993. Length of the fire season in a changing climate. For. Chron. **69** (2): 187-192.

Wright, H.A., and Bailey, A.W. 1982. *Fire ecology*. Wiley, New York.

CHAPITRE V

MODELING TREE MORTALITY FOLLOWING WILDFIRE IN THE SOUTHEASTERN CANADIAN MIXEDWOOD BOREAL FOREST

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5.1. RÉSUMÉ

La mortalité des espèces d'arbres de la forêt boréale mixte a été analysée trois mois après le feu naturel survenu à Val Paradis (Québec, Canada). Cette étude se base sur l'échantillonnage de 1963 arbres répartis dans 36 peuplements ayant subi des conditions d'intensité de feu très variées. Les résultats suggèrent que la composition des peuplements influence l'intensité du feu. En effet, les peuplements feuillus ont brûlé moins intensément que les peuplements mixtes et conifères. L'analyse des taux de mortalité spécifiques montre que *Populus tremuloides* Michx. est l'espèce la moins résistante au feu, comparé à *Picea mariana* Moench et *Pinus banksiana* Lamb. Chez ces deux conifères, des interactions efficaces existent, entre la résistance cambiale et celle du houppier, pour résister aux dommages occasionnés par le passage du feu. A l'opposé, le peuplier ne présente qu'une résistance au niveau du houppier. La hauteur de la base du houppier est une variable active dans la résistance au feu des conifères lorsque le houppier est bas car celui-ci protège le tronc et le cambium de la chaleur directe par contact des flammes. Toutefois, au delà d'une certaine intensité maximale, tous les arbres de toutes les espèces sont morts trois mois après le feu. Pour chaque espèce, la régression logistique la plus performante, prédisant la probabilité de mortalité causée par le feu à partir des variables dendrométriques et des variables décrivant le feu, utilise le diamètre à hauteur de poitrine (DHP) ou la hauteur totale de l'arbre et la hauteur de charbon sur le tronc. Le coefficient de Kappa, utilisé pour valider les modèles, révèle que ceux-ci sont plus performants que des modèles utilisant uniquement les variables dendrométriques. Des nomogrammes ont été créés à partir de la régression logistique trouvée pour chaque espèce dans le but d'aider les aménagistes forestiers dans l'exercice de certaines pratiques sylvicoles.

5.2. ABSTRACT

We modeled tree mortality occurring three months following a wildfire in the mixedwood boreal forest (Quebec, Canada) using data from 1963 trees in 36 stands burned under a wide range of fire intensity conditions during the 1997 Val Paradis fire. Stand composition influenced the fire intensity: deciduous stand were less intensely burned than mixed and coniferous stands. Analysis of species mortality rates revealed that *Populus tremuloides* Michx. was the less fire-resistant species, whereas *Picea mariana* Moench and *Pinus banksiana* Lamb. were the most resistant species. Efficient interactions for conifers exist between crown and cambial resistance to injury and fire behaviour, whereas aspen only shows crown resistance. Crown base height is a potential efficient fire-resistance variable when the base is low, because it protects the stem and cambium from direct-flame heating. However, beyond threshold fire intensity, all trees died three months after the fire occurred. The best logistic regressions, relating probability of wildfire-induced mortality to tree morphological and fire variables, used diameter at breast height (DBH) or total tree height and char height. Kappa coefficients used for model validations revealed that they were very efficient as compared to those based only on morphology variables. These logistic models were used to create species nomograms that can assist land managers in attaining silvicultural objectives.

5.3. INTRODUCTION

Fire is a major environmental factor in boreal forests (Johnson 1992, Flannigan 1993, Bergeron et al. 1999), but little published information exists on patterns of tree mortality following fire. However, managers need to be able to predict fire-caused tree mortality for different objectives. First, they need to plan post-fire management with salvage operations after wildfires (Ryan et al. 1988). Secondly, the use of effective prescribed fires (Brown and Debyle 1987) is another type of forest management practice that requires plans specifying acceptable levels of mortality in standing trees and describing desired fire behaviour, particularly flame length or frontal fire intensity (Van Wagner 1973, Reinhardt and Ryan 1988).

Most of the published studies on tree mortality hasdealt with coniferous species such as *Pseudotsuga menziesii* (Ryan et al. 1988, Peterson and Arbaugh 1989, Keane et al. 1990), *Pinus ponderosa* (Thomas and Agee 1986, Ryan 1998, Keane et al. 1990, Swezy and Agee 1991, Harrington 1993, Regelbrugge and Conard 1993), *Pinus contorta* (Peterson and Arbaugh 1986, Reinhardt and Ryan 1988, Swezy and Agee 1991), and *Picea engelmannii* (Ryan 1998). Several fir species were also studied such as *Abies lasiocarpa* by Peterson (1985) and Ryan and Reinhardt (1988), and *Abies concolor* by Thomas and Agee (1986) and Swezy and Agee (1991). Few deciduous species have been selected for fire-caused tree mortality studies: *Populus tremuloides* has been studied by Brown and Debyle (1987), while some oaks associated with maples (Huddle and Pallardy 1999) or with pines (Harmon 1984, Regelbrugge and Conard 1993) were also considered in mortality studies.

Tree mortality has often been modeled as a function of tree size and fire damage to foliage, stem, and roots (Peterson and Arbaugh 1986, Brown and Debyle 1987, Regelbrugge and Conard 1993, Ryan 1998). Among the different injury types that can kill trees, crown injury is the most commonly observed fire effect, whereas little is known about the extent to which roots are affected by fires (Ryan 1998). However, combined damage should be taken into account to increase the accuracy of mortality predictions. The height or percent crown scorched have been significant predictors of tree mortality for many species and models (Peterson 1985, Ryan and Reinhardt 1988). Indirect representation of bole damages such as bark charring (percent circumference charred, char depth or char height), has also been found to increase crown scorch as a predictor of mortality (Peterson and Arbaugh 1986, 1989, Ryan and Reinhardt 1988). According to several authors (Brown and Debyle 1987, Finney and

Martin 1993, Ryan 1998), tree diameter is linearly correlated to the bark thickness above about 50 cm on the bole. Because resistance to cambium injury increases with the square of the bark thickness, fire injury resistance is expected to increase with the square of stem diameter (Ryan 1998). However, several studies have successfully used diameter as a linear term in their models (Brown and Debyle 1987, Ryan et al. 1988, Finney and Martin 1993, Harrington 1993), while studies have shown that tree mortality was not correlated to tree diameter for some species, or above a diameter threshold (Ryan 1998). Thus mortality may be independent of diameter for some species, or for small diameter trees (Gutsell and Johnson 1996). Moreover, diameter is correlated to tree height, and height to crown base is, in part, a function of the tree species and of the site conditions (open versus closed) (Ryan 1998). The greater the tree size, the greater the portion of the crown is high enough to avoid direct scorching by the flame or scorching by hot gases (Ryan et al. 1988).

In contrast, few direct relationships between fire behaviour characteristics and effects on trees have been developed. Van Wagner (1973) derived a relationship between fireline intensity and height of convective crown scorch on conifers. Reinhardt and Ryan (1988) estimated tree mortality of the northwestern United States conifer species from underburning experiments, and they created nomograms relating tree mortality to fireline intensity and to flame length. Ryan and Frandsen (1991) related cambium mortality of Rocky Mountain conifers to fuel loading and consumption variables. Brown and Debyle (1987) developed models for mortality of small trees based on artificial fireline intensity treatments and tree diameter at ground level.

In the eastern Canadian mixedwood boreal forests (Rowe 1972), the main tree species are balsam fir (*Abies balsamea* (L.) Mill.), white birch (*Betula papyrifera* Marsh.), black spruce (*Picea mariana* (Mill.). B.S.P.), Jack pine (*Pinus banksiana* Lamb.), and trembling aspen (*Populus tremuloides* Michx.). However, these species have never quite been considered of interest for fire-caused tree mortality research (excepted *Populus tremuloides* by Brown and Debyle (1987)), even though they are key species for the natural forest dynamics. Moreover, some of them like aspen, Jack pine, and black spruce represent valuable species for the forest industry in eastern Canada. The objectives of this study were fourfold: first we wanted to verify that the effect of stand composition influenced fire intensity. Indeed, Hély et al. (submitted), using two different fire behaviour prediction systems, have shown that deciduous stands sustained less intense fire than mixed or conifer stands. Secondly, we wanted to analyze the effects of fire behaviour on mortality by comparing the species mortality rates a few months after the fire has occurred, with the hypothesis that deciduous species are less fire-resistant than the conifer species. Thirdly, we wanted to determine which mensurational variables and fire damage indicators were important in predicting tree mortality by the use of logistic regression models. Probability of mortality was hypothesized to be negatively correlated with tree size descriptors, and positively correlated with fire-injury descriptors. Finally, we wanted to develop nomograms as tools that managers could use to evaluate tree mortality after a wildfire, in the perspective of timber slavaging, or to keep tree mortality within specifications during a planned prescribed burn.

5.4. MATERIALS AND METHODS

5.4.1. Study area

The wildfire of Val Paradis (49° 09’ N, 79° 26’ W, Nothwestern Quebec, Canada) spread during two days in June 1997, and it covered 12557 ha of mixedwood boreal forest. The principal species involved in this fire were *Populus tremuloides*, *Picea mariana*, and *Pinus banksiana,* with *Abies balsamea* and *Betula papyrifera* as secondary species. The meteorological data during these two days and the corresponding Fire Weather Indices (Van Wagner 1987) were provided by the SOPFEU (Society for the protection of forests against fire in Quebec) and are reported in Table 5.1. The fire was an intermittent crown fire, where most of the area was burned during the second day. The fire was extinguished on the third day.

Table 5.1 Weather data and fire weather indexes for June 9 and 10, 1997 from the Saint-Gilles meteorological station

Day	Temperature (° C)	Relative humidity (%)	Wind speed (km/h)	Wind direction (degrees)	FFMC	DMC	DC	ISI	BUI	FWI
June 9	27.8	37	9	270	92	59	131	9	59	22
June 10	28.9	37	20	320	92	64	140	11	64	27

Note: FFMC = Fine Fuel Moisture Content; DMC = Duff Moisture Content; DC = Drought Code; ISI = Initial Spread Index; BUI = Buildup index; and FWI = Fire Weather Index.

5.4.2. Data collection

Thirty-six permanent plots (0.04 ha each) were set up within the area burned during the second fire day to provide tree mortality information. Plots were characterized as being deciduous, mixed or coniferous types according to the coniferous and deciduous basal areas. These plot types encompassed homogeneous structure and composition characteristics (Table 5.2). In this study, the fire severity is defined as the impacts on the vegetation more than the impacts on soil (such as depth of burn and exposed percentage of mineral soil). The fire severity among plots presented a wide range from 0 to 100% tree death to have a good representation of the natural variability within a fire. We classified the thirty-six plot burns as either being low, moderate, or extreme fire severity, if stand mortality was less than 25%, between 25 and 75%, or more than 75%, respectively. In each plot, we recorded all trees with diameter at breast height (DBH) greater than 5 cm. We recorded the following variables: species, DBH, total tree height (TOTH) and crown base height (CBH), status (live or dead from the fire), char height (CH) on the stem at the four orientations, orientation of maximum and minimum char heights, and bark thickness (BT). We sampled at the end of the first growing season following the fire (three months after the fire occurred), and it was difficult to distinguish foliage killed by the fire from foliage that had simply died some time before the fire. The crown status (live or dead) was used to determine if the tree was still alive or dead.

The head fire intensity (I in kilowatts per meter) was calculated for each tree using the following series of equations. In the first step, the wind speed above the canopy (Equation 5.1) and on the ground (Equation 5.2) were calculated (Cionco 1978, Amiro and Davis 1988) using the airport wind speed ($u_{airport}$ = 6 m/s), recorded at 10 meters above ground ($z_{airport}$):

$$\frac{u_h}{u_{airport}} = \frac{\ln((h-d)/z_0)}{\ln((z_{airport}+h-d)/z_0)} \qquad (5.1)$$

Table 5.2 Prefire basal area and species tree density (Mean and Standard Deviation) for the 36 sampled plots burned in the Val Paradis fire

Stand type	Species	Basal area (m^2 / ha)		Density (stems / ha)	
		Mean	SD	Mean	SD
Deciduous	*Abies balsamea*	0.33	0.33	42	42
	Betula papyrifera	2.48	2.46	181	179
	Pinus banksiana	0.23	0.20	8	6
	Picea glauca	0.06	0.06	2	2
	Picea mariana	2.53	1.09	188	75
	Populus tremuloides	27.83	3.16	705	99
	Total coniferous	**3.15**	N/A [a]	**240**	N/A
	Total deciduous	**30.31**	N/A	**886**	N/A
	Stand total	**33.46**	**3.08**	**1126**	**137**
Mixed	*Abies balsamea*	0.67	0.37	58	15
	Betula papyrifera	0.05	0.05	8	0
	Pinus banksiana	2.55	1.57	56	44
	Picea glauca	0.00	0.00	0	0
	Picea mariana	13.72	2.07	585	81
	Populus tremuloides	19.21	2.49	458	84
	Total coniferous	**16.94**	N/A	**699**	N/A
	Total deciduous	**19.26**	N/A	**466**	N/A
	Stand total	**36.20**	**2.51**	**1167**	**121**
Coniferous	*Abies balsamea*	0.00	0.00	0	0
	Betula papyrifera	0.00	0.00	0	0
	Pinus banksiana	8.89	1.97	623	137
	Picea glauca	0.00	0.00	0	0
	Picea mariana	20.00	2.45	2021	294
	Populus tremuloides	0.02	0.02	2	2
	Total coniferous	**28.90**	N/A	**2644**	N/A
	Total deciduous	**0.02**	N/A	**2**	N/A
	Stand total	**28.92**	**1.73**	**2646**	**276**

Note: [a] N/A = not applicable.

and

$$u_z = u_h * e^{(a(\frac{z}{h}-1))} \qquad (5.2)$$

where u is the wind speed (in meters per second) at any height, z, in the canopy, h is the canopy height (in meters), d = 0.7 * h, z_0 = 0.13 * h, and a is an attenuation coefficient or the canopy flow index. We used the values found by Amiro (1990) for a (3.2, 2.6, and 4.8 for *Populus tremuloides*, *Pinus banksiana*, and *Picea mariana* stands, respectively), and we assumed constant wind speed ($u_z = u_{0.6h}$) from ground surface to 0.6h (Alexander 1998).

In the second step, the flame angle (A) in degrees, measured between the flame front or flame plume and the unburned fuelbed, was related to the wind speed on the ground (Alexander 1998):

$$\tan A = (0.82 * (9.8 h_f / u^2)^{0.5}) \qquad (5.3)$$

where h_f is the flame height in meters, and u is the above-ground wind speed (i.e., u_z) in meters per second. We assumed that flame height was equivalent to maximum char height. When A was extracted from Equation 3, the fire intensity was calculated using the Alexander (1982) relationships:

$$L = h_f / \sin A \qquad (5.4)$$

and

$$I = 259.833 * L^{2.174} \qquad (5.5)$$

where L is the flame length in meters, and I is the surface fire intensity. We didn't calculate crown fire intensities as recommended by Alexander (1998) because we found that adding one-half of the canopy height to L resulted in unrealistic fire intensities (between 10000 and 90000 kW/m) as compared to those usually related to intermittent crown fires (5000 to 10000 kW/m).

5.4.3. Data analysis

Analyses include only *Pinus banksiana*, *Picea mariana*, and *Populus tremuloides* because the other species were underrepresented (n = 49 trees for *Abies balsamea*) or presented only dead trees (such as *Betula payrifera*).

Effect of stand type and species composition on fire intensity

We analyzed the respective role of stand types and species composition on fire intensity using a two-way ANOVA design on rank scores (Conover and Iman 1981) with the General Linearized Model procedure (SAS Institute Inc. 1985). We tested both factor effects and their potential interaction. We also looked at the differences in the fire intensity among the three stand types and the three species using a Kruskal-Wallis test followed by a Hsu's MCB test (JUMP IN 1989).

Heterogeneity tests for species mortality rates

We performed non-parametric likelihood ratio tests, also called G tests or heterogeneity tests (Scherrer 1984), to study the differences among species mortality rates. For all species, each tree was classified into one of the four surface fire intensity classes (< 100, 100-1000, 1000-5000, and >5000 kW/m, respectively). The G tests were performed within each fire intensity class. The G test is based on a Chi-square distribution when the sample size is large. When significant differences were found a multiple comparisons test (simultaneous test procedure in Scherrer 1984) was performed to find out the significantly different species groups.

Characteristic tree variables potentially explaining the fire-caused mortality

Wilcoxon-Mann-Whitney tests were performed on the morphological variables (DBH, TOTH, CBH, and mean BT, respectively) and on maximum char height (MACH) for each species. The calculated fire intensity was categorized within 4 classes (<100, 100-1000, 1000-5000, and >5000 kW/m, respectively) for each species. Results of comparisons were plotted to find out within each fire intensity class the variables showing significant differences between dead and live trees. A characteristic variable is a variable that presents, for a given species, several significant differences along the fire intensity gradient. Interactions between fire behaviour and dendrometric variables exist when MACH and at least one dendrometric variable are simultaneously characteristic variables.

