UNIVERSITÉ DU QUÉBEC À MONTRÉAL

THE EFFECT OF CLIMATE CHANGE ON BOREAL FOREST FIRE SEVERITY AND THE POTENTIAL CONSEQUENCES FOR THE BOREAL FOREST CARBON STOCK IN QUÉBEC, CANADA

RAPPORT DE SYNTHÈSE ENVIRONNEMENTALE
PRÉSENTÉ
COMME EXIGENCE PARTIELLE
DU DOCTORAT EN SCIENCES DE L'ENVIRONNEMENT

PAR
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List of scientific names

Scientific name	Common name
Abies balsamea (L.) Mill.	Balsam fir
Alnus crispa (Ait.) Pursh.	Green alder
Betula glandulosa Michx.	Dwarf birch
Betula papyrifera Marsh.	Paper birch
Carex sp.	Sedge species
Ceratodon purpureus (Hedw.) Brid.	Fire moss
Choristoneura fumiferana Clem.	Spruce budworm
Cladina sp.	Reindeer lichen species
Cladonia stellaris (Opiz.) Pouz. et Vezda	Cup lichen
Cladonia sp.	Cup lichen species
Cornus stolonifera Michx.	Red osier dogwood
Equisetum sylvaticum L.	Woodland horsetail
Ericaceae sp.	Ericaceous species
Kalmia angustifolia L.	Sheep laurel
Kalmia polifolia Wang.	Pale laurel
Larix laricina (Du Roi) K. Koch	Eastern larch
Picea glauca (Moench) Voss.	White spruce
Picea mariana (Mill.) BSP	Black spruce
Pinus banksiana Lamb.	Jack pine
Pleurozium schreberi (Brid.) Mitt.	Red-stemmed feather moss
Polytrichum sp.	Hair moss species
Populus balsamifera L.	Balsam poplar
Populus tremuloides Michx.	Trembling aspen
Rhododendron groenlandicum (Oeder)	Labrador tea
Sphagnum sp.	Peat moss species
Thuja occidentalis L.	White cedar
Vaccinium angustifolium Ait.	Lowbush blueberry
Vaccinium myrtilloides Michx.	Velvet-leaf blueberry

List of abbreviations

C	Carbon
CFFWIS	Canadian Forest Fire Weather Index System
DC	Drought Code
DMC	Duff Moisture Code
FFMC	Fine Fuel Moisture Code
F horizon	Upper duff
FWI	Fire Weather Index
GCM	Global Circulation Model
RCM	Regional Circulation Model
H horizon	Deeper duff
L horizon	Litter layer
LIA	Little Ice Age
N	Nitrogen
NEE	Net Ecosystem Exchange
Ppm	Parts per million

Preface

This literature study was performed as part of a Ph.D. study on the influence of Holocene peat fires on carbon accumulation in three ombrotrophic bogs in the James Bay region of Québec, Canada. The question to be answered here was:

Which are the factors that may cause an increase in future fire severity in eastern Canada and what could be the impact of this new fire severity regime on the boreal forest organic matter and carbon balance (at the level of forest regeneration as well as the loss of carbon by burning)?

I have tried to answer this question by synthesizing available literature on this subject.

Abstract

The boreal forest comprises a number of biomes that possess large quantities of terrestrial carbon (C). A change in climate will be largest in the higher northern latitudes and can be an important factor in determining fire occurrence in these regions, being the most important disturbance factor in the boreal biomes. Of all fire traits, fire severity is the one that directly affects both direct C release to the atmosphere as well as the potential for postfire vegetation establishment.

Using climate scenarios and scenarios for future forest floor contents obtained from the scientific literature, I investigated the factors that influence the fire severity regime and estimated the effect of climate change on the occurrence of severe burning and the effects on the C stock for each of the Québec boreal biomes.

The scenarios show an increase in both mean summer temperature and mean summer precipitation for most of the Québec boreal forest. In addition, the increase in winter precipitation may influence the deep forest floor moisture contents and cause the potential for deep burning to shift during the fire season. A lower-fire severity regime was confirmed by Fire Weather Index scenarios that indicate a general lower intensity of spreading fires. The occurrence of less frequent severe fires may well induce thicker forest floors and lower tree biomass, resulting in increasing boreal C stocks. However, the direct influence of increasing summer temperatures may cause an increase in tree production.