Logistic regression models for tree species mortality prediction

Tree mortality data are binary (dead or live trees) and were thus modeled using logistic regression analysis (Peterson and Ryan 1986, Ryan et al. 1988, Swezy and Agee 1991, Finney and Martin 1993, Harrington 1993, Regelbrugge and Conard 1993, Mutch and Parsons 1998). Logistic equations predict the probability of the occurrence of an event, such as mortality of an individual tree, based on a number of fire variables, fire effects, or tree-feature predictors (Finney and Martin 1993). Discriminant analysis could also be an efficient statistical analysis to use with binary dependent variables, but logistic regression models are more adapted when some of the dependent variables are qualitative (Press and Wilson 1978), and it is independent of the assumption of multivariate normality (Daniels et al. 1979). Logistic regression analysis was used to model the probability of postfire tree mortality for each species, as a function of tree size and fire injury. The logistic regression model has the following form:

$$P(m) = \frac{1}{1 + e^{-(b0 + b1x1 + \ldots + bkxk)}} \qquad (5.6)$$

where P(m) is the probability of postfire mortality, X_1 through X_k are independent variables, and b_0 through b_k are model parameters estimated from the data. Variables were considered independent and potential inputs when the Pearson's correlation coefficient between two given variables was less than 0.5. We didn't use the significant probability information because the species sample sizes are very large (more than 300) and numerous variables could then be significantly correlated although they had a small correlation coefficient. The data set for each species was randomly divided in two parts. The first part, corresponding to 75% of the specific total data set, was used to create the logistic model. The SAS logistic procedure (SAS Institute Inc. 1985) was used to perform the stepwise regression and to obtain the maximum likelihood estimate of the model. A generalized Wald’s statistic was performed to test regression coefficients of the model. The model goodness-of-fit was first assessed by the Hosmer and Lemeshow test (SAS Institute Inc. 1985). To further evaluate the performance of the species mortality model, the remaining 25% part of the species data set was used *a posteriori* to validate the model. The comparison between the predicted probabilities and the observed outcomes was performed using the Kappa test (Jensen 1996).

First, to find out if species present natural intrinsic sensitivity, we analyzed tree species mortality using only the morphological variables (DBH, TOTH CBH, the CBH / TOTH ratio (CR), and BT (mean (MEBT), minimum (MIBT), or maximum (MABT)). We then added the fire characteristic variables such as stand fire severity (SEV), calculated intensity on each tree (INT), and char height mean (MECH), minimum (MICH), or maximum (MACH)) to the above dendrometric variables.

Nomograms

Species nomograms for *Pinus banksiana*, *Picea mariana*, and *Populus tremuloides* were created from the logistic regression models using tree and fire characteristics. Each species nomogram is a series of five graphs (six for *Populus tremuloides*). The upper left graph presents the mortality probability chart relating tree characteristics (DBH or TOTH) to the char height (MACH, MICH, or MECH). The char height was then related to the fire characteristics such as calculated frontal fire intensity or flame length (lower right quadrant) according to Alexander's (1982, 1998) equations by assuming that the maximum char height was equivalent to the flame height. We calculated the flame length and the frontal fire intensity for a flame angle ranging between 40 and 90°; the flame angle depends on the wind speed. The mean flame angle, calculated from the airport weather conditions, was 82° during the Val Paradis fire progression, which corresponds to a light wind speed within the stand. We only used the flame characteristics on live trees to identify the maximum fire behaviour threshold beyond which trees die. We kept the surface fire intensities instead of using the crown fire intensities to complete the nomograms because the Val Paradis fire presented a highly variable fire behaviour with only intermittent crown fire behaviour.

5.5. RESULTS

5.5.1. Stand composition influence

For a given fire severity, stand type and species composition both have always significant effects on the fire intensity (Table 5.3), and there is an interaction between the two factors (stand type and species), only for the extreme fire severity. The Tukey tests show that for a stand mortality level less than 75% (low and moderate fire severities), deciduous plots supported a significantly less intense fire than mixed and coniferous plots, which means that *Populus tremuloides* trees, mainly found in deciduous and mixed plots, burned significantly less intensely than *Pinus baksiana* and *Picea mariana* trees.

5.5.2. Species mortality rates

Comparisons between species mortality rates (Table 5.4) show that species present constant significantly different mortality rates according to fire intensity. In the 0 - 100 kW/m fire intensity class, *Picea mariana* and *Pinus banksiana* have different mortality rate from *Populus tremuloides*. There is a statistical difference for *Picea mariana* because this species is represented by numerous trees (369 trees), but not for *Pinus banksiana* (62 trees). Moreover, the mortality rate difference between *Picea mariana* or *Pinus banksiana* (the same rate) and *Populus tremuloides* is high enough (10%) to be considered as different. *Populus tremuloides* is the least fire-resistant species because its mortality rates are the highest, with all aspen trees dying when conditions reach over 5000 kW/m in fire intensity. Conversely, the two coniferous species are significantly more fire-resistant than *Populus tremuloides*, and even though *Pinus banksiana* seems to be more fire-resistant than *Picea mariana*, a statistical differentiation between conifers is not possible at this level of the analysis.

Table 5.3 Results from the two-way ANOVA done on ranked fire intensities and from the multiple comparison tests on the effects of stand types and species on the fire intensity

Fire severity	Source	ANOVA characteristics			Multiple comparisons among stand types			Multiple comparisons among species		
		n	F	p	Deciduous mean ± SE	Mixed mean ± SE	Coniferous mean ± SE	*Populus tremuloides* mean ± SE	*Pinus banksiana* mean ± SE	*Picea mariana* mean ± SE
Low	Overall model	715	5.34	0.0003	348 ± 132b	911 ± 178a	669 ± 80a	94 ± 121b	972 ± 194a	796 ± 80a
	Stand type		5.22	0.0056						
	Species		6.97	0.0010						
Moderate	Overall model	770	51.96	0.0001	371 ± 286b	2658 ± 225a	2965 ± 149a	253 ± 235b	2744 ± 257a	3217 ± 143a
	Stand type		12.08	0.0001						
	Species		34.14	0.0001						
Extreme	Overall model	478	53.37	0.0001	7807 ± 297ab	7702 ± 158b	8126 ± 123a	6591 ± 238b	8415 ± 185a	8085 ± 112a
	Stand type		124.99	0.0002						
	Species		35.66	0.0003						
	Interaction		11.78	0.0004						

Note: Intensities (mean ± standard error) in a row followed by the same letter are not significantly different at $\alpha = 0.05$ for the Hsu's test.

Table 5.4 Likelihood ratio tests and multiple comparison results for the species sensitivity (susceptibility to be dead three months after the fire has occured)

Fire intensity kW /m	N	N. dead trees	Under Ho: Pmortality	χ^2 calculated	Pmortality and multiple comparisons		
					P. tremuloides	*P. banksiana*	*P. mariana*
0 - 100	674	95	0.141	12.790	0.21 a	0.10 ab	0.10 b
100 - 1000	413	143	0.346	6.380	0.33 ab	0.23 b	0.39 a
1000 -5000	337	194	0.576	58.940	0.86 a	0.28 c	0.51 b
> 5000	634	571	0.901	11.620	1.00 a	0.89 b	0.89 b

Notes: χ^2 (2; 0.05) = 5.99, so Ho is rejected for all tests. Specific mortality probabilities with the same letter for a given fire intensity test are not significantly different at $\alpha = 0.05$.

5.5.3. Morphological characteristics variables

For all species and any given fire intensity class, dead trees, on average, have smaller DBH, higher crown base height or thinner bark, and they are smaller in height than surviving trees (Figures 5.1, 5.2, and 5.3). However, all dendrometric variables do not show significant differences between dead and live trees for all species and within all fire intensity classes. It seems that the higher the fire intensity, the higher the mean value for live trees (DBH for *Pinus banksiana* and *Picea mariana*, TOTH for all three species, and MEBT for *Picea mariana*), except for CBH, which presents an interesting constant mean value with increasing fire intensity. For *Pinus banksiana* (Figure 5.1) and *Picea mariana* (Figure 5.2), all variables are characteristic variables, whereas *Populus tremuloides* records only TOTH as a characteristic variable (Figure 5.3).

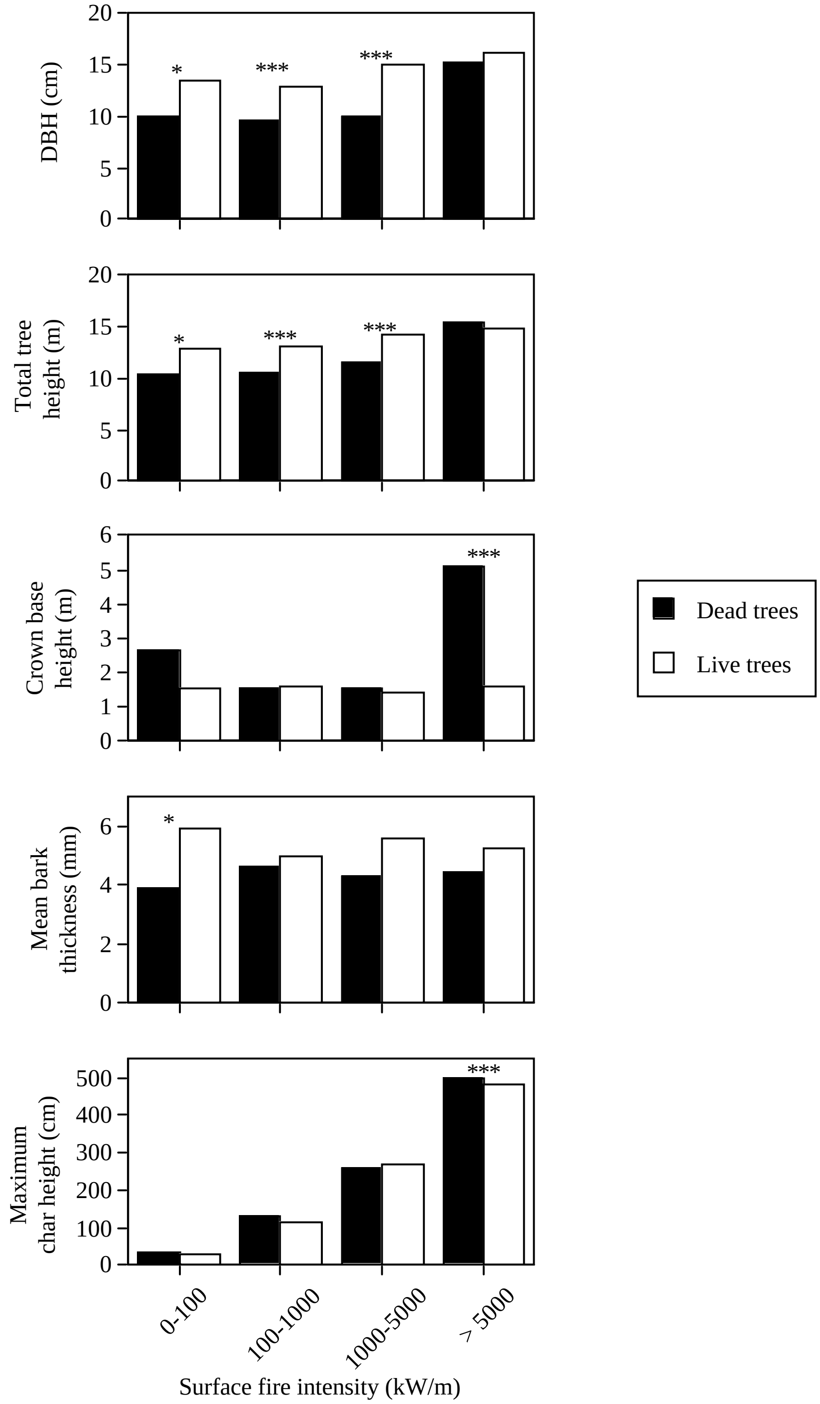

Figure 5.1 Comparisons in morphological characteristics and maximum char height for dead and live *Pinus banksiana* through the fire intensity gradient.
Differences result from the Wilcoxon-Mann-Whitney tests (* for $p < 0.05$; ** for $p < 0.01$; *** for $p < 0.001$).

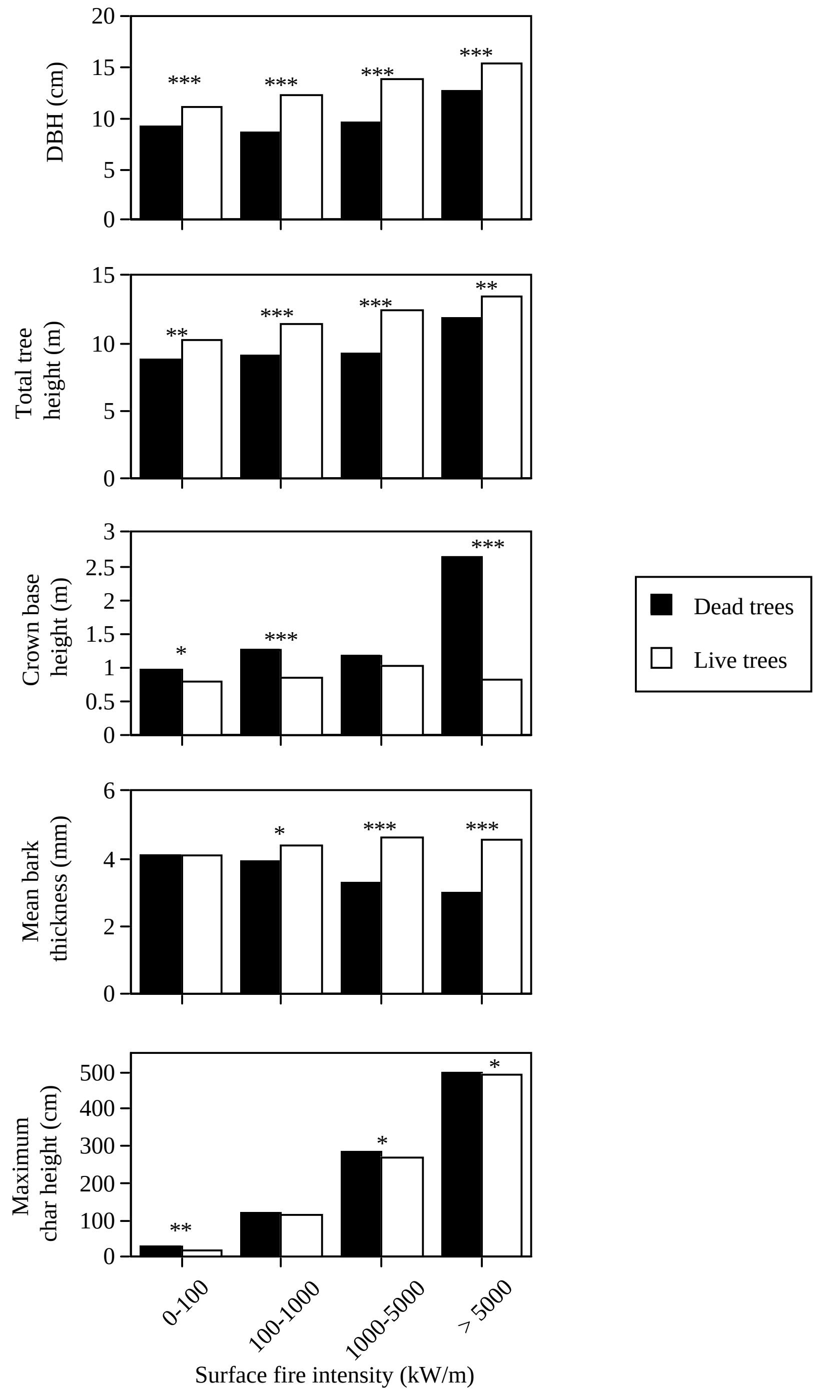

Figure 5.2 Comparisons in morphological characteristics and maximum char height for dead and live *Picea mariana* through the fire intensity gradient.
Differences result from the Wilcoxon-Mann-Whitney tests (* for $p < 0.05$; ** for $p < 0.01$; *** for $p < 0.001$).

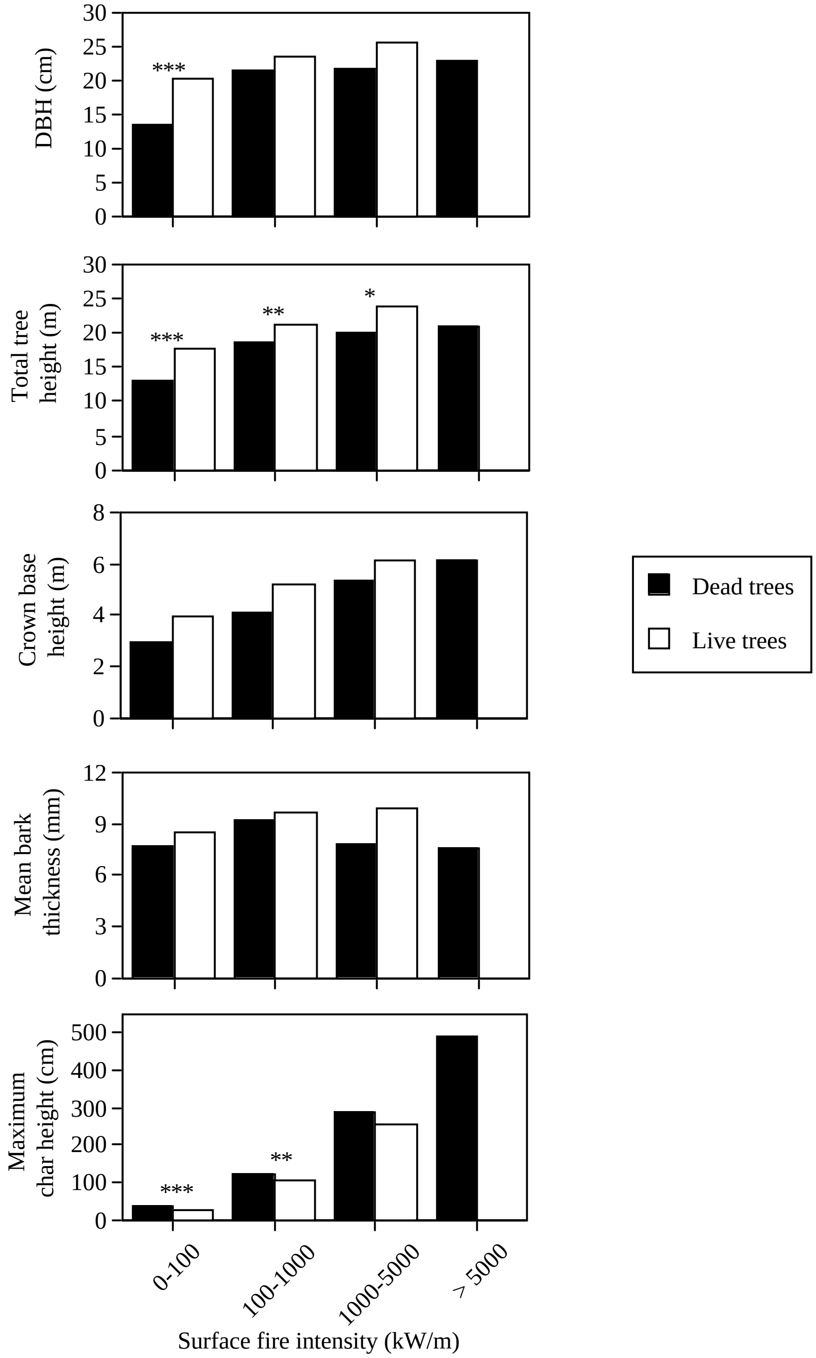

Figure 5.3 Comparisons in morphological characteristics and maximum char height for dead and live *Populus tremuloides* through the fire intensity gradient.
Differences result from the Wilcoxon-Mann-Whitney tests (* for $p < 0.05$; ** for $p < 0.01$; *** for $p < 0.001$).

5.5.4. Tree species mortality modeling

All species logistic regressions performed only on tree characteristics are significant (Table 5.5). Independent selected variables are bark thickness (MABT or MEBT), CBH, TOTH, and the crown base height/total tree height ratio CR. The bark thickness presents the expected negative effect on mortality. The total tree height also increases the fire resistance and is negatively affecting mortality, whereas the crown base height is positively correlated with mortality. Moreover the concordance between the observed outcomes and the predicted probabilities ranges from 60.5% for *Populus* to 87.3% for *Pinus*, with the two conifer species showing a concordance higher than 70%. The Hosmer and Lemeshow goodness-of-fit test presents only a significant difference from a theorically perfect model for *Pinus banksiana*, which means that this model is perhaps not as good as the two other ones. Because Kappa coefficients are quite low (except for *P. mariana*), these significant models exclusively built on morphological variables are not efficient. Indeed the Kappa coefficient shows for *P. banksiana* that the model (Table 5.6, tree line only) increases only by 19% the prediction of tree mortality from the random prediction. Its overall accuracy of 0.65 corresponds with a 65% concordance between the predicted probabilities and the observed outcomes from the validating 25% of the data set. The model's weakness is explained both by omission and commission errors. The 0.69 omission error for dead trees shows that 69% of dead trees have been predicted alive by the model, while the 0.41 commission error for the dead trees shows that 41% of living trees have been predicted dead.

Table 5.5 Logistic regression models for predicting tree mortality with only dendrometric variables

Species	$Pm = 1/(1+e^{-y})$ where:					-2 Log L	Hosmer & Lemeshow goodness (p)	Concordance between predicted probabilities and observed responses (%)	Kappa coefficient
	y = bo + b1*TOTH + b2*CBH								
P. banksiana	Variable	b_i	S.E.	Wald	p				
n = 245	Constant	1.2142	0.9298	1.7055	0.1916	223.62	0.0001	87.3	0.19
$R^2 = 0.3726$	TOTH	-0.2666	0.0746	12.7793	0.0004				
	CBH	1.0287	0.1616	40.5154	0.0001				
	y = bo + b1*CBH + b2*MEBT								
P. mariana	Variable	b_i	S.E.	Wald	p				
n = 914	Constant	0.0011	0.1937	0.8065	0.9955	1037.39	0.1766	76.9	0.49
$R^2 = 0.2162$	CBH	0.8461	0.0903	87.8798	0.0001				
	MEBT	-0.2461	0.0389	40.0068	0.0001				
P. tremuloides	y = bo + b1*CR + b2*MABT								
n = 380	Variable	b_i	S.E.	Wald	p				
$R^2 = 0.0478$	Constant	0.1690	0.3235	0.2729	0.6014	506.41	0.5992	60.5	0.12
	CR	2.3067	0.8079	8.1527	0.0043				
	MABT	-0.0864	0.0263	10.7980	0.0010				

Note: MABT = Maximum bark thickness (in millimeters); MEBT = Mean bark thickness (in millimeters); CBH = Crown base height (in meters); TOTH = Total tree height (in meters); CR = crown base height/total tree height ratio.

Table 5.6 Overall acuracy, omisson and commission errors and Kappa coefficient for each specific logistic regression model

Species	Regression type	Under Ho	Overall accuracy	Omission error		Comission error		Kappa coefficient
				Live trees	Dead trees	Live trees	Dead trees	
P. banksiana	tree	0.39	0.65	0.14	0.69	0.34	0.41	0.19
	tree+fire	0.39	0.79	0.02	0.50	0.25	0.06	0.52
P. mariana	tree	0.55	0.74	0.24	0.27	0.31	0.21	0.49
	tree+fire	0.55	0.79	0.12	0.28	0.29	0.12	0.58
P. tremuloides	tree	0.42	0.57	0.35	0.53	0.37	0.51	0.12
	tree+fire	0.45	0.83	0.07	0.31	0.21	0.10	0.64

However, the addition of fire characteristics such as the char height improves the model efficiency (Table 5.7). Indeed, the improved concordance between the observed outcomes and the predicted probabilities ranges now from 87.2% to 93.5%, the Hosmer and Lemeshow goodness-of-fit tests show that all models fit the data well, and the Kappa coefficients are higher (Table 5.6, tree + fire line). *Populus tremuloides* presents the highest concordance improvement (27.4%) as compared to the exclusively morphological model. *Pinus banksiana* still presents the highest concordance. The best performance is attributed to the *Picea* model, where only 12% of dead trees are predicted alive while 12% of live trees are predicted dead (Table 5.6).

Char height (MICH, MACH, or MECH) is the only fire variable included, and it is always entered at the first step of the stepwise procedure (Table 5.7). *Pinus banksiana* and *Picea mariana* present the same behaviour in reaction to the fire, as their models include DBH and char height (MECH or MICH), whereas *Populus* mortality depends on MACH and TOTH. Because the intensity was calculated from the char height, the fire intensity was never the best significant variable. The hypothesis about relationships between fire-caused mortality and fire characteristics is confirmed.

Table 5.7 Logistic regression models for predicting tree mortality using dendrometric and fire variables

Species	Pm = $1/(1+e^{-y})$ where:	Variable	b_i	S.E.	Wald	p	-2 Log L	Hosmer & Lemeshow goodness (p)	Concordance between predicted probabilities and observed responses (%)	Kappa coefficient
P. banksiana	y = bo + b1*MECH + b2*DBH									
n = 246		Constant	3.2362	0.8593	14.1821	0.0002	158.03	0.6777	93.5	0.52
$R^2 = 0.5209$		MECH	0.0141	0.0017	65.5284	0.0001				
		DBH	-0.4821	0.0835	33.3478	0.0001				
P. mariana	y = bo + b1*MICH + b2*CBH									
n = 914		Constant	-0.1715	0.2378	0.5201	0.4708	783.68	0.0576	87.2	0.58
$R^2 = 0.4062$		MICH	0.0133	0.0012	128.1090	0.0001				
		DBH	-0.0859	0.0203	17.9330	0.0001				
P. tremuloides	y = bo + b1*MACH + b2*TOTH									
n = 389		Constant	0.4501	0.4415	1.0393	0.3080	320.63	0.6585	87.9	0.64
$R^2 = 0.4258$		MACH	0.0177	0.0020	81.6326	0.0001				
		TOTH	-0.1511	0.0278	29.4847	0.0001				

Note: MACH = Maximum char height (in centimeters); MECH = Mean char height (in centimeters); MICH = Minimum char height (in centimeters); DBH = Diameter at breast height (in centimeters); TOTH = Total tree height (in meters).

5.5.5. Species nomograms

The three specific nomograms (Figures 5.4, 5.5, and 5.6) were constructed from the logistic regression models using the dendrometric and fire variables. The first graph on the upper left corresponds, in Figure 5.4 for example, to the probability of fire-caused mortality for *Pinus banksiana*. Because the logistic regression involves the mean char height, this variable has then been related to the maximum char height, which was finally related to the frontal fire intensity and to the flame length. To use the mortality nomograms, choose an acceptable level of mortality (e.g., 20% or 0.2 probability of mortality). Acceptable mortality will depend on the size of trees and on the objectives of the managers. Successful underburning, for example, involves choosing and staying within a reasonable level of mortality. Consider, for example, a homogeneous *Pinus banksiana* stand averaging 20 cm in diameter. Entering the nomogram at the upper left corner with the observed diameter, draw a horizontal line until you intersect the acceptable probability of mortality (0.2 for example). Then turn a right angle and draw a line straight down. Where the line crosses the lower edge of the box, mean char height can be read, if desired. Continue the line straight down until it intersects the relationship line between mean and maximum char heights, and then to the right, to find the acceptable frontal fire intensity (depending on the wind speed through the flame angle) to reach a maximum mortality rate of 20%. The second option is to continue the line straight down from the middle left graph to reach the lower left graph. When turning a right angle to the right the reader will find out the acceptable equivalent flame length range. In the example with an average of 20 cm DBH, 0.2 probability of mortality, and a flame angle of 80°, which corresponds to the light wind speed within stands that burned the Val Paradis area, the frontal fire intensity will be acceptable up to 5400 kW/m and the corresponding flame length up to 4 m. If the *Pinus banksiana* stand would average 15 cm in diameter, the corresponding maximum frontal fire intensity acceptable would be 1600 kW/m and the flame length 2.4 m. For *Picea mariana*, the same choice (20 cm DBH and 0.2 probability of mortality) would suggest managers to use a low intensity surface fire with flame 1.2 m long maximum and frontal fire intensity 400 kW/m (Figure 5.5). The *Populus tremuloides* nomogram starts with a quadratic polynomial relationship between DBH and TOTH (Figure 5.6) for an easier use of the nomogram when working with the three species. Indeed, it allows the comparisons among species by starting with the same DBH variable. Moreover, because the *Populus tremuloides* mortality depends on total tree height and directly on maximum scorch height, the middle and lower left graphs represent the first diagonal. For a 0.2 probability of mortality, a *P. tremuloides* stand averaging 20 cm DBH

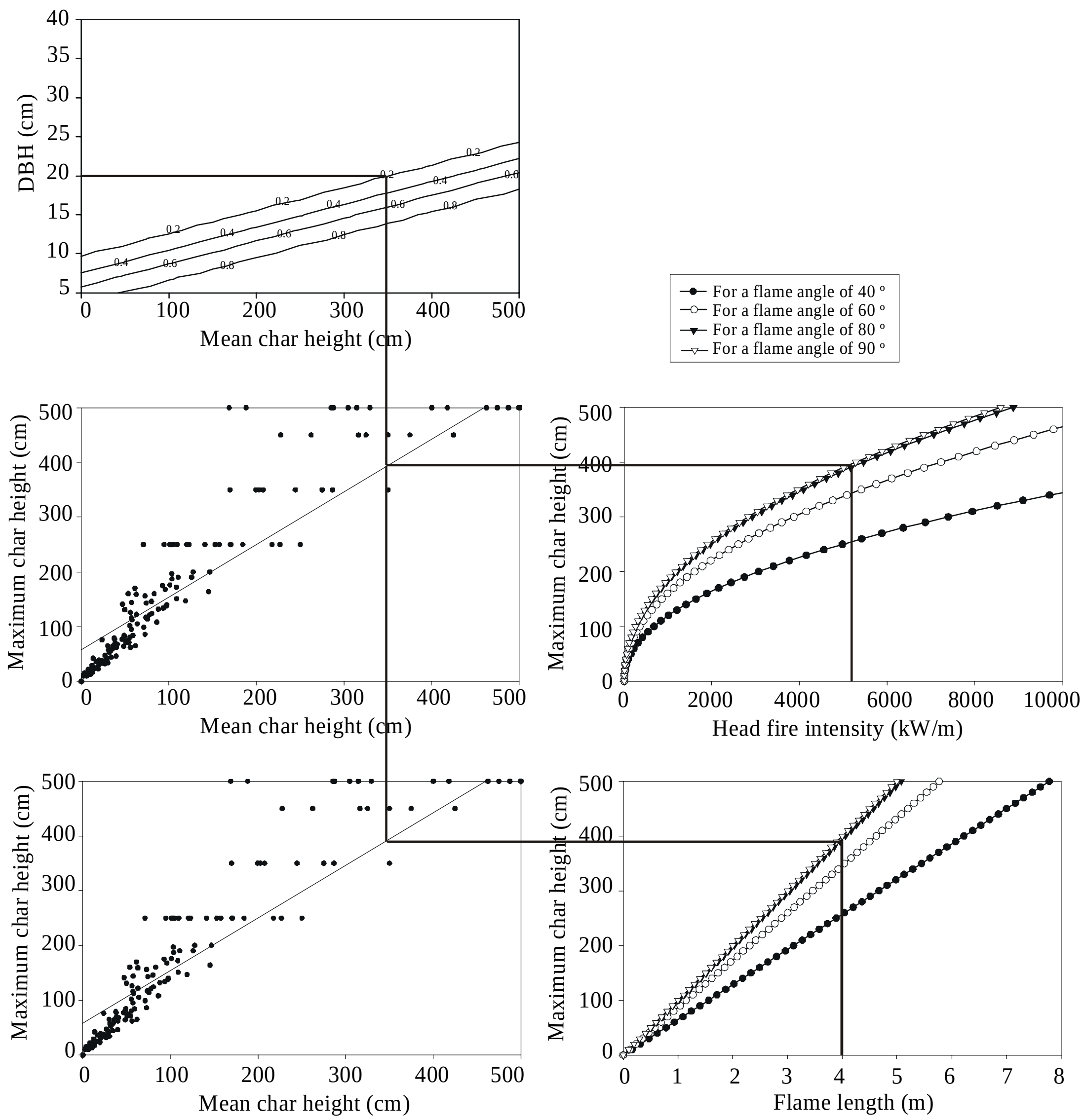

Figure 5.4 *Pinus banksiana* nomogram.

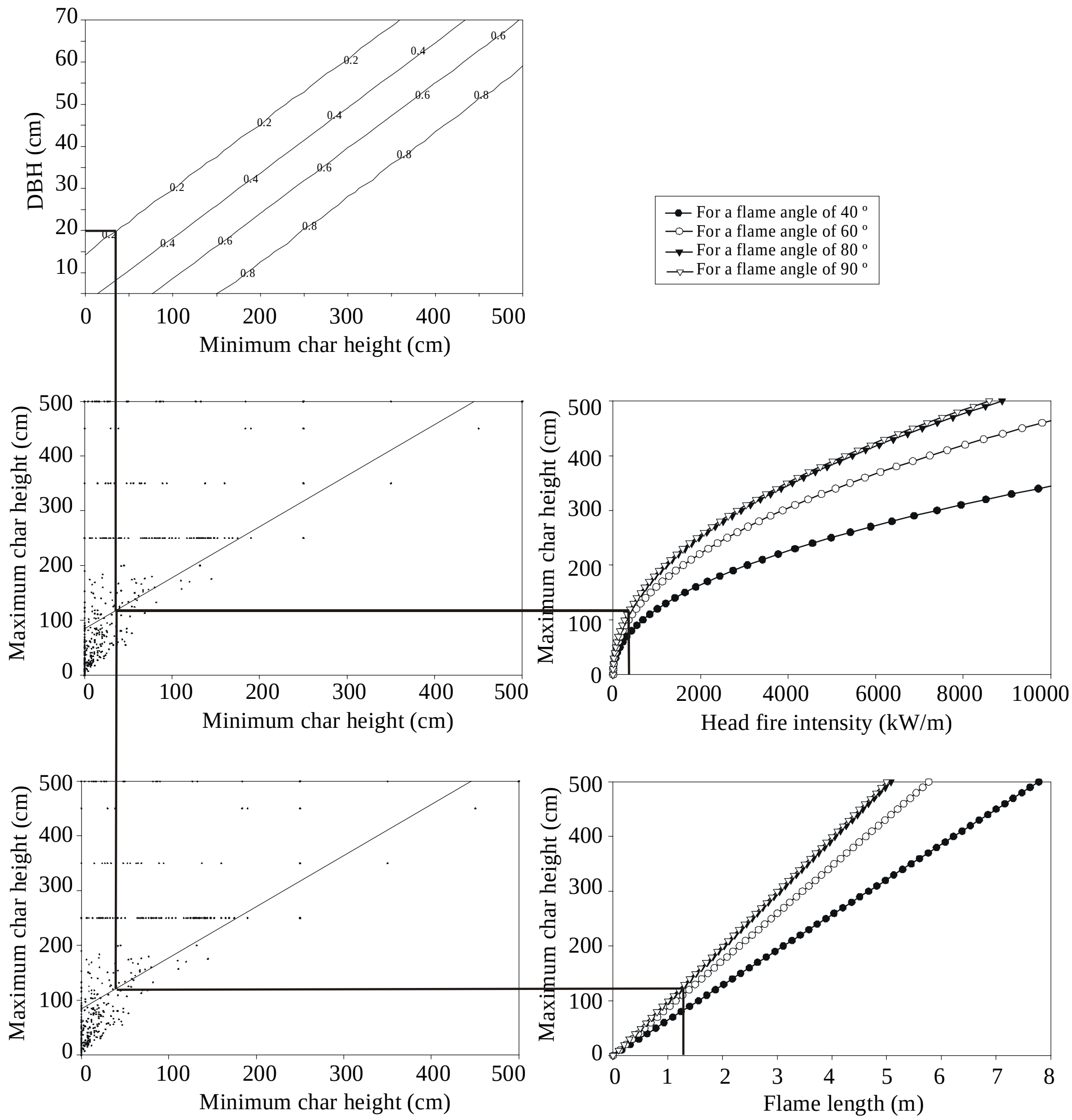

Figure 5.5 *Picea mariana* nomogram.