The moisture index used was not directly applicable in estimating forest floor consumption by fire. In addition, the link between fire severity and fire frequency needs to be taken into account in determining the long-term boreal C stocks. Therefore, further research is desired to obtain more precise scenarios on the potential for future high-severity fire presence as well as a better picture of long-term stand development concerning C dynamics in the absence of fire.

Keywords: fire severity, carbon dynamics, boreal forest, climate change, forest floor, duff consumption, Québec.

1 Introduction

The boreal forest biomes (boreal forest, peatlands and tundra) contain approximately 882 Pg carbon (C) in living biomass, detritus and soil (Apps *et al.*, 1993), being 49 to 64 % of earth's forest C (Kasischke, 2000; Lal, 2005). The global boreal forests are supposed current sinks of C at a magnitude of 707 Tg C yr^{-1} (Apps *et al.*, 1993), with 25 % of the sequestered C fixed by the North American boreal biomes. A 43-year average of 82 Tg C yr^{-1} has been reported for boreal North America but negative values (i.e. a source) can be present in years with high fire activity (Balshi *et al.*, 2007). This results in great temporal as well as regional variations in C exchange within each biome, depending on vegetation, climate and fire regimes. Typical C emissions from a single fire are ~ 1.0 kg C m^{-2} for the ensemble of the Québec boreal biomes from 1959 to 1999 (Amiro *et al.*, 2001). Without disturbance boreal forests potentially accumulate 0.010 to 0.040 kg C m^{-2} y^{-1}, expressed as Net Ecosystem Exchange (NEE) (Kurz and Apps, 1999; O'Connell *et al.*, 2003). These data show the importance of ecosystem C release by fire relative to the annual ecosystem C accumulation (Figure 1.1).

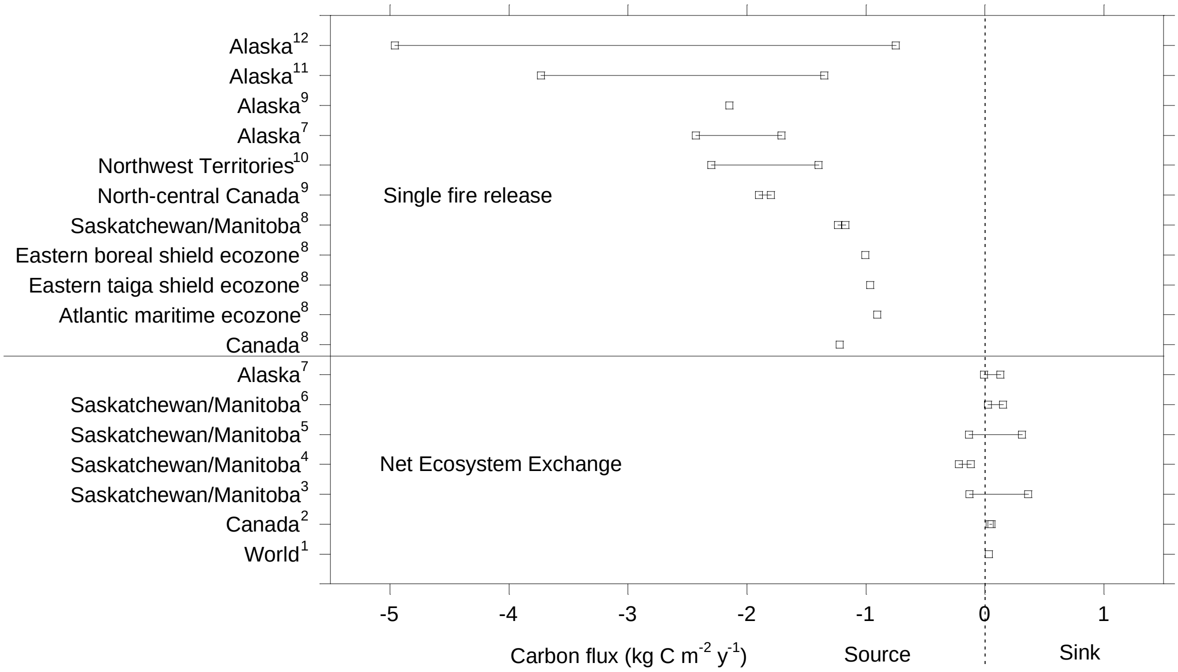

Figure 1.1: NEE and fire C fluxes for a number of boreal regions. [1]Pregitzer and Euskirchen (2004); [2]Kurz and Apps (1999); [3]Amiro *et al.* (2006); [4]O'Connell *et al.* (2003); [5]Bond-Lamberty *et al.* (2004); [6]Litvak *et al.* (2003); [7]French *et al.* (2003); [8]Amiro *et al.* (2001); [9]Turquety *et al.* (2007); [10]Stocks *et al.* (2004); [11]Tan *et al.* (2007); [12]Kasischke and Johnstone (2005).