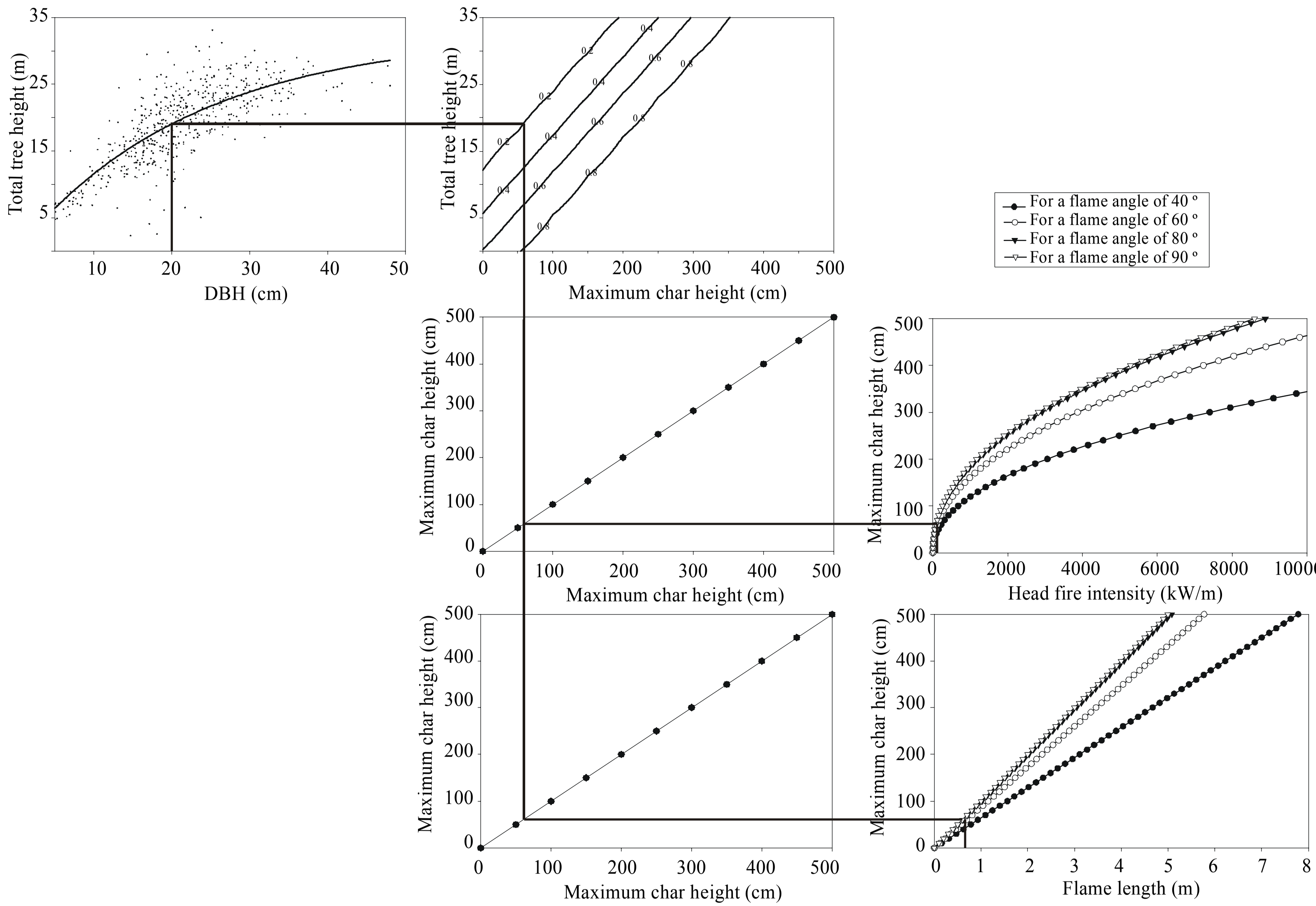

Figure 5.6 *Populus tremuloides* nomogram.

(corresponding roughly to 20 m tall), would require a less intense fire behaviour (less than 150 kW/m and 0.5 m flames long) than the *Picea mariana* stand.

5.6. DISCUSSION

This study, based on a wildfire, has confirmed that stand composition influenced fire behaviour, which in turn influenced tree species mortality. Given the same level of fire severity (based on the same percentage of stand mortality), deciduous stands supported less intense fire than mixed and coniferous stands. This implies that deciduous species are less fire-resistant than conifer species. Fire behaviour prediction systems such as the FBP System (Forestry Canada Fire Danger Group 1992) simulate significant fire behaviour for distinct stand types, but they were not created to look at the species reaction to fire in terms of mortality. The BEHAVE System (Burgan and Rothermel 1984, Andrews 1986, Andrews and Chase 1989) can predict tree mortality, but it takes into account only a few species, mainly from western USA, which cannot be related to boreal species. Moreover, Hély et al. (Submitted) have shown that the BEHAVE System was not well adapted to simulate fire behaviour in the mixedwood boreal forest, even using the new fuel model. The following discussion will develop the mortality causes and the differences found in fire-resistance among boreal species.

5.6.1. Mortality causes

This study focused on immediate stem and crown injuries, but several fire-resistance mechanisms confirmed in previous studies (Ryan et al. 1988, Harrington 1993, Regelbrugge and Conard 1993, Mutch and Parsons 1998) have been highlighted for the boreal forest species such as DBH, bark thickness, and total tree height that are negatively related to post-fire tree mortality. Moreover, in trees that survived fire the values of DBH and bark thickness increased with a rise in fire intensity. The greater fire resistance of trees with larger diameter is generally attributed to increased insulation of the cambium by the thicker bark of larger trees (Hare 1965, Ryan and Reinhardt 1988, Peterson and Arbaugh 1989, Mutch and Parsons 1998). Several authors (Harmon 1984, Ryan and Reinhardt 1988) have been criticized by Regelbrugge and Conard (1993) for their use of relationships between DBH and bark thickness, when the latter variable was not measured during the field sampling. This study directly measured bark thickness to take into account its natural variability due to site differences, random variability, and age (Regelbrugge and Conard 1993). Gutsell and Johnson (1996) have shown than small-diameter trees will always die because they won't be able to create fire scars as a resistance mechanism. This is because the leeward and windward flames linger for the same amount of time and thus the critical temperature to affect cambial kill is present on all sides of the tree. Nevertheless, Ryan and Frandsen (1991) have found that some larger trees could die easily when these trees were highly fissured because resistance to fire injury varies somewhat between plates and fissures. Mortality due to crown scorching has often been related to the total tree height, especially in studies on conifers, with mortality increasing with decreasing tree size (Ryan and Reinhardt 1988). This happens because crowns of smaller trees are closer to the fire and sustain greater injury through crown scorching (Harmon 1984) and also because these small trees are frequently in a low vigor, suppressed state with little tolerance for damage (Harrington 1993). The crown base height for the boreal conifer species is a variable that reacts differently than for the western conifer species. In boreal species, it seems to reinforce the resistance from other variables, particularly the cambium resistance (DBH and MEBT), as we have seen that survivor trees have lower CBH than dead trees. The crown base seems to act as a muff and protects the stem from direct flame contact. This kind of protection and resistance has never been found before, and even this characteristic of low CBH has been associated with a less efficient resistance to fire mortality in previous studies (Harmon 1984, Ryan and Reinhardt 1988). However, if we add to this protection the fact that the photosynthetic power is lower in the lower one-third of the crown than in the middle and upper thirds (Brown and Davis 1973, Ryan 1998), trees that can protect their cambium through the presence of low branches near the ground, if not completely scorched, will be able to recover from this injury by active and efficient photosynthesis on the upper crown part. The interactions between the different resistances to fire injury (cambium and crown) can be found in the relationships that links total tree height and DBH for many species (Regelbrugge and Conard 1993). In this study, the two conifer species are good examples of species presenting interactions between cambium resistance, crown resistance, and fire behaviour (MACH is a significant variable), and these simultaneous relationships exist throughout the fire intensity range up to the threshold beyond which all trees die. Conversely, *Populus tremuloides* does not show an interaction between its low fire resistance and the fire behaviour (MACH not involved), and it reaches the threshold as low as 1000 kW/m. Few studies have mentioned this kind of threshold.

As we have seen, mortality can result from one or several types of injuries simultaneously, as mentioned by different authors (Swezy and Agee 1991, Ryan 1998). The mechanisms coupling fire behaviour and tissue injury are complex and still only approximately understood. As fire behaviour changes, so does the magnitude and relative importance of radiation, convection, and conduction. The importance of the heat transfer mechanisms to tree injury varies with tree morphology (Ryan 1998). Variations in local heating conditions lead to variations in stem, crown, and root injuries. Resistance of an individual tree to fire-induced mortality depends on several aspects such as morphological characteristics that protect vital tissues from injury, the ability to recover from some degree of injury, the season when the fire occurs with dormant season creating less damage on buds and better recovery (Peterson and Arbaugh 1986, Swezy and Agee 1991, Harrington 1993, Regelbrugge and Conard 1993), and the possible effects induced by fire injury such as damage by disease and insects (Thomas and Agee 1986, Regelbrugge and Conard 1993, Mutch and Parsons 1998).

5.6.2. Species comparisons

Eastern boreal species are for the fist time analyzed for post-fire mortality, and they differ considerably in shape and morphology from most of the studied species, mainly from the western part of North America. Indeed, *Pinus banksiana* and *Picea mariana* are thin bark conifer species with conical crown shapes and bases near the ground, and *Populus tremuloides* presents fissures (Brown and Debyle 1987) where injury is higher (Ryan 1998), even though it is a tall spherical crown shape species with a relatively thick bark. We think that all of the above effects of morphology and injury variables on mortality depend on the geometrical shape of the tree and the fuel flammability of its foliage. Indeed, among reviewed species, pine species dominate with a paraboloid or ellipsoid crown shape at the top of the trunk (Peterson 1985, Peterson and Ryan 1986), which allow trees to escape scorching early during the regeneration phase when they are taller than a minimum height. These species with tall crowns are often characterized as thick bark species (Peterson and

Ryan 1986, Mutch and Parsons 1998) with efficiently insulated cambium. Moreover, Brown and Davis (1973) explained that conifer species are more susceptible to crown scorch than deciduous species because their foliage is more flammable, and if scorched, these conifer trees would die more easily. However, even though boreal species seem to be very susceptible to fire-induced mortality by their morphology, and they have been classified as moderately resistant species (*Pinus banksiana*) or the least fire-resistant species (*Picea mariana* and *Populus tremuloides*) by Brown and Davis (1973), this study has shown that they may resist, to a certain extent, post-fire mortality, up to the fire intensity threshold beyond which they die. We confirmed the Brown and Davis (1973) classification by the mortality rate analysis that showed *Populus tremuloides* as significantly the less resistant species as compared to the two conifer species, and discrimination between *Pinus banksiana* and *Picea mariana* has been showed by the nomograms. Low resistance of *Populus tremuloides* may be explained first by the non-interaction between its crown resistance (almost only effective through the TOTH variable) and fire behaviour, and secondly by fissures in its bark. Indeed, even at low or moderate fire intensity, flames can reach a certain height on the trunk where the fissures are more common than near the base where the bark is also thicker. Ryan and Frandsen (1991) have found the same pattern with large *Pinus ponderosa* trees. This could explain why the threshold is reached at the 1000 kW/m level. The two coniferous species have comparable survival behaviour against fire, with efficient resistance from cambium and crown through DBH, MEBT, CBH, and TOTH, respectively. Moreover, for *Pinus banksiana* the threshold is reached at around 57760 kW/m, beyond which fire intensity killed all pines (54 trees). The threshold is also reached for *Picea mariana*, but the mortality delay was greater. Indeed, even though two trees seemed to be alive immediately after the fire with an intensity of approximately 93820 kW/m, they were dead the following year (c. Hély, personal observation).

5.6.3. Model evaluation and uses

When stands that contain trees with a wide range of fire resistance burn in a variable fire, there is a high probability that some non fire-resistant trees will escape serious injury due to localized areas of low fire severity. Likewise, some fire-resistant trees will receive serious injury due to localized areas of high fire severity. In such cases both morphology and fire behaviour variables are needed to adequately predict tree response (Ryan 1998). This illustrates why we obtained a better performance for logistic regressions using both tree morphology and fire behaviour variables than using only tree morphology variables. Moreover, our logistic regression models have been validated from independent data sub-sets as only a few studies on tree mortality have done before (Regelbrugge and Conard 1993). In general, the sole model performance appreciation came from the same data and directly from the model. The use of the Kappa coefficient analysis, commonly used in different sciences such as geography (Jensen 1996) is perfectly adapted to validate these models with binary dependent variables, and it should be used in future studies. The char height was the only fire variable included in the stepwise procedure, and it has been previously found as a significant estimator for the post-fire mortality (Wyant et al. 1986, Regelbrugge and Conard 1993). Moreover, the models also revealed that DBH was only a significant predictor of postfire mortality when fire characteristics were added. We tried to use the square diameter as recommended by Ryan (1998), but it was not retained as the best significant descriptor in this study. Regelbrugge and Conard (1993), working on *Pinus ponderosa*, have found that the best prediction model for post-fire mortality was based on char height and DBH. These results are similar to those from our study for *Pinus banksiana*. Char height, found in previous studies to be a poor estimator of absolute flame length or fire intensity was, however, reported as a useful estimator for relative differences in flame length and fire intensity (Cain 1984).

Even though nomograms presented here are based only on one wildfire, they take into account the high natural variability that exists within the fire (wide fire intensity and flame length ranges) and within the morphological characteristics of each species. That is why these nomograms are thought to be close to the final product that could be created with the addition of several fires, and improvement of flame height - flame length relationships. Brown and Debyle (1987) have indeed found that char height represented half flame height in aspen stands with sustained fire. Because we used a unit ratio between char height and flame height, our nomograms should be considered as conservative tools. Because the studied species have never been analyzed before, these nomograms could improve to a certain extent some existing mortality prediction systems (Andrews and Chase 1989) by addition of new species. Contrary to the nomograms based on experimental burns for prescribed burns objectives only, the nomograms reported here are based on a spring wildfire intense enough to be considered as an intermittent or passive crown fire. The use of these nomograms has to be done in conjunction with fuel models and fire behaviour predictions to develop realistic fire conditions (Rothermel 1983, Andrews and Chase 1989, Hirsch 1996). The nomograms could be useful in evaluating the natural post-fire tree mortality, by measuring the mean stand DBH or tree height and maximum char height on stems, in the perspective of wood salvage guidelines. These nomograms could be used with prescribed burns for a regeneration perspective, such as the ones Brown and Debyle (1987) suggested for *Populus tremuloides* stands through suckering stimulation, or to achieve the desired number and species proportions after fire treatment (Reinhardt and Ryan 1988). In the sustainable management of the mixedwood boreal forests (Bergeron and Harvey 1997), managers could use these nomograms to rejuvenate some stands (e.g., a mixed-conifer stand toward a mixed-deciduous stand). In all prescribed burns, however, the desired ecological effects of fire have to be defined first to select the appropriate fire behaviour and subsequent fire intensity and flame length (Johnson and Miyanishi 1995). Indeed, if the management objective was to consume large amounts of fuel and to expose mineral soil, a fire with a low rate of spread and high fuel consumption would be used. On the other hand, if the objective was to kill woody stems but not to expose large amounts of mineral soil, then a fire with a high rate of spread, high flame height, and low fuel consumption would be used. Field crews can attempt to limit rate of spread and flame length to the selected levels (from fire behaviour prediction system and nomogram) through different ignition and propagation patterns (Reinhardt and Ryan 1988) such as strip, back, or flanks spreading.

5.7. CONCLUSION

The post-fire tree mortality study in the eastern Canadian mixedwood boreal forest has confirmed that local variability within stand influences the fire behavior at the stand level and also at the landscape mosaic level. Indeed, char height on the trunc, related to fire intensity, was lower in deciduous stands than in mixed or coniferous stands. This study has also shown that the eastern boreal species react differently than those in the western part of North America. The low crown base height for *Pinus banksiana* and *Picea mariana* seems to reinforce the resistance from other variables, particularly the cambium resistance (thin bark and small DBH) by acting as a muff and protecting the stem from direct flame contact. This kind of protection and resistance has never been found before. Moreover, the two conifer species are good examples of species presenting interactions between cambium resistance, crown resistance, and fire behaviour, and these simultaneous relationships exist throughout the fire intensity range up to the threshold beyond which all trees die. Conversely, *Populus tremuloides*, with only the total tree height as an active variable in fire-resistance, presents the highest mortality rates, and it reaches the threshold of 100% dead trees as low as 1000 kW/m. The addition of fire behavior component such as char height to dendrometric variables in the species logistic regressions has greatly improved the post-fire mortality models efficiency. Moreover these logistic regression models seem to be potentially efficient to create forest management tools such as nomograms.

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5.9. REFERENCES

Alexander, M.E. 1982. Calculating and interpreting forest fire intensities. Can. J. Bot. 60:349-357.

Alexander, M.E. 1998. Crown fire thresholds in exotic pine plantations of Australasia. Ph.D. Thesis, Australian National Univ., Canberra, ACT, Australia. 228p.

Amiro B.D. 1990. Comparison of turbulence statistics within three boreal forest canopies. Boundary-Layer Meteorol. 51:99-121.

Amiro, B.D., and P.A. Davis. 1988. Statistics of atmospheric turbulence within a natural black spruce forest canopy. Boundary-Layer Meteorol. 44:267-283.

Andrews, P.L. 1986. BEHAVE:fire behavior prediction and fuel modeling system--BURN subsystem, PART 1. USDA, For. Serv., Ogden, UT. Interm. For. Range Exp. Stn., Gen. Tech. Rep. INT-194.

Andrews, P.L., and C.H. Chase. 1989. BEHAVE: fire behavior prediction and fuel modeling system--BURN subsystem, PART 2. USDA, For. Serv., Ogden, UT. Interm. For. Range Exp. Stn., Gen. Tech. Rep. INT-260.

Bergeron, Y., and B. Harvey. 1997. Basing silviculture on natural ecosystem dynamics: an approach applied to the southern boreal mixedwood forest of Quebec. For. Ecol. Manage. 92:235-242.

Bergeron, Y., S. Gauthier, C. Carcaillet, M.D. Flannigan, Y. Prairie and P.J.H. Richard. 1999. Variability in fire frequency and forest composition in Canada's southeastern boreal forest: a challenge for sustainable forest management. P. 74-80 *in* Proc. of conference on Science and Practice: Sustaining the boreal forest, T.S.Veeman, et al. (eds.) Edmonton, Alberta, Canada.

Brown, A.A., and K.P. Davis. 1973. Forest fire: control and use. McGraw-Hill, New York. 686p.

Brown, J.K., and N.V. Debyle. 1987. Fire damage, mortality, and suckering in aspen. Can. J. For. Res. 17:1100-1109.

Burgan, R.E., and R.C. Rothermel. 1984. BEHAVE: fire behavior prediction and fuel modeling system--FUEL subsystem. USDA, For. Serv., Ogden, UT. Interm. For. Range Exp. Stn., Gen. Tech. Rep. INT-167.

Cain, M.D. 1984. Height of stem-bark char underestimates flame length in prescribed burns. Fire Manage. Notes 45:17-21.

Cionco, R.M. 1978. Analysis of canopy index values for various canopy densities. Boundary-Layer Meteorol. 15:81-93.

Conover, W.J., and R.L. Iman. 1981. Rank transformations as a bridge between parametric and nonparametric statistics. Am. Statistician 35:124-129.

Daniels, R.F., W.A. Leuschner, S.J. Zarnoch, H.E. Burkhart and R.R. Hicks. 1979. A method for estimating the probability of southern pine beetle outbreaks. For. Sci. 25:265-269.

Finney, M.A., and R.E. Martin. 1993. Modeling effects of prescribed fire on young-growth coast redwood trees. Can. J. For. Res. 23:1125-1135.

Flannigan, M.D. 1993. Fire regime and the abundance of red pine. Int. J. Wild. Fire 3:241-247.

Forestry Canada Fire Danger Group. 1992. Development and structure of the Canadian Forest Fire Behavior Prediction System. For. Can., Ottawa, Ontario, Inf. Rep. ST-X-3:63p.

Gutsell, S.L., and E.A. Johnson. 1996. How fire scars are formed: coupling a disturbance process to its ecological effect. Can. J. For. Res. 26:166-174.

Hare, R.C. 1965. Contribution of bark to fire resistance of southern trees. J. For. 4:248-251.

Harmon, M.E. 1984. Survival of trees after low-intensity surface fires in Great Smoky Mountains national park. Ecology 65:796-802.

Harrington, M.G. 1993. Predicting Pinus ponderosa mortality from dormant season and growing season fire injury. Int. J. Wild. Fire 3:65-72.

Hély, C., M.D. Flannigan, Y. Bergeron, and D.J. Mcrae. 2000. Role of vegetation and weather on fire behavior in the Canadian mixedwood boreal forest using two fire behavior prediction systems. Can. J. For. Res. submitted.

Hirsch, K.G. 1996. Canadian Forest Fire Behavior Prediction (FBP) System: user's guide. Nat. Resour. Can., Can. For. Serv., Northwest Reg., North. For. Cent., Edmonton, Alberta. Spec. Rep. 7: 121p.

Huddle, J.A., and S.G. Pallardy. 1999. Effect of fire on survival and growth of Acer rubrum and *Quercus* seedlings. For. Ecol. Manage. 118:49-56.

Jensen, J.R. 1996. Introductory digital image processing, a remote sensing perspective. Second edition. K.C. Clarke ed., Prentice Hall, New Jersey. 318p.

Johnson, E.A. 1992. Fire vegetation dynamics: Studies from the North American boreal forest. R.S.K. Barnes, et al. (eds.) Cambridge studies in ecology, Cambridge Univ. Press, Cambridge. 129p.

Johnson, E.A., and K. Miyanishi. 1995. The need of consideration of fire behavior and effects in prescribed burning. Restoration Ecology 3:271-278.

JUMP IN 1989. Statistical visualization for the Macintosh. Version 2.0.4. SAS Institute, Inc., Cary, NC.

Keane, R.E., S.F. Arno, and J.K. Brown. 1990. Simulating cumulative fire effects in Ponderosa pine / Douglas-fir forests. Ecology 71:189-203.

Mutch, L.S., and D.J. Parsons. 1998. Mixed conifer forest mortality and establishment before and after prescribed fire in Sequoia National Park, California. For. Sci. 44:341-355.

Peterson, D.L. 1985. Crown scorch and scorch height: estimates of postfire tree condition. Can. J. For. Res. 15:596-598.

Peterson, D.L., and M.J. Arbaugh. 1986. Postfire survival in Douglas-fir and lodgepole pine: comparing the effects of crown and bole damage. Can. J. For. Res. 16:1175-1179.

Peterson, D.L., and M.J. Arbaugh. 1989. Estimating postfire survival of Douglas-fir in the Cascade range. Can. J. For. Res. 19:530-535.

Peterson, D.L., and K.C. Ryan. 1986. Modeling postfire conifer mortality for long-range planning. Environ. Manage. 10:797-808.

Press, S.J., and S. Wilson. 1978. Choosing between logistic regression and discriminant analysis. J. Am. Statisticial Association. 73:699-705.

Regelbrugge, J.C., and S.G. Conard. 1993. Modeling tree mortality following wildfire in *Pinus pomderosa* forests in the central Sierra Nevada of California. Int. J. Wild. Fire 3:139-148.

Reinhardt, E.D., and K.C. Ryan. 1988. How to estimate tree mortality resulting from underburning. Fire Manage. Notes 49:30-36.

Rothermel, R.C. 1983. How to predict the spread and intensity of forest and range fires. USDA, For. Serv., Ogden, UT. Interm. For. Range Exp. Stn., Gen. Tech. Rep. INT-143. 161p.

Rowe, J.S. 1972. Forest regions of Canada. Environ. Can., Ottawa, Ontario, Publ. 1300.

Ryan, K.C. 1998. Analysis of the relative value of morphological variables in predicting fire-caused tree mortality. Pages 1511-1526 *in* Proc. of the III International Conference on forest fire research and the 14th Conference on fire and forest meteorology, Luso, Portugal.

Ryan, K.C., and W.H. Frandsen. 1991. Basal injury from smoldering fires in mature *Pinus ponderosa* Laws. Int. J. Wild. Fire 1:107-118.

Ryan, K.C., D.L. Peterson, and E.D. Reinhardt. 1988. Modeling long-term fire-caused mortality of Douglas-fir. For. Sci. 34:190-199.

Ryan, K.C., and E.D. Reinhardt. 1988. Predicting postfire mortality of seven western conifers. Can. J. For. Res. 18:1291-1297.

SAS Institute Inc. 1985. SAS user's guide: statistics, version 6 ed. SAS Institute Inc., Cary, N.C.

Scherrer, B. 1984. Biostatistique. G. Morin. ed., Quebec, Canada. 849p.

Swezy, D.M., and J.K. Agee. 1991. Prescribed-fire effects on fine-root and tree mortality in old-growth ponderosa pine. Can. J. For. Res. 21:626-634.

Thomas, T.L., and J.K. Agee. 1986. Prescribed fire effects on mixed conifer forest structure at Crater Lake, Oregon. Can. J. For. Res. 16:1082-1087.

Van Wagner, C.E. 1973. Height of crown scorch in forest fires. Can. J. For. Res. 3:373-378.

Van Wagner, C.E. 1987. Development and structure of the Canadian Forest Fire Weather Index System. Can. For. Serv., Ottawa, Ontario, Tech. Rep. 35. 36p.

Wyant, J.G., P.N. Omi, and R.D. Laven. 1986. Fire induced tree mortality in a Colorado ponderosa pine/Douglas-fir stand. For. Sci. 32:49-59.

CHAPITRE VI

CONCLUSION GÉNÉRALE

Par une approche originale qui analyse les conditions prévalant avant, pendant et immédiatement après l'incendie, cette thèse offre une meilleure compréhension de la dynamique de la sapinière à bouleau blanc et de son fonctionnement, et elle améliore notre connaissance des incendies en forêt boréale mixte. En effet, les incendies étaient caractérisés jusqu'à présent comme des événements ponctuels passés, selon leur année d'occurrence et la saison, leur localisation, et la superficie brûlée. De plus, dans la plupart des cas l'incendie était supposé avoir été suffisamment intense pour être létal afin de pouvoir expliquer la régénération de la zone brûlée selon les patrons successionnels. Cette thèse améliore donc l'étude du régime du feu en se concentrant plus particulièrement sur le comportement du feu. Elle démontre l'importance de l'influence locale de la composition végétale sur le comportement du feu au sein des peuplements, ainsi que les répercussions sur la mosaïque paysagère à travers la mortalité post-incendie.

La première contribution importante de cette thèse concerne l'étude des débris ligneux et leur évolution temporelle au cours de la succession. En effet, ce type d'analyse est le premier du genre en forêt boréale mixte au Canada et les résultats qui en découlent vont à l'encontre des principaux modèles d'accumulation de matière morte des milieux forestiers nord Américains. En effet, si dans la sapinière à bouleau blanc les patrons temporels d'accumulation des débris ligneux au sol suivent une courbe sigmoïde et les chicots une courbe de croissance progressive, à l'opposé la plupart des autres écosystèmes forestiers présentent des accumulations selon une courbe en "U". Cette différence majeure en forêt boréale mixte s'explique notamment par le remplacement des espèces dominantes au sein de la canopée, des productivités spécifiques différentielles, des taux de décomposition élevés, et l'occurrence épisodique de certaines perturbations naturelles telles que les épidémies de la tordeuse des bourgeons de l'épinette (TBE) qui représentent des pulses d'apport en débris ligneux.

Le second point important de cette thèse sur la sapinière à bouleau blanc est l'apport de connaissance au sujet des changements temporels de susceptibilité au feu dans les peuplements. En effet, cette étude montre que la susceptibilité à brûler ("fire hazard"), tant au niveau du départ potentiel du feu (agissant sur la fréquence des feux) qu'au niveau de l'intensité pendant la propagation de la flamme (se traduisant après l'incendie par des sévérités différentes), est influencée par la composition végétale et la structure du peuplement. Cette susceptibilité au feu augmente au cours de la succession avec la proportion croissante des conifères. En effet, les peuplements pionniers sur argiles mésiques, dominés généralement par les espèces feuillues héliophiles, se caractérisent par des combustibles de surface de faibles quantités et de médiocre qualité pour le brûlage: les débris ligneux feuillus sont majoritaires et leurs diamètres sont assez grands, il y a peu de arbustes, et la litière est principalement feuillue et peu épaisse. Leur susceptibilité au départ du feu est donc faible, et si toutefois ces peuplements brûlent, l'intensité dégagée sera faible et la sévérité post-incendie atteinte légère. Au cours de la succession, la proportion de conifères augmente au niveau de la canopée, se traduisant au sol par la présence simultanée de charges de combustibles plus importantes et de meilleure qualité pour allumer un feu. Il faut toutefois noter que l'augmentation de charge des combustibles de surface est le résultat net de la disparition rapide des débris feuillus largement compensée par un apport important de débris conifères, ainsi que de la succession des espèces de sous-bois favorisant les espèces qui brûlent plus facilement. La présence supplémentaire de branches basses, typiques chez les conifères et considérées comme combustible étagé, ainsi que des chicots, de plus en plus nombreux dans les vieux peuplements conifères, favorisent la propagation de la flamme du sol vers la canopée. Ainsi, la susceptibilité au feu est le résultat non pas de la simple présence de conifères dans le peuplement ni d'une accumulation de combustible à long terme, mais plutôt de la présence simultanée de différents types de combustibles susceptibles au feu.

La troisième contribution de cette thèse met l'emphase sur la compréhension du comportement du feu en forêt boréale mixte. Dans l'environnement du feu en forêt boréale mixte, une fois la topographie fixée constante, le climat et la végétation sont deux facteurs qui ont un rôle significatif dans le comportement du feu. Ce résultat va à l'encontre de ceux obtenus en forêt boréale de l'Ouest Canadien (Bessie et Johnson 1995) pour lesquels le climat est le seul facteur significatif dans la détermination du comportement du feu. En forêt boréale mixte, le climat est toutefois le plus significatif (de 70 à 90% de la variance expliquée). Ceci s’explique premièrement par le fait qu’il est déterminant dans la mise en place des conditions favorables au départ d’un feu: les anomalies de haute pression dessèchent rapidement les combustibles et elles sont suivies d’une entrée d’air chaud et instable accompagnée de foudre. Par la suite, le climat, principalement par l’intermédiaire du vent ou de la pluie, permet la propagation ou l’extinction de la flamme. Selon les analyses effectuées, la végétation aussi agit sur le comportement du feu (7 à 25% de la variance expliquée). Par exemple, la présence de feuillus ralentit la propagation de la flamme et atténue l'intensité dégagée; à l'opposé, la présence de conifères accélère la propagation de la flamme et accroît l'intensité dégagée. Ces résultats étant testés uniquement sur les argiles mésiques, il se pourrait que le rôle de la végétation ait été sous-estimé. En effet, la mosaïque forestière naturelle actuelle inclut d'autres substrats sur lesquels se développent des peuplements de pins gris, d’épinettes noires, de mélèzes ou de frênes, dont la variabilité en terme de composition et d'effets sur le comportement du feu est encore plus importante. De plus, la présence ou non de feuillage chez les essences décidues constitue une autre source saisonnière de variation locale de la végétation qui influence significativement le comportement du feu, les feux de printemps étant plus intenses que les feux d'été.

Cette thèse confirme que le système canadien du FBP (Fire Behavior Prediction) est un simulateur du comportement du feu très bien adapté à la forêt boréale mixte. En effet, les prédictions quantitatives du FBP concernant le comportement du feu (vitesse de propagation, intensité dégagée, et surface brûlée pour une période donnée) se rapprochent fortement des données mesurées sur le terrain par l'équipe de Douglas McRae lors de trois feux expérimentaux réalisés en Ontario. A l’opposé, le système américain BEHAVE sous-estime exagérément les prédictions quantitatives. La non-réalisation des feux expérimentaux prévus initialement aurait pu entraver la comparaison des deux simulateurs. Toutefois, cette étude a été conduite à terme grâce aux données de terrain des premiers feux expérimentaux réalisés en forêt boréale mixte par Douglas McRae. L'emploi de BEHAVE fut toutefois nécessaire et justifié pour tester la susceptibilité des peuplements au feu car, contrairement au FBP, le système BEHAVE intègre les différents types de combustibles de surface et leurs charges. De plus, la prédiction qualitative de BEHAVE relative à l'importance des facteurs climat et végétation se révèle quand même pertinente. Bien qu'étant le plus adapté à la forêt boréale mixte, le système du FBP pourrait bénéficier de nouvelles améliorations, rendues bientôt possibles grâce aux données des feux expérimentaux réalisés en Ontario, premières valeurs réelles enfin disponibles pour ce modèle général empirique.