Boreal forest biomes contain C in living plant biomass, plant detritus and soil. The boreal forest floor forms a large potential stock of organic C due to the low decomposition rates of organic matter that are typical for the northern boreal ecosystems. In addition, the forest floor determines the potential for vegetation growth by its temperature, humidity and nutrient content characteristics.

Fire is the main disturbance factor in the boreal forest, influencing vegetation production and decomposition processes and thus C dynamics in several ways (Lal, 2005). The release of organic C to the atmosphere occurs directly by (a partial) consumption by burning of the living biomass as well as the forest floor, defined as the ensemble of L (litter) and F and H (duff) layers above the mineral soil. Mean surface fuel consumption per fire in Canada's boreal forest was shown to be around four times larger (~ 1.0 kg C m^{-2}) than crown fuel consumption (~ 0.25 kg C m^{-2}) (Amiro *et al*., 2001), indicating the importance of the forest floor in studying boreal forest C dynamics. The term fire severity (i.e. depth of burn) is an indication of the proportion of the forest floor affected by fire as well as the ecological effects of fire on the plant community and the habitat (Payette, 1992). A high fire severity indicates a deep burning of the forest floor and thus a comparatively large release of C to the atmosphere. Considering C dynamics, the long-term effects of fire are important as well, as fires strongly determine forest re-establishment of vegetation on the production side as well as forest floor processes on the decomposition side of the C balance. The release of C during a single fire event is much greater than the net yearly C accumulation in absence of disturbance (potentially a factor 100, Figure 1.1) indicating the importance of fire severity and fire frequency on the boreal C stock. In this study, the focus will be on both the loss of ecosystem C during a fire event and the long-term effects of fire severity on stand development and the stand C contents, typically being in the range of 0 to 50 years for initial tree establishment, but possibly increasing to several hundreds of years, at least if disturbance remains absent.

The aim of the research is to estimate the effect of climate change on future fire severity and the consequences for the boreal C stock in Québec. The secondary questions are: (1) what is the importance of the forest floor on boreal forest development and carbon balance? (2) what determines fire severity and what is its influence on postfire vegetation re-establishment? (3) which are the climate and fire regime scenarios and how do they influence the future C stock in the boreal biomes of Québec?

The reaction of the various boreal biomes to a changing fire severity resulting from a region-specific change in climate can be studied at several time and spatial scales. In this study, the ensemble of the boreal biomes was divided into four regions, roughly following the classification scheme by Payette (1992). From south to north, the regions are the southern closed-crown forest, the northern closed-crown forest, the lichen woodland and the forest-tundra. These regions are distinguished by specific vegetation communities and patterns that are shaped by typical postfire regeneration processes while fire regimes are primarily climatically controlled (Payette, 1992), with a secondary role for vegetation composition and forest structure. The interest in the province of Québec as focus in the research is because of the particular climate conditions as compared to western Canada combined with the extensive presence of the various boreal biomes. The timeframe on which variations in climate were observed was around 60 years, i.e. the present boreal forest characteristics were compared with the expected situation around 2050 - 2075. This is because most studies on future climate scenarios use the 2050 situation as a reference (Flannigan *et al*., 2001). Flannigan *et al*. (2001) modeled the 2050 fire regimes supposing the atmospheric carbon dioxide (CO_2) concentration would attain 660 ppm by then; this would represent a doubling of the 330 ppm-CO_2 concentration of the 1960 – 1980 period. However, IPCC (2007) reported a 2040 – 2060 atmospheric CO_2 concentration of approximately 525 ppm and do preview a 660 ppm concentration only around 2075. Therefore, I will refer to the period 2050 – 2075 when discussing the $2 \times CO_2$ scenario.

In chapter 2 I will discuss the importance of the forest floor in boreal C dynamics including C distribution and the effect of the forest floor on vegetation development. Subsequently, fire severity as a characteristic of fire and an index to estimate forest floor moisture contents is treated as well as the influence of a varying general fire severity on postfire vegetation re-establishment. In chapter 4 a number of climate scenarios and fire occurrence for 2050 - 2075 are evaluated; chapter 5 discusses the potential vegetation structure and C stock of the Québec boreal biomes in the mid-21st century.