Grâce à l'étude de mortalité post-incendie mise en place suite au feu naturel de printemps survenu en juin 1997 à Val Paradis (Abitibi), cette thèse confirme que les variations locales de la composition végétale au sein des peuplements, influencent non seulement le comportement du feu à cette échelle, mais également la mosaïque forestière à l'échelle du paysage. En effet, les hauteurs brûlées reflétant l'intensité dégagée sont, à classe de sévérité égale, significativement plus faibles chez les peuplements feuillus que chez les mixtes ou les résineux. Ainsi donc, si les peuplements de compositions différentes ne subissent pas le même comportement du feu, ni la même sévérité, ils ne réagiront pas de façon identique lors de la période de régénération post-incendie. Ces résultats concernant les relations entre la composition de la végétation et la perturbation feu contribuent à l'explication de la forte résilience de la foret boréale mixte en terme de composition forestière puisque les essences en place avant l'incendie créent des conditions de feux qui sont favorables à la régénération des mêmes espèces.

La classification synoptique des espèces de Brown et Davis (1973), basée sur leur sensibilité au feu, n'est pas remise en question puisque cette thèse démontre que *Populus tremuloides* est effectivement l'espèce la moins résistante au feu comparée à *Pinus banksiana* et *Picea mariana*. Par contre, l'analyse des variables dendrométriques dans la résistance au feu révèle l'importance capitale de la résistance cambiale chez les deux conifères étudiés. Cette résistance cambiale est le résultat d'interactions entre différentes variables morphologiques telles que le diamètre de l'arbre, l'épaisseur de l'écorce, et la hauteur de la base du houppier. Cette thèse montre que la survie de ces conifères est d'autant plus importante que la base du feuillage est proche du sol. En effet, l'écorce fine chez ces espèces rendant la résistance cambiale plus faible, le houppier, présent jusqu'à la base du tronc joue le rôle de barrière aux flammes et protège ainsi le cambium. Il est intéressant de noter que ce phénomène s'oppose aux observations constatées sur les conifères de l'ouest de l'Amérique du nord, pour lesquels la survie est plus forte lorsque les houppiers sont élevés. Chez ces espèces de l'Ouest, la survie du cambium étant assurée par une écorce épaisse, les fortes intensités des incendies dans ces régions, sélectionnent les arbres dont les houppiers élevés peuvent échapper aux flammes.

Le climat est donc reconnu comme le facteur qui délimite l'étendue de la forêt boréale et cette thèse démontre qu'il est également le facteur le plus important dans le comportement du feu en forêt boréale mixte. Or, dans la perspective des changements climatiques survenant actuellement, il est intéressant de voir quelles pourraient être les modifications survenant en forêt boréale mixte. Les climats passés (COHMAP Members 1988; Prentice *et al*. 1991) ont déjà présenté des variations de températures et d'humidité dont les amplitudes ont atteint ou dépassé les valeurs actuelles ou prédites pour un futur proche. Toutefois, la grande différence est que les changements actuels, principalement de sources anthropiques, présentent des vitesses incroyablement plus rapides que ceux survenus dans les temps géologiques (Harrington, 1987). Cela laisse à penser que la végétation n'aura pas partout le temps de s'adapter (vitesses de migration trop lentes), altérant alors les écosystèmes à travers leur structure et leur fonctionnement. La superficie de la forêt boréale mixte devrait diminuer en raison de son invasion rapide par les espèces tempérées plus au sud, non compensée par sa lente progression vers le nord (Weber et Stocks 1998). A cela s'ajoute l'augmentation des températures et des précipitations estivales dans l'est du Canada qui devrait diminuer la fréquences des feux (Bergeron et Flannigan 1995). Les perturbations naturelles font partie intégrante de l'écosystème et sont à l'origine du maintien de la diversité au sein de la mosaïque forestière. Une fréquence de feu plus faible se traduirait donc par un intervalle de feu plus long, entraînant un vieillissement de peuplements où la composante conifère augmenterait. Les peuplements mixtes et conifères domineraient alors la mosaïque. De plus, ces peuplements représenteraient alors des zones à la fois plus susceptibles au feu et aux épidémies de la tordeuse des bourgeons de l’épinette (TBE), et ils brûleraient plus intensément quand un feu surviendrait. La diversité des trouées créées par la TBE étant plus ou moins propice à la régénération des différentes espèces, cette compensation entre feux moins fréquents mais plus sévères et épidémies plus abondantes (voire aussi plus sévères) pourrait maintenir les différentes espèces dans la mosaïque. Cependant, cela pourrait créer une configuration différente de la mosaïque actuelle. En effet, au cours de la période plus longue entre deux feux, les parcelles mixtes pourraient être plus nombreuses et plus étendues, aux dépends des peuplements feuillus, confinés dans les trouées de tailles importantes, et des peuplements purs de conifères confinés aux îlots ayant échappé aux épidémies. Les peuplements de thuya devraient faire exception et proliférer dans la mosaïque, puisque cette espèce de fin de succession ne serait ni affectée par le feu ni la cible de la TBE. Les changements proposés au sein de la mosaïque sont cependant théoriques compte tenu du caractère non naturel des pratiques forestières actuelles (coupes surtout) utilisées dans la région de la sapinière à bouleau blanc.

L’aménagement durable écosystémique en forêt boréale mixte se met peu à peu en place dans le but d’améliorer son exploitation en créant des conditions proches des conditions naturelles rencontrées au sein des peuplements et de la mosaïque. L’exemple expérimental en cours dans la forêt d'enseignement et de recherche du lac Duparquet (Harvey 1999) tente de démontrer qu’il est possible de développer des pratiques sylvicoles qui s’inspireraient des effets des perturbations naturelles (Bergeron et Harvey 1997; Bergeron *et al*. 1999b). Les résultats inclus dans cette thèse peuvent concrètement participer à l’aménagement durable écosystémique qui se met en place dans cette région car ils présentent et/ou simulent la variabilité qui devrait être préservée par l'aménagement écosystémique et quantifient certains aspects.

Les débris au sol sont reconnus pour être l'habitat privilégié de nombreux organismes et micro-organismes et les chicots, résultant d'un feu ou de la mort naturelle des arbres, des habitats de choix pour de nombreux oiseaux (Bull *et al.* 1997; Crites et Dale 1998). Les pratiques sylvicoles retenues devraient donc, dans le cadre de la conservation de la biodiversité, maintenir en quantité suffisante ces habitats. Toutefois, la dynamique des débris ligneux (charges, compositions, et changements temporels) se révélant très différente en forêt boréale mixte des autres écosystèmes forestiers, il va sans dire que leur gestion devra être spécialement adaptée.

En raison de la variabilité du comportement du feu dans la mosaïque et de ses répercussions en terme de mortalité et de régénération post-incendie, il serait bon de développer des pratiques sylvicoles simulant à l'échelle du peuplement les conditions naturelles dues à un incendie. Au regard des analyses effectuées sur la mortalité post-incendie, le nombre d'arbres coupés ne devrait pas représenter la totalité des arbres présents, certains vivants ou morts devant rester dans les peuplements pour jouer le rôle de semenciers ou de chicots. L'impact sur les sols, conséquence de l'intensité atteinte lors du feu, devrait être pris en compte pour déterminer la proportion de sol minéral exposé dans les peuplements à dominance feuillue. La taille des parcelles de coupe ne devrait pas être uniforme mais plutôt s'inspirer des surfaces brûlées naturellement, les peuplements feuillus brûlant sur des surfaces plus petites que les peuplements mixtes ou résineux.

Les différentes pratiques forestières appliquées à l'échelle du peuplement devraient permettre de conserver, à l'échelle du paysage, la composition actuelle de la mosaïque naturelle. Une comparaison de la structure spatiale de la mosaïque forestière soumise aux coupes avec celle d'une mosaïque naturelle permettrait d'évaluer la pertinence d'un tel aménagement ainsi que la résilience de la mosaïque. Pour ce faire, le développement de modèles spatiaux du comportement du feu à l'échelle du paysage, fondés sur ceux existant déjà à l'échelle du peuplement, offrent des potentialités prometteuses. Ces outils permettraient de mieux comprendre le comportement des incendies et de raffiner notre compréhension de la dynamique naturelle de la forêt boréale mixte dans une perspective d'aménagement plus durable.

CHAPITRE VII

RÉFÉRENCES GÉNÉRALES

Agee, J. K. 1997. The severe weather wildfire **-** too hot to handle? Northwest Science **71** (1): 153-156.

Andrews, P. L. 1986. BEHAVE: fire behavior prediction and fuel modeling system--BURN subsystem, PART 1. U. S. Department of agriculture, Forest service, Ogden, UT. Intermountain forest and range experiment station, General technical report INT-194.

Attiwill, P.M. 1994. The disturbance of forest ecosystems: the ecological basis for conservation management. For. Ecol. Manage. **63**: 247-300.

Bergeron, Y. 1991. The influence of island and mainland lakeshore landscapes on boreal forest fire regimes. Ecology **72** (6): 1980-1992.

Bergeron, Y. 2000. Species and stand dynamics in the mixed-woods of Quebec's southern boreal forest. Ecology, In-press.

Bergeron, Y. et Dubuc, M. 1989. Succession in the southern part of the Canadian boreal forest. Vegetatio **79** : 51-63.

Bergeron, Y. et Archambault, S. 1993. Decreasing frequency of forest fires in the southern boreal zone of Quebec and its relation to global warming since the end of the "Little Ice Age". The Holocene **3** (3): 255-259.

Bergeron, Y. et Dansereau, P. R. 1993. Predicting the composition of Canadian southern boreal forest in different fire cycles. Journal of Vegetation Science **4** : 827-832.

Bergeron, Y. et Brisson, J. 1994. Effect of climatic fluctuations on post-fire regeneration of two jack pine and red pine populations during the twentieth century. Géographie Physique et Quaternaire 48 (2): 145-149.

Bergeron, Y. et Flannigan, M. D. 1995. Predicting the effects of climate change on fire frequency in the southern Canadian boreal forest. Water, Air, and Soil Pollution **82** : 437-444.

Bergeron, Y. et Harvey, B. 1997. Basing silviculture on natural ecosystem dynamics: an approach applied to the southern boreal mixedwood forest of Quebec. Forest Ecology and Management **92** : 235-242.

Bergeron, Y., Gauthier, S., Carcaillet, C., Flannigan, M. D., Prairie, Y. et Richard, P. J. H. 1999a. Variability in fire frequency and forest composition in Canada's southeastern boreal forest: a challenge for sustainable forest management. The sustainable forest management network. Science and Practice: Sustaining the boreal forest, Edmonton, Alberta, Canada: 74-80.

Bergeron, Y., Harvey, B., Leduc, A. et Gauthier, S. 1999b. Stratégies d'aménagement forestier qui s'inspirent de la dynamique des perturbations naturelles: considérations à l'échelle du peuplement et de la forêt. The Forestry Chronicle **75** (1): 55-61.

Bessie, W. C. et Johnson, E. A. 1995. The relative importance of fuels and weather on fire behavior in subalpine forest. Ecology **76** (3): 747-762.

Bilgili, E. 1995. "Fuel characterization and fire behavior prediction in even-aged conifer stands". Ph D Thesis, New Brunswick, Canada: 111p.

Brand, G.J., and Smith W.B., 1985. Evaluating allometric biomass equations fit to generated data. Can. J. Bot. 63: 64-67.

Brown, A. A. et Davis, K. P. 1973. Forest fire: control and use. McGraw-Hill book company, New York. 686p.

Brown, J.K., 1976. Estimating shrub biomass from basal stem diameters. Can. J. For. Res. **6**: 153-158.

Brown, J. K., Oberheu, R. D. et Johnston, C. M. 1982. Handbook for inventoring surface fuels and biomass in the interior West. US Forest Service, General Technical Report INT-129.

Brown, J. K. et Debyle, N. V. 1987. Fire damage, mortality, and suckering in aspen. Canadian Journal of Forest Research **17** : 1100-1109.

Buech, R.R., and Rugg, D.J., 1995. Biomass relations for components of five Minnesota shrubs. USDA, Forest Service, North central forest experiment station, research paper NC-325.

Bull, E. L., Parks, C. G., et Torgersen, T. R. 1997. Trees and logs important to wildlife in the interior Columbia river basin. U. S. Department of agriculture, Forest service, Portland, OR. Pacific northwest research station, General technical report PNW-GTR-391.

Burgan, R. E. et Rothermel, R. C. 1984. BEHAVE: fire behavior prediction and fuel modeling system--FUEL subsystem. U. S. Department of agriculture, Forest service, Ogden, UT. Intermountain forest and range experiment station, General technical report INT-167.

Clark, J. S. 1988. Effect of climate change on fire regimes in northwestern Minnesota. Nature **334** : 233-235.

COHMAP Members 1988. Climatic changes of the last 18000 years : observations and model simulations. Science **241** : 1043-1052.

Conseil Canadien des ministres des forêts, 1997. Critères et indicateurs de l'aménagement durable des forêts au Canada. Ressources naturelles Canada, Service canadien des forêts, Ottawa, Ontario, Rapport technique .

Crites, S. et Dale, M. R. T. 1998. Diversity and abundance of bryophytes, lichens, and fungi in relation to woody substrate and successional stage in aspen mixedwood boreal forests. Canadian Journal of Botany **76** : 641-651.

De Grandpré, L., Gagnon, D. et Bergeron, Y. 1993. Changes in the understory of Canadian southern boreal forest after fire. Journal of Vegetation Science **4** : 803-810.

Flannigan, M. D. 1993. Fire regime and the abundance of red pine. International Journal of Wildland Fire **3** (4): 241-247.

Flannigan, M. D. et Wotton, B. M. 1991. Lightning-ignited forest fires in northwestern Ontario. Canadian Journal of Forest Research **21** : 277-287.

Forestry Canada Fire Danger Group, 1992. Development and structure of the Canadian Forest Fire Behavior Prediction System. Forestry Canada, Ottawa, Ontario, Information report ST-X-2.

Franklin, J.F. 1993. Preserving biodiversity: species, ecosystems or landscapes. Ecological Applications **3**: 202-205.

Furyaev, V. V., Wein, R. W. et Mac Lean, D. A. 1983. Fire influences in *Abies*-dominated forests. *In The role of fire in Northern circumpolar ecosystems*. R. W. Wein, McLean, D. A. (éd.): 221-234.

Galindo-Leal, C. et Bunnell, F.L. 1995. Ecosystem management: implications and opportunities of a new paragdim. For. Chron. **71**: 601-606.

Galipeau, C., Kneeshaw, D., et Bergeron, Y. 1997. White spruce and balsam fir colonization of a site in the southeastern boreal forest as observed 68 years after fire. Can. J. For. Res. **27**: 139-147.

Gauthier, S., Leduc, A. et Bergeron, Y. 1996. Forest dynamics modelling under natural fire cycles: a tool to define natural mosaic diversity for forest management. Environmental Monitoring and Assessment **39** : 417-434.

Gouvernement du Québec 1996. Biodiversité du milieu forestier: bilan et engagements du Ministère des ressources naturelles. Gouvernement du Québec, Ministère des Ressources naturelles, Québec. 152p.

Green, D. G. 1983. Shapes of simulated fires in discrete fuels. Ecological Modelling **20** : 21-32.

Green, D. G., Gill, A. M. et Noble, I. R. 1983. Fire shapes and the adequacy of fire-spread models. Ecological Modelling **20** : 33-45.

Gutsell, S. L. et Johnson, E. A. 1996. How fire scars are formed: coupling a disturbance process to its ecological effect. Canadian Journal of Forest Research **26** : 166-174.

Harmon, M. E. 1984. Survival of trees after low-intensity surface fires in Great Smoky Mountains national park. Ecology **65** (3): 796-802.

Harmon, M. E., Franklin, F. J., Swanson, F. J., Sollins, P., Gregory, S. V., Lattin, J. D., Anderson, N. H., Cline, S. P., Aumen, N. G., J.R., S., Lienkaemper, G. W., Cromack, K. J. et Cummins, K. W. 1986. Ecology of Coarse Woody Debris in temperate ecosystems. Advances in Ecological Research **15** : 133-302.

Harrington, J. B. 1987. Climatic change : a review of causes. Canadian Journal of Forest Research **17** : 1313-1339.

Harrington, M. G. 1993. Predicting *Pinus ponderosa* mortality from dormant season and growing season fire injury. International Journal of Wildland Fire **3** (2): 65-72.

Harvey, B. 1999. The Lake Duparquet Research and Teaching Forest: Building a foundation for ecosystem management. The Forestry Chronicle **75** (3) : 389-393.

Heinselman, M. L. 1971. The natural role of fire in northern conifer forests. Fire in the Northern Environment. A symposium College, Alaska: 61-72.

Hirsch, K. G. 1996. Canadian Forest Fire Behavior Prediction (FBP) System: user's guide. Canadian forestry service, Northern forestry centre, Special report #7.

Huddle, J. A. et Pallardy, S. G. 1999. Effect of fire on survival and growth of *Acer rubrum* and *Quercus* seedlings. Forest Ecology and Management **118** : 49-56.

Jean, F. 1992. Modélisation du comportement du feu. Influence de la pente et de la charge d'une litière d'aiguilles de pin maritime. I.N.R.A., Avignon, PIF 9205.

Johnson, E. A. 1992. Fire vegetation dynamics : Studies from the North American boreal forest, Cambridge. 129p.

Johnson, E.A., and K. Miyanishi. 1995. The need of consideration of fire behavior and effects in prescribed burning. Restoration Ecology 3:271-278.

Keane, R. E., Arno, S. F. et Brown, J. K. 1990. Simulating cumulative fire effects in Ponderosa pine / Douglas-fir forests. Ecology **71** (1): 189-203.

Larsen, J. A. 1980. The Boreal Ecosystem. Academic Press, New York. 500p.

McRae, D. J., Alexander, M. E. et Stocks, B. J. 1979. Measurement and description of fuels and fire behavior on prescribed burns : a handbook. Canadian Forestry Service, 0-X-287.

Ordre des ingénieurs forestiers du Québec 1996. *Manuel de foresterie*. Communications Sciences - Impact. Les Presses de l'Université Laval. 1428p.

Overpeck, J. T., Rind, D. et Goldberg, R. 1990. Climate - induced changes in forest disturbance and vegetation. Nature **343** : 51-53.

Paré, D., Bergeron, Y. et Camiré, C. 1993. Changes in the forest floor of Canadian southern boreal forest after disturbance. Journal of Vegetation Science **4** : 811-818.

Paré, D. et Bergeron, Y. 1995. Above-ground biomass accumulation along a 230- years chronosequence in the southern portion of the Canadian boreal forest. Journal of Ecology **83** : 1001-1007.

Payette, S. et Gagnon, R. 1985. Late Holocene deforestation and tree regeneration in the forest-tundra of Québec. Nature **313** : 570-572.

Peterson, D. L. 1985. Crown scorch and scorch height: estimates of postfire tree condition. Canadian Journal of Forest Research **15** : 596-598.

Peterson, D. L. et Arbaugh, M. J. 1986. Postfire survival in Douglas-fir and lodgepole pine: comparing the effects of crown and bole damage. Canadian Journal of Forest Research **16** : 1175-1179.

Peterson, D. L. et Arbaugh, M. J. 1989. Estimating postfire survival of Douglas-fir in the Cascade range. Canadian Journal of Forest Research **19** : 530-535.

Post, L.J. 1969. Dry-matter production of Mountain maple and Balsam fir in northwestern New Brunswick. Ecology **51** (3): 548-550.

Prentice, I.C., Bartlein, P.J., et Webb III, T. 1991. Vegetation and climate change in eastern North America since the last maximum. Ecology **72** (6): 2038-2056.

Pyne, S.J. 1984. Introduction to Wildland Fire: Fire management in the United States. A wiley - Interscience publication, John Wiley & Sons, Inc. 455p.

Regelbrugge, J. C. et Conard, S. G. 1993. Modeling tree mortality following wildfire in *Pinus ponderosa* forests in the central Sierra Nevada of California. International Journal of Wildland Fire **3** (3): 139-148.

Reinhardt, E. D. et Ryan, K. C. 1988. How to estimate tree mortality resulting from underburning. Fire Management Notes **49** (4): 30-36.

Roussopoulos, P. J. 1978. An appraisal of upland forest fuels and potential fire behavior for a portion of the Boundary Waters Canoe area. Ph D Thesis, Michigan State University, Department of Forestry: 166p.

Ryan, K. C. 1998. Analysis of the relative value of morphological variables in predicting fire-caused tree mortality. III International Conference on forest fire research. 14th Conference on fire and forest meteorology, Luso, Portugal: 1511-1526.

Ryan, K. C., Peterson, D. L. et Reinhardt, E. D. 1988. Modeling long-term fire-caused mortality of Douglas-fir. Forest Science **34** (1): 190-199.

Ryan, K. C. et Reinhardt, E. D. 1988. Predicting postfire mortality of seven western conifers. Canadian Journal of Forest Research **18** : 1291-1297.

Ryan, K. C. et Frandsen, W. H. 1991. Basal injury from smoldering fires in mature *Pinus ponderosa* Laws. International Journal of Wildland Fire **1** (2): 107-118.

Schimmel, J. et Granström, A. 1997. Fuel succession and fire behavior in the Swedish boreal forest. Canadian Journal of Forest Research **27** : 1207-1216.

Schroeder, M.J. et Buck, C.C. 1970. Fire weather: a guide for application of meteorological information to forest fire control operations. USDA, Forest Service, Agriculture handbook 360.

Service Canadien des forêts, 1996. The state of Canada's forests :1995 -1996, OTTAWA. 112p.

Shugart, H. H., Leemans, R. et Bonan, G. B. 1992. A systems analysis of the global boreal forest. Cambridge University Press, Cambrigde. 565 p.

Simard, M-J., Bergeron, Y. et Sirois, L. 1998. Conifer seedling recruitment in a southeastern Canadian boreal forest: the importance of substrate. Journal of Vegetation Science **9**: 575-582.

Smith, C. K., Munson, A. D. et Coyea, M. R. 1998. Nitrogen and phosphorus release from humus and mineral soil under black spruce forests in central Quebec. Soil Biology and Biochemistry **30** (12): 1491-1500.

Solomon, A. M. 1992. The nature and distribution of past, present and future boreal forests : lessons for a research and modeling agenda. *In A systems analysis of the global boreal forest*. H. H. Shugart, Leemans, R., Bonan, G.B. (éd.). Cambrige University Press :291-307.

Stocks, B. J. 1987. Fire behavior in immature jack pine. Canadian Journal of Forest Research **17** : 80-86.

Swezy, D. M. et Agee, J. K. 1991. Prescribed-fire effects on fine-root and tree mortality in old-growth ponderosa pine. Canadian Journal of Forest Research **21** : 626-634.

Telfer, E. S. 1969. Weight-diameter relationships for 22 woody plant species. Canadian Journal of Botany **47** : 1851-1855.

Thomas, T. L. et Agee, J. K. 1986. Prescribed fire effects on mixed conifer forest structure at Crater Lake, Oregon. Canadian Journal of Forest Research **16** : 1082-1087.

Trabaud, L. 1976. Inflammabilité et combustibilité des principales espèces des garrigues de la région méditerranéenne. Acta Oecologica, Oecol. Plant. **11** (2): 117-136.

Trabaud, L. 1989. Les feux de forêts: Mécanismes, comportement et environnement. Editions France-Sélection, Aubervilliers, 278p.

UICN 1980. Union Internationale pour la conservation de la nature et de ses ressources. Stratégie mondiale de la conservation: la conservation des ressources vivantes au service du développement durable, Suisse.

Valette, J. C. 1990. Inflammabilité, indice de végétation et température de surface. I.N.R.A, Avignon, PIF 9016.

Van Der Maarel, E. 1993. Some remarks on disturbance and its relations to diversity and stability. Journal of Vegetation Science **4** : 733-736.

Van Wagner, C. E. 1973. Height of crown scorch in forest fires. Canadian Journal of Forest Research **3** : 373-378.

Van Wagner, C. E. 1977. Conditions for the start and spread of crown fire. Canadian Journal of Forest Research **7** : 23-34.

Van Wagner, C. E. 1983. Fire behaviour in Northern conifer forests and shrublands. *In The role of fire in Northern Circumpolar Ecosystems.* R. W. Wein. et D. A. Mclean (éd.) scope 18: 65-81.

Weber, M.G. et Stocks, B.J. 1998. Forest fires and sustainability in the boreal forests of Canada. Ambio 27 (7): 545-550.

Weber, M.G. et Flannigan, M.D. 1997. Canadian boreal forest ecosystem structure and function in a changing climate : impact on fire regimes. Environmental Reviews 5: 145-166.

Wein, R. S. et McLean, D. A. 1983. Fire and succession in the conifer forests of Northern North America. *In The role of fire in Northern Circumpolar Ecosystems.* J. W. a. Sons (éd.), New York : 374-405.

Zasada, J.C., Sharik, T.L., et Nygren, M. 1992. The reproductive process in boreal forest trees. Dans A system analysis of the global boreal forest. edité par H.H. Shugart, R. Leemans et G.B. Bonan. Cambridge University Press, Cambridge: 85-125.

Appendix 2.1 Characteristics of sampled stands

Stand no.	Time since last fire (years)	Total living density (stems/ha)	Total living basal area (m^2/ha)	Deciduous living basal area (%)	Total log loads (t/ha)	Total basal snag area (m^2/ha)	Total volume of snags (m^3/ha)	Total snag density (stems/ha)
p0301	32	2382	51.27	100	27.7	17.41	81.57	6897
p0302	32	1978	42.8	100	22.6	3.58	25.26	2122
p0303	32	1234	35.33	100	26.2	7.15	125.13	531
b0551	52	1072	39.41	70	35.4	13.8	42.45	1592
b0552	52	1478	18.98	96	43.61	15.59	28.84	1061
p0551	52	1055	22.43	93	27.59	8.46	17.26	1592
p0552	52	1901	37.53	96	26.96	2.67	5.6	531
b0801	80	2309	64.94	77	111.51	2.87	8.41	1592
b0802	80	1139	45.06	72	70.02	6.69	62.59	1061
p0801	80	3422	125.57	93	89.22	0.00	0.00	0
p0802	80	2823	60.49	92	17.8	9.81	18.4	1061
b1201	126	1702	31.81	81	51.1	55.85	294.63	4244
b1202	126	1581	25.91	74	26.02	0.00	0.00	0
b1203	126	2168	21.2	81	51.9	9.6	63.87	1061
b1204	126	1057	25.8	52	95.58	0.00	0.00	0
p1201	126	1205	47.73	92	29.76	14.84	15.59	1061
p1202	126	1265	35.81	71	24.05	32.51	32.51	1061
p1203	126	753	44.08	81	64.86	16.92	74.01	2122
p1204	126	632	54.67	78	31.37	0.00	0.00	0
p1205	126	712	70.96	91	65.35	2.28	10.27	531
b1701	173	1065	15.5	36	69.76	8.76	83.22	531
b1702	173	878	22.83	78	67.86	0.00	0.00	0
b1703	173	1650	29.57	29	35.64	15.52	97.78	531
p1701	173	1160	31.4	68	56.75	5.9	19.18	1061
p1702	173	1646	26.69	78	68.14	41.5	254.53	1592
p1703	173	861	32.36	77	36.1	25.2	251.14	1592
p1704	173	825	64.38	85	60.99	7.64	32.09	1592
p1705	173	1800	39.5	84	82.56	107.89	708.5	3183
p1706	173	1111	47.09	96	41.2	7.89	34.72	1592
p1707	173	748	31.66	95	46.6	74.93	569.44	3183
p1708	173	627	32.14	68	42.42	35.06	358.35	2653
p1709	173	1430	57.41	82	56.96	70.82	540.57	1592
p1710	173	1045	50.9	93	39.63	23.95	196.38	1592
b2101	236	444	30.44	59	70.49	84.39	485.84	3714
b2102	236	673	37.74	65	54.9	11.9	29.75	531
b2103	236	775	22.98	49	103.42	19.75	147.1	1061
b2104	236	1084	24.06	59	108.31	25.43	149.18	1592
p2101	236	906	52.66	54	44.49	10.17	70.15	1061
p2102	236	570	37.15	56	41.13	22.68	123.98	1592

Appendix 2.1 Continued

Stand no.	Time since last fire (years)	Total living density (stems/ha)	Total living basal area (m^2/ha)	Deciduous living basal area (%)	Total log loads (t/ha)	Total basal snag area (m^2/ha)	Total volume of snags (m^3/ha)	Total snag density (stems/ha)
p2103	236	1002	68.88	70	46.4	6.86	37.76	1592
s2101	236	999	23.3	21	46.89	47.87	380.6	3714
s2102	236	961	26.3	18	34.85	14.31	64.39	1592
t2101	236	820	36.33	34	47.23	33.49	159.09	2122
t2102	236	578	68.12	22	41.98	20.99	185.4	1592
t2103	236	1086	15.1	0	44.68	26.43	179.36	2122
t2104	236	1288	33.03	7	47.97	33.78	340.29	2122
t2105	236	437	43.58	11	26.77	10.97	67.47	1061
t2106	236	1136	37.33	10	52.24	21.55	272.63	1061

Appendix 2.2 Characteristics of sampled logs: number of woody counts per class for the first five diameter size classes, and sum of square diameters per species for the sixth diameter size class

Stand	Number of wood pieces in Class					Sum of square diameters per species in the Class VI				
no.	I	II	III	IV	V	fir	birch	spruce	aspen	cedar
p0301	285	157	132	50	32	0	0	0	890	0
p0302	705	228	127	26	13	0	0	0	537	0
p0303	850	372	208	36	8	0	0	0	289	0
b0551	385	114	85	19	11	1312	1762	0	0	0
b0552	580	195	102	15	11	0	3014	0	204	0
p0551	345	129	78	37	28	59	146	0	1284	0
p0552	565	210	121	24	18	0	50	0	1638	0
b0801	940	145	94	21	18	535	5628	0	8909	0
b0802	705	183	87	30	16	1351	263	0	7488	848
p0801	220	111	65	16	13	1716	1624	0	11202	0
p0802	315	96	79	12	21	0	282	0	587	0
b1201	545	154	133	13	14	1008	3599	0	0	0
b1202	605	142	97	10	18	1329	483	0	0	0
b1203	890	220	88	13	9	2236	2350	149	1029	0
b1204	960	205	133	25	11	2089	7879	0	0	231
p1201	315	134	86	24	14	1288	145	0	1875	0
p1202	390	133	74	20	5	240	132	139	2045	0
p1203	695	193	135	17	9	1009	420	0	8309	0
p1204	345	137	85	12	14	1263	860	0	999	0
p1205	595	172	73	30	9	1924	659	533	6446	0
b1701	1080	198	101	17	27	3314	456	1389	3863	0
b1702	1035	217	97	49	22	2590	3220	149	576	0
b1703	1470	230	120	33	15	1802	420	0	0	0
p1701	1060	245	124	19	13	5221	1024	0	400	250
p1702	800	238	140	32	29	5111	243	0	3605	0
p1703	865	138	80	17	15	2337	0	0	1741	317
p1704	770	237	111	33	26	2550	636	1001	3255	0
p1705	710	176	127	38	28	3675	594	0	7441	0
p1706	860	304	161	44	23	1683	0	0	1228	388
p1707	990	264	85	36	31	1903	0	0	2814	0
p1708	800	235	116	28	22	2113	365	916	793	0
p1709	1420	277	114	47	25	1623	1011	77	2874	0
p1710	1020	259	111	32	19	2097	803	0	345	0
b2101	730	258	107	35	67	1565	2819	773	0	168
b2102	1760	266	151	28	12	4563	86	0	0	1368
b2103	740	206	163	20	20	6480	3622	0	0	5716
b2104	690	161	89	17	14	689	3214	1156	0	1399
p2101	1470	319	155	27	16	3206	135	0	625	106

Appendix 2.2 Continued

Stand no.	Number of wood pieces in Class					Sum of square diameters per species in the Class VI				
	I	II	III	IV	V	fir	birch	spruce	aspen	cedar
p2102	940	143	89	12	6	2877	189	1146	1024	0
p2103	1400	313	137	15	11	3680	703	0	586	135
s2101	1165	330	142	34	28	4638	0	0	0	0
s2102	1270	224	104	20	13	2639	0	0	0	1174
t2101	1125	247	126	13	7	2559	0	2614	0	811
t2102	900	202	129	17	14	2876	1138	0	0	541
t2103	1475	337	153	6	14	4193	59	502	0	279
t2104	2020	329	164	31	12	4495	0	0	0	178
t2105	1065	192	106	15	4	1140	650	0	625	291
t2106	1345	221	152	28	17	4622	1076	0	0	67

Appendix 2.3 Species percentages of sampled logs for the different diameter classes. f. b. s. a., and c are abbreviations for fir, birch, spruce, aspen, and cedar respectively