2 The importance of the forest floor in boreal forest C dynamics

The forest floor is defined here as the organic matter resting on a mineral base, comprising the litter (undecomposed, L horizon) and duff (the slightly decomposed F horizon and the well-decomposed H horizon) layers (Johnson, 1992; Gleixner *et al*., 2005). The litter, upper and deeper duff layers each have specific decomposition rates, bulk densities, moisture contents and drying rates (Miyanishi and Johnson, 2002; Table 2.1).

2.1 Forest floor C accumulation

The forest floor organic C stock is a function of input and loss of organic C. The rate of litter input is a function of stand density and composition. Litter is mainly composed of leaves and twigs from the over- and understorey vegetation. High litter input rates are generally found in productive, warm stands in the southern boreal biomes as well as in young stands with thin forest floors because of a high tree biomass production (Lecomte *et al*., 2006). However, despite both spatial and temporal variations in litterfall, the amount of organic matter accumulated on the mineral soil depends more strongly on the strength of local decomposition processes (Miyanishi, 2001), as forest floors are generally thicker in cool, wet sites (Johnson, 1981) and in old stands with low productivity compared to young stands with high productivity (Wardle *et al*., 2003). Thus, high temperatures and low moisture contents generally cause higher decomposition rates. In addition, species-specific litter quality characteristics including pH as well as the nature and abundance of decomposing organisms potentially influence decomposition rates (Coûteaux *et al*., 1995; Krause, 1998). Generally, coniferous leaves decompose more slowly than deciduous leaves, and the leaves from a poor-nutrient environment decompose more slowly than the leaves from the same species in a nutrient-rich stand (Chapin *et al*., 2002; Wardle *et al*., 2003). Thus, litter and duff decomposition rates vary with climate, hydrology and vegetation type between and within stands (Flanagan and Van Cleve, 1983).

a)

<table>
<tr><th rowspan="3"></th><th colspan="3">Bulk density (g cm^{-3})</th><th colspan="5">Thickness (cm)</th></tr>
<tr><th rowspan="2">FWI</th><th colspan="2">Black spruce stand</th><th rowspan="2">FWI</th><th rowspan="2">Aspen stand</th><th rowspan="2">Jack pine stand</th><th rowspan="2">Jack pine – black spruce stand</th><th rowspan="2">Black spruce stand</th></tr>
<tr><th></th><th>360 year old</th></tr>
<tr><td>Litter</td><td>0.021 [1]</td><td></td><td>0.032 [3]</td><td>1.2 [1]</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Upper duff</td><td>0.071 [1]</td><td>0.03 [2]</td><td rowspan="2">0.084 [3]</td><td>7.0 [1]</td><td rowspan="2">8.0 [4]</td><td rowspan="2">5.0 [4]</td><td rowspan="2">6.9 [3]</td><td rowspan="2">10.3 [3]
20 [4]</td></tr>
<tr><td>Deeper duff</td><td>0.139 [1]</td><td>0.22 [2]</td><td>18 [1]</td></tr>
</table>

b)

<table>
<tr><th rowspan="3"></th><th colspan="8">C contents (g cm^{-2})</th></tr>
<tr><th rowspan="2">FWI</th><th colspan="2" rowspan="2">Trembling aspen stand</th><th colspan="2" rowspan="2">Jack pine stand</th><th colspan="3">Black spruce stand</th></tr>
<tr><th></th><th>Sphagnum cover</th><th>Feather moss cover</th></tr>
<tr><td>Litter</td><td>0.013 [1]</td><td></td><td rowspan="3">0.194 [5]
0.159 [5]
0.283 [6]</td><td></td><td rowspan="3">0.146 [5]
0.181 [5]
0.132 [6]</td><td></td><td rowspan="3">0.844 [7]</td><td rowspan="3">0.331 [7]</td></tr>
<tr><td>Upper duff</td><td>0.250 [1]</td><td rowspan="2">0.255 [4]</td><td rowspan="2">0.194 [4]</td><td rowspan="2">0.435 [4]</td></tr>
<tr><td>Deeper duff</td><td>1.250 [1]</td></tr>
</table>

Table 2.1.a: Litter and duff layer mean bulk density and layer thickness for a variety of stand types; b. Litter and duff layer mean C contents. [1]Van Wagner (1987); [2]Kasischke *et al*. (2005); [3]Lecomte *et al*. (2006); [4]Yu *et al*. (2002); [5]Gower *et al*. (1997); [6]Nalder and Wein (1999); [7]O'Connell *et al*. (2003).

The amount of C stored in the forest floor was shown to be well correlated with the thickness of the duff layer (Yu *et al*., 2002) and increases downwards due to compaction and humification (Kasischke *et al*., 2005). Underneath the forest floor C can be present in the mineral soil (the A horizon) by percolating rainwater or digging soil organisms (Gleixner *et al*., 2005).

2.2 The forest floor as a C stock

Boreal ecosystems generally have large C stocks that have the potential to influence atmospheric composition by release of e.g. CO_2 and methane (CH_4) in varying climate regimes (Bonan *et al*., 1995). The boreal forest biomass, forest floor and mineral soil C vary strongly with stand age, climate, species composition and disturbance regimes (Nalder and

Wein, 1999; Bhatti and Apps, 2000; Tremblay *et al*., 2002). Mean forest floor C stocks of 3.7 kg C m^{-2} were reported from around 100 sites in boreal central Canada (Bhatti and Apps, 2000) while a mean 4.4 kg C m^{-2} was obtained by modelling sites in Québec (Tremblay *et al*., 2002); the difference may be the result of less severe burning and a wetter climate in the Québec boreal regions. During stand development, the major C stock generally moves from living biomass to the forest floor (Wardle *et al*., 1997; Lecomte *et al*., 2006). As a result, forest floor C values potentially constitute over half of the stand C content in the absence of fire (Dixon *et al*., 1994; Gower *et al*., 1997; Figure 2.1).

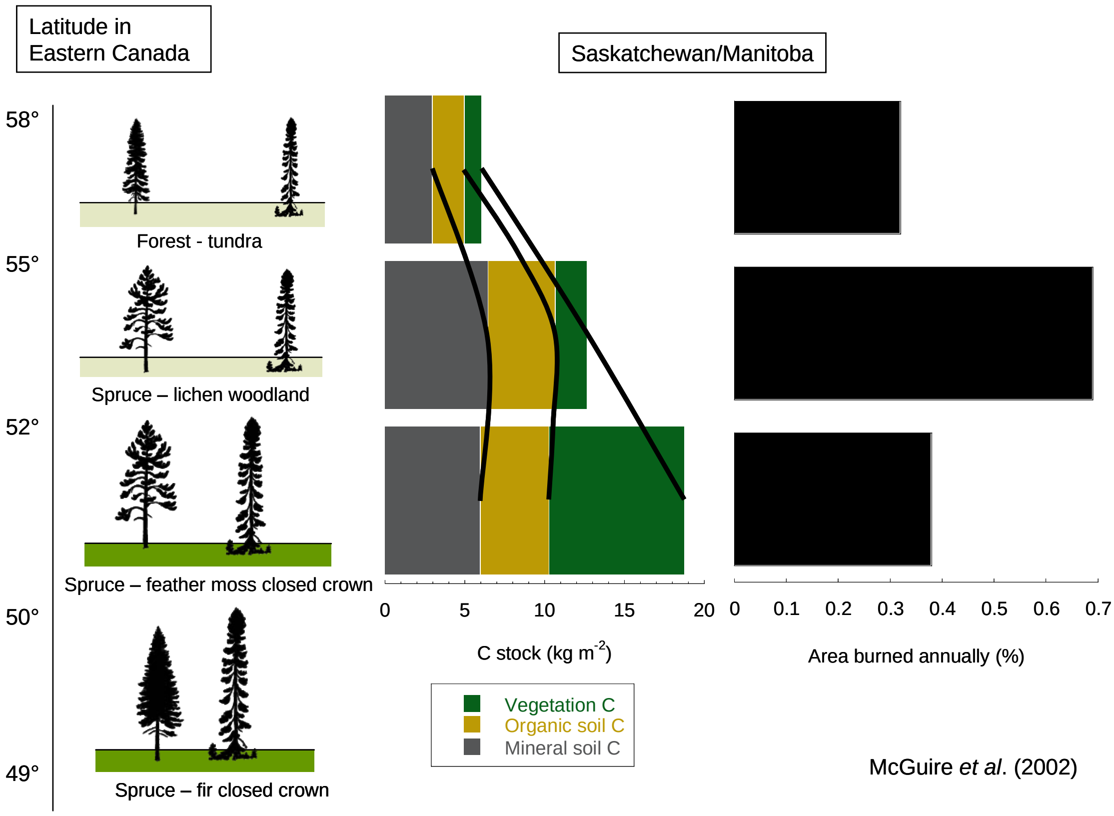

Figure 2.1: Boreal forest biomes with corresponding C allocation and burning patterns.