Stand no.	Species percentages in the first five CWD diameter classes														
	Class I					Class II					Classes III, IV, and V				
	f.	b.	s.	a.	c.	f.	b.	s.	a.	c.	f.	b.	s.	a.	c.
p0301	0	0	0	1	0	0	0	0	1	0	0	0	0	1	0
p0302	0	0.2	0	0.8	0	0	0.2	0	0.8	0	0	0.2	0	0.8	0
p0303	0	0	0	1	0	0	0	0	1	0	0	0	0	1	0
b0551	0.1	0.9	0	0	0	0.4	0.6	0	0	0	0.4	0.6	0	0	0
b0552	0	1	0	0	0	0	0.9	0	0.1	0	0	1	0	0	0
p0551	0	0.2	0	0.8	0	0	0.2	0	0.8	0	0	0.2	0	0.8	0
p0552	0	0	0	1	0	0	0.1	0	0.9	0	0	0	0	1	0
b0801	0.3	0.7	0	0	0	0.2	0.8	0	0	0	0.2	0.5	0	0.3	0
b0802	0.2	0.7	0	0.1	0	0.2	0.7	0	0.1	0	0.2	0.7	0	0.1	0
p0801	0.1	0.1	0	0.8	0	0.1	0.1	0	0.8	0	0.1	0.1	0	0.8	0
p0802	0.2	0	0	0.8	0	0.2	0	0	0.8	0	0.2	0	0	0.8	0
b1201	0.6	0.4	0	0	0	0.6	0.4	0	0	0	0.6	0.4	0	0	0
b1202	0.5	0.5	0	0	0	0.5	0.5	0	0	0	0.4	0.6	0	0	0
b1203	0.3	0.7	0	0	0	0.3	0.7	0	0	0	0.4	0.6	0	0	0
b1204	0.6	0.3	0	0	0.1	0.5	0.4	0	0	0.1	0.4	0.6	0	0	0
p1201	0.3	0	0	0.7	0	0.3	0	0	0.7	0	0.4	0	0	0.6	0
p1202	0.3	0.2	0	0.5	0	0.3	0.2	0	0.5	0	0.3	0.2	0	0.5	0
p1203	0.5	0	0	0.5	0	0.5	0	0	0.5	0	0.5	0	0	0.5	0
p1204	0.2	0.3	0	0.5	0	0.2	0	0	0.8	0	0.2	0	0	0.8	0
p1205	0.1	0.1	0	0.6	0.2	0.3	0	0	0.7	0	0.2	0.1	0	0.6	0.1
b1701	0.6	0.4	0	0	0	0.5	0.5	0	0	0	0.5	0.5	0	0	0
b1702	0.5	0.5	0	0	0	0.6	0.4	0	0	0	0.6	0.4	0	0	0
b1703	0.7	0.2	0	0.1	0	0.7	0.3	0	0	0	0.6	0.4	0	0	0
p1701	0.3	0	0	0.7	0	0.4	0	0	0.6	0	0.4	0	0	0.6	0
p1702	0.5	0	0	0.5	0	0.6	0	0	0.4	0	0.5	0	0	0.5	0
p1703	0.8	0	0	0.2	0	0.5	0	0	0.5	0	0.5	0	0	0.5	0
p1704	0.3	0	0	0.7	0	0.6	0	0	0.4	0	0.5	0	0	0.5	0
p1705	0.3	0	0	0.7	0	0.4	0	0	0.6	0	0.3	0	0	0.7	0
p1706	0.3	0	0	0.7	0	0.4	0	0	0.6	0	0.4	0	0	0.6	0
p1707	0.5	0	0	0.4	0.1	0.8	0	0	0.2	0	0.8	0	0	0.2	0
p1708	0.4	0	0	0.6	0	0.3	0	0.1	0.6	0	0.6	0	0	0.4	0
p1709	0.6	0.2	0	0.2	0	0.7	0	0	0.3	0	0.6	0.1	0	0.3	0
p1710	0.3	0	0	0.7	0	0.5	0	0	0.5	0	0.4	0	0	0.6	0
b2101	0.1	0.9	0	0	0	0.4	0.6	0	0	0	0.6	0.4	0	0	0
b2102	0.5	0.2	0.2	0	0.1	0.4	0.3	0	0	0.3	0.4	0.3	0	0	0.3
b2103	0.6	0.4	0	0	0	0.5	0.5	0	0	0	0.9	0.1	0	0	0
b2104	0.2	0.7	0	0	0.1	0.6	0.4	0	0	0	0.4	0.6	0	0	0
p2101	0.7	0	0	0.3	0	0.6	0	0	0.3	0.1	0.5	0	0	0.4	0.1
p2102	0.2	0.6	0.1	0.1	0	0.1	0.7	0.1	0.1	0	0.5	0.2	0.1	0.2	0

Appendix 2.3 Continued

Stand no.	Species percentages in the first five CWD diameter classes														
	Class I					Class II					Classes III, IV, and V				
	f.	b.	s.	a.	c.	f.	b.	s.	a.	c.	f.	b.	s.	a.	c.
p2103	0.6	0.1	0	0.1	0.2	0.7	0.1	0	0.1	0.1	0.6	0.1	0	0.2	0.1
s2101	0.7	0.1	0	0.2	0	0.9	0	0	0.1	0	1	0	0	0	0
s2102	0.7	0.1	0	0	0.2	0.5	0.2	0	0	0.3	0.9	0	0	0	0.1
t2101	0.2	0.3	0	0	0.5	0.3	0.3	0.1	0	0.3	0.4	0.1	0	0	0.5
t2102	0.3	0	0	0.1	0.6	0.5	0	0	0.1	0.4	0.5	0.1	0	0.1	0.3
t2103	0.7	0.2	0	0	0.1	0.8	0.1	0	0	0.1	0.8	0	0	0	0.2
t2104	0.6	0.1	0	0	0.3	0.7	0.1	0	0	0.2	0.6	0.2	0	0	0.2
t2105	0.1	0	0	0	0.9	0.8	0.1	0	0	0.1	0.9	0.1	0	0	0
t2106	0.8	0.1	0	0	0.1	0.8	0	0	0	0.2	0.9	0	0	0	0.1

Note: the calculated multiplication factors for white cedar are : class I = 0.0002; class II = 0.0020
class III = 0.0047; class IV = 0.0141; class V = 0.0113; class VI = 0.0004.

Appendix 3.1 Stand and surface fuel characteristics of the forty-eight sampled stands.

Stand no.	Total live tree basal area (m^2/ha)	Percent conifer basal area	Total live tree density (stems/ha)	Time since last fire (years)	Total dead wood load (t/ha)	L. depth (cm)	L. load (kg/m^2)	L. bulk density (kg/m^3)	D. bulk density (kg/m^3)	1-hour time lag (t/ha)	10-hour time lag (t/ha)	100-hour time lag (t/ha)
p0301	51.27	0	2382	32	27.7	0.62	0.429	69.6	152.3	0.06	1.18	0.00
p0302	42.80	0	1978	32	22.6	1.33	0.451	33.8	33.7	0.07	2.40	2.30
p0303	35.33	0	1234	32	26.2	1.46	0.435	29.8	81.0	0.08	0.99	1.75
b0552	18.98	4	1478	52	43.6	1.92	0.235	12.2	61.1	0.05	1.35	3.17
p0552	37.53	4	1901	52	27.0	2.90	0.467	16.1	77.2	0.06	0.93	0.00
p1706	47.09	4	1111	173	41.2	2.13	0.435	20.5	62.3	0.15	1.79	1.31
p1707	31.66	5	748	173	46.6	2.33	0.387	16.6	68.8	0.02	0.03	0.00
p0551	22.43	7	1055	52	27.6	1.46	0.336	23.0	96.6	0.04	0.83	0.00
p0801	125.57	7	3422	80	89.2	2.25	0.440	19.6	65.0	0.04	0.38	1.57
p1710	50.90	7	1045	173	39.6	3.08	0.539	17.5	54.3	0.30	0.32	0.00
p0802	60.49	8	2823	80	17.8	2.07	0.413	20.0	67.2	0.10	0.44	1.46
p1201	47.73	8	1205	126	29.8	1.83	0.491	26.8	94.3	0.10	0.39	0.70
p1205	70.96	9	712	126	65.4	2.63	0.357	13.6	77.0	0.19	0.07	0.00
p1704	64.38	15	825	173	61.0	2.58	0.304	11.8	71.5	0.23	1.29	0.64
p1705	39.50	16	1800	173	82.6	2.38	0.333	14.0	93.2	0.31	1.48	0.00
p1709	57.41	18	1430	173	57.0	2.75	0.552	20.1	65.0	0.15	0.03	0.00
b1201	31.81	19	1702	126	51.1	2.92	0.372	12.8	68.6	0.03	0.32	0.00
b1203	21.20	19	2168	126	51.9	2.46	0.445	18.1	85.4	0.08	3.88	3.56
p1203	44.08	19	753	126	64.9	1.96	0.539	27.5	67.7	0.20	0.10	0.00
p1204	54.67	22	632	126	31.4	1.63	0.491	30.2	61.6	0.06	0.03	0.00
b1702	22.83	22	878	173	67.9	2.13	0.547	25.7	72.4	0.31	0.39	0.00
p1702	26.69	22	1646	173	68.1	2.71	0.413	15.3	89.8	0.33	5.56	0.96

Appendix 3.1 Continued.

Stand no.	Total live tree basal area (m^2/ha)	Percent conifer basal area	Total live tree density (stems/ha)	Time since last fire (years)	Total dead wood load (t/ha)	L. depth (cm)	L. load (kg/m^2)	L. bulk density (kg/m^3)	D. bulk density (kg/m^3)	1-hour time lag (t/ha)	10-hour time lag (t/ha)	100-hour time lag (t/ha)
b0801	64.94	23	2309	80	111.5	3.08	0.469	15.2	66.6	0.08	0.57	0.00
p1703	32.36	23	861	173	36.1	2.42	0.419	17.3	51.2	0.02	0.02	0.00
b1202	25.91	26	1581	126	26.0	1.75	0.557	31.8	52.9	0.07	0.59	0.00
b0802	45.06	28	1139	80	70.0	2.75	0.507	18.4	78.2	0.17	1.76	1.68
p1202	35.81	29	1265	126	24.1	2.00	0.523	26.1	124.9	0.10	2.26	0.00
b0551	39.41	30	1072	52	35.4	1.25	0.523	41.8	77.8	0.09	0.66	0.92
p2103	68.88	30	1002	236	46.4	2.38	0.611	25.7	77.8	0.68	3.05	4.77
p1701	31.40	32	1160	173	56.8	2.08	0.387	18.6	83.6	0.12	2.08	2.89
p1708	32.14	32	627	173	42.4	3.13	0.544	17.4	63.2	0.17	0.34	0.00
b2102	37.74	35	673	236	54.9	2.38	0.411	17.3	68.1	0.47	2.32	6.36
b2101	30.44	41	444	236	70.5	3.00	0.573	19.1	74.2	0.15	3.36	0.00
b2104	24.06	41	1084	236	108.3	2.29	0.139	6.1	59.4	0.16	1.47	0.00
p2102	37.15	44	570	236	41.1	1.90	0.517	27.2	52.5	0.04	2.33	11.78
p2101	52.66	46	906	236	44.5	2.71	0.491	18.1	60.3	0.31	6.00	0.00
b1204	25.80	48	1057	126	95.6	1.83	0.453	24.7	77.4	0.24	4.11	5.70
b2103	22.98	51	775	236	103.4	2.04	0.187	9.1	73.7	0.05	1.43	3.48
b1701	15.50	64	1065	173	69.8	2.90	0.408	14.1	72.7	0.31	2.27	0.79
t2101	36.33	66	820	236	47.2	3.13	0.531	17.0	76.0	0.37	4.01	0.00
b1703	29.57	71	1650	173	35.6	2.00	0.475	23.7	73.0	0.03	0.00	0.00
t2102	68.12	78	578	236	42.0	2.08	0.472	22.7	68.7	0.22	1.46	8.43
s2101	23.30	79	999	236	46.9	2.92	0.533	18.3	51.5	0.56	4.13	0.00
s2102	26.30	82	961	236	34.9	0.75	0.237	31.6	58.6	0.16	1.73	0.00

Appendix 3.1 Continued.

Stand no.	Total live tree basal area (m^2/ha)	Percent conifer basal area	Total live tree density (stems/ha)	Time since last fire (years)	Total dead wood load (t/ha)	L. depth (cm)	L. load (kg/m^2)	L. bulk density (kg/m^3)	D. bulk density (kg/m^3)	1-hour time lag (t/ha)	10-hour time lag (t/ha)	100-hour time lag (t/ha)
t2105	43.58	89	437	236	26.8	1.75	0.424	24.2	75.0	0.67	1.64	1.40
t2106	37.33	90	1136	236	52.2	2.08	0.627	30.1	91.2	0.12	2.07	20.16
t2104	33.03	93	1288	236	48.0	2.38	0.451	19.0	80.0	0.55	2.73	12.88
t2103	15.10	100	1086	236	44.7	2.13	0.400	18.8	72.3	0.45	4.41	2.76

Note: only stems greater than 5 cm in DBH were measured. L. = litter; D. = duff.

Appendix 3.2 Species composition (%) of dead wood material less than 7 cm in diameter.

Stand no.	Percent conifer basal area	Time since fire (yrs)	Percent of species composition of dead wood material by diameter size class														
			Class I (0 - 0.5 cm)					Class II (0.5 -1 cm)					Classes III, IV, and V (1 - 7 cm)				
			a	b	s	c	f	a	b	s	c	f	a	b	s	c	f
p0301	0	32	100	0	0	0	0	100	0	0	0	0	100	0	0	0	0
p0302	0	32	80	20	0	0	0	77	23	0	0	0	77	23	0	0	0
p0303	0	32	100	0	0	0	0	100	0	0	0	0	100	0	0	0	0
b0552	4	52	0	100	0	0	0	10	90	0	0	0	0	100	0	0	0
p0552	4	52	100	0	0	0	0	93	7	0	0	0	100	0	0	0	0
p1706	4	173	70	0	0	0	30	57	0	0	0	43	57	0	0	0	43
p1707	5	173	43	0	0	7	50	23	0	0	0	77	23	0	0	0	77
p0551	7	52	80	20	0	0	0	80	20	0	0	0	80	20	0	0	0
p0801	7	80	87	10	0	0	3	87	10	0	0	3	87	10	0	0	3
p1710	7	173	67	0	0	0	33	47	0	0	0	53	60	0	0	0	40
p0802	8	80	80	0	0	0	20	80	0	0	0	20	80	0	0	0	20
p1201	8	126	70	0	0	0	30	70	0	0	0	30	60	0	0	0	40
p1205	9	126	67	3	0	20	10	70	0	0	0	30	63	3	0	3	30
p1704	15	173	70	0	0	0	30	43	0	0	0	57	47	0	0	0	53
p1705	16	173	70	0	0	0	30	60	0	0	0	40	67	0	0	0	33
p1709	18	173	23	17	0	0	60	33	0	0	0	67	33	7	0	0	60
b1201	19	126	0	37	0	0	63	0	37	0	0	63	0	37	0	0	63
b1203	19	126	0	67	0	0	33	0	67	0	0	33	0	57	0	0	43
p1203	19	126	50	0	0	0	50	50	0	0	0	50	50	0	0	0	50
p1204	22	126	50	33	0	0	17	83	0	0	0	17	83	0	0	0	17
b1702	22	173	0	53	0	0	47	0	40	0	0	60	0	43	0	0	57
p1702	22	173	47	0	0	0	53	40	0	0	0	60	50	0	0	0	50
b0801	23	80	0	67	0	0	33	0	77	0	0	23	27	50	0	0	23
p1703	23	173	20	0	0	0	80	47	0	0	0	53	47	0	0	0	53
b1202	26	126	0	53	0	0	47	0	53	0	0	47	0	60	0	0	40
b0802	28	80	7	73	0	0	20	7	73	0	0	20	10	67	0	0	23
p1202	29	126	53	17	0	0	30	53	17	0	0	30	53	17	0	0	30
b0551	30	52	0	93	0	0	7	0	63	0	0	37	0	63	0	0	37
p2103	30	236	13	3	0	20	63	12	3	0	8	77	15	10	0	12	63
p1701	32	173	67	0	0	0	33	63	0	0	0	37	57	0	0	0	43
p1708	32	173	63	0	0	0	37	57	0	7	0	37	40	0	0	0	60

Appendix 3.2 Continued.

Stand no.	Percent conifer basal area	Time since fire (yrs)	Percent of species composition of dead wood material by diameter size class														
			Class I (0 - 0.5 cm)					Class II (0.5 -1 cm)					Classes III, IV, and V (1 - 7 cm)				
			a	b	s	c	f	a	b	s	c	f	a	b	s	c	f
b2102	35	236	0	23	23	10	43	0	27	0	33	40	0	33	0	23	43
b2101	41	236	0	90	0	0	10	0	60	0	0	40	0	43	0	0	57
b2104	41	236	0	70	0	13	17	0	43	0	0	57	0	60	0	0	40
p2102	44	236	7	60	10	0	23	13	67	7	0	13	23	17	13	0	47
p2101	46	236	33	0	0	0	67	33	0	0	7	60	37	0	0	10	53
b1204	48	126	0	37	0	7	57	0	40	0	3	57	0	60	0	0	40
b2103	51	236	0	37	0	0	63	0	50	0	0	50	0	10	0	0	90
b1701	64	173	0	43	0	0	57	0	47	0	0	53	0	47	0	0	53
t2101	66	236	0	37	0	47	17	0	27	17	30	27	0	3	0	50	47
b1703	71	173	3	27	0	0	70	0	27	0	0	73	0	40	0	0	60
t2102	78	236	7	0	0	57	37	3	0	0	47	50	3	10	0	33	53
s2101	79	236	20	3	0	0	77	3	0	0	0	97	0	0	0	0	100
s2102	82	236	0	8	0	18	73	0	17	0	27	57	0	0	0	3	97
t2105	89	236	0	0	0	87	13	0	3	0	10	87	0	13	0	0	87
t2106	90	236	0	7	0	13	80	0	2	0	15	83	0	0	0	3	97
t2104	93	236	0	3	0	30	67	0	7	0	23	70	0	17	0	17	67
t2103	100	236	0	23	0	10	67	0	10	0	13	77	0	0	0	17	83

Note: a, b, s, c, and f are abbreviations for aspen, birch, spruce, cedar, and fir, respectively.

Appendix 4.1 Characteristics of the research burn plots

Characteristic		Plot #1	Plot #2	Plot #3
Density (no. tree/ha)				
	Stand living density	3148	2474	2242
	Stand living coniferous density	2456	1659	1601
	Stand living deciduous density	692	815	641
	Stand snag density	88	15	6
Basal area (m^2/ha)				
	Stand living basal area	23.17	22.4	29.59
	Stand living coniferous basal area	11.87	17.66	19.08
	Stand living deciduous basal area	11.3	4.74	10.51
	Tree conifer percentage based on BA	51.23	78.83	64.48
	Stand type based on classification	MC	C	MC
Prescribed burning				
	Time spent to burn 1 ha (min)	16	11	38
	Total slash comsumed (kg/m^2)	0.35	0.24	0.08
	Total duff comsumed (kg/m^2)	0.43	0.59	0.56
	Total fuel consumed (kg/m^2)	0.78	0.84	0.64
	Total heat release (kJ/m^2)	14909	15785	11832
Fire behavior				
	Rate of spread (m/min)	8.84	12.86	3.72
	Head fire Intensity (kW/m)	2236	3420	789

Note: BA = basal area, MC = Mixed-coniferous, and C = Coniferous stands.