Mineral soils have been found to potentially contain 23 to 88 % of the total stand C (Gower *et al*., 1997). However, the high relative C content of 87 and 88 % measured probably included the forest floor C fraction. Excluding these data, mineral C would constitute 23 to 55 % of the total stand C. Despite the fact that there is an obvious link between forest components considering C storage, the components will be treated separately here. This is primarily because each forest layer will respond in a different way to disturbance and to changes within the system itself. At the long time scale, climate determines vegetation patterns at the regional or continental level, while at short time scales, disturbance like fire or insect outbreaks can alter stand development. Fire is a typical short-scale process, potentially removing C from both plant biomass and forest floor, but can cause on longer time scales a renewed increase in total stand C resulting from stand re-establishment.

The forest floor C dynamics at a local scale can be typified by the occurrence of a fire at the start of the cycle. This fire is characterized by a depth of burn that determines the amount of C released from the forest floor. After the fire, vegetation re-establishes during a fire-free period, as forest floor fuels build up until a new passage of fire. The fact that boreal forests are generally accumulating C on the long term suggests that the slow net C accumulation in periods lacking disturbance should be slightly higher than the rapid C loss during fires (Figures 1.1 and 2.2). On the landscape scale, a shift in general fire severity e.g. influenced by large-scale climatic variability may imply a larger average C loss during fire and a shift to a net C source. However, the complexity of the vegetation reaction on differential fire severity needs to be taken into account.

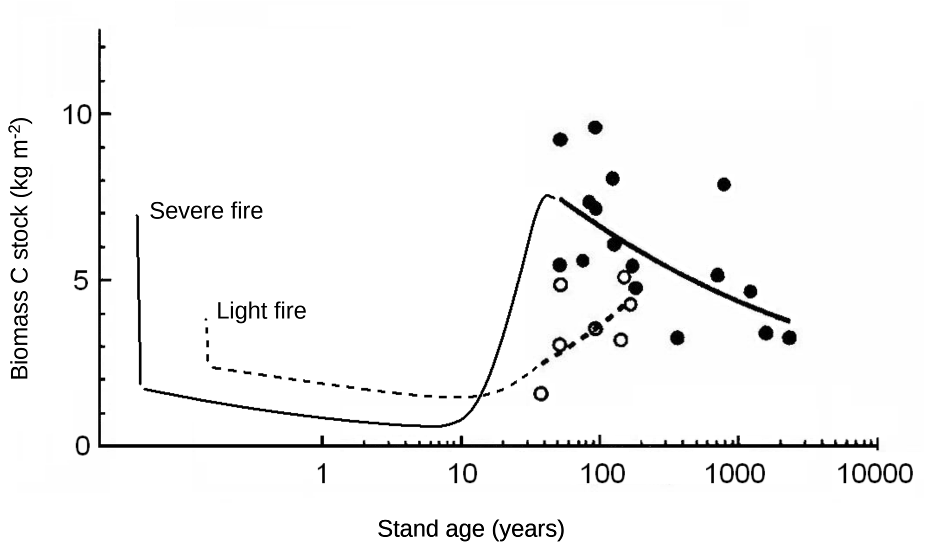

Figure 2.2: A model of the development of a boreal ecosystem biomass C stock in time with the occurrence of a severe fire (solid line) and a light fire (dashed line), based on data by O'Neill *et al*. (2003), Litvak *et al*. (2003), Kasischke and Johnstone (2005), Lecomte *et al*. (2006) and Simard *et al*. (2007).

The amount of C released during fire is to a great extent dependent on where C is located: in biomass, forest floor or mineral soil, especially when a single fire type is dominant, which is the case with stand-replacing crown fires in North America (Johnson, 1992). C release from the ground layer constitutes between 41 and 73 % of the total C emitted in a fire, being an average for the entire North American boreal forest (Kasischke *et al*., 2005). Of the total C present in the stand organic matter, an average of 24 % of the aboveground (vegetation) C and

5 % of the ground layer (soil) is lost in a single fire in the eastern Canadian ecozones, which include the Boreal Shield East and the Taiga Shield East (Balshi *et al*., 2007; Figure 2.3). However, forest fires in Québec are suspected to be less severe than forest fires in western Canada (Balshi *et al*., 2007) and therefore the percentages by Kasischke *et al*. (2005) may be too high when considering only the Québec biomes. As heat does hardly penetrate into the mineral soil (Schimmel and Granström, 1996), C stored in the mineral soil will not be released even during the most severe fires (Neff *et al*., 2005). For this reason, the mineral soil is not studied here. However, plant resprouting can take place from rhizomes present in the mineral soil (Schimmel and Granström, 1996), showing that the mineral soil is not completely inert concerning organic C dynamics.

Figure 2.3: The Canadian ecozones (redrawn from Amiro *et al*., 2004).

2.3 The forest floor as a factor in vegetation development

The forest floor is an important factor in C storage patterns through postfire plant regeneration by determining composition, density and spatial distribution of overstorey trees (Miyanishi and Johnson, 2002). Trees typically establish within 3 to 9 years after fire, including the early successional jack pine (*Pinus banksiana* Lamb.) and black spruce (*Picea mariana* (Mill.) BSP) (DesRochers and Gagnon, 1997; Gutsell and Johnson, 2002). In addition, however especially in western Canada, the length of the fire cycle can be close to the length of most boreal tree species' life span, underlining the importance of first-arriving tree species on the boreal forest composition. Further, the thickness of the forest floor (e.g. after fire disturbance) is a major factor in determining the potential for tree establishment. For example, jack pine as well as balsam poplar (*Populus balsamifera* L.) establish easily on mineral soil deprived of any duff (Chrosciewicz, 1974; Comtois and Payette, 1987), while the more shade-tolerant black spruce grows moderately well in remaining duff (Johnson, 1992); balsam fir (*Abies balsamea* (L.) Mill.) shows the greatest probability in mesic stands that developed from mixedwood stands (Messaoud *et al*., 2007). Foster (1985) found that the period of seedling establishment in black spruce forests was much longer in sites with a charred forest floor, producing a multi-aged postfire stand. In contrast, the combination of high fire frequency and high-severity fires in central and western Canada cause the complete removal of the forest floor, resulting in favourable seedbeds and even-aged, dense stands. On the understorey level, some plants have the capacity to resprout from belowground adventitious buds (e.g. trembling aspen (*Populus tremuloides* Michx.), *Rhododendron groenlandicum* (Oeder) and *Equisetum sylvaticum* L.), with the chance of survival depending on their position (depth) within the forest floor and the severity of the fire (Rowe, 1983; Viereck, 1983; Schimmel and Granström, 1996). The storage of seeds in the forest floor is a common technique for some trees, herbs and shrubs (e.g. *Cornus stolonifera* Michx.) (Rowe, 1983; Nguyen-Xuan *et al*., 2000; Wirth, 2005). Another important trait of the forest floor that influences the potential for vegetation development is the availability of nutrients. Nitrogen (N) is the nutrient most limiting to production in boreal forests (Pastor and Mladenoff, 1992). Leaf litter in the coniferous forests has general low N and high lignin contents (Flanagan and Van Cleve, 1983), providing a poorly decomposable organic accumulation on the forest floor and consequently lower N amounts for production. During stand development, the low availability of N in the forest floor may therefore cause a continuing decrease in forest productivity. In addition, low nutrient amounts in the forest floor cause a decrease in forest floor decomposition rates (Wardle *et al*., 1997).

Black spruce stands were shown to have among the highest forest floor C contents (Tremblay *et al*., 2002; Table 2.2) with distinct F and H layers (Miyanishi and Johnson, 2002). In addition, they are characterized by low forest floor temperature, pH and nutrient availability and a high C/N ratio as compared to jack pine stands (Krause, 1998). While some of these traits are the result of lower decomposition rates, some also positively influence the slowing down of decomposition processes, e.g. low N concentrations and low pH (Moore, 1981), thus functioning as a positive feedback. Deciduous forests generally have the lowest forest floor C (Tremblay *et al*., 2002).

	Mean stand forest floor and mineral soil organic C content (g cm^{-2})					
	Deciduous stands	Jack pine stands	Mixed stands	Balsam fir stands	Black spruce stands	Mean
Forest floor	0.39 (0.08-1.28)	0.41 (0.07-1.16)	0.44 (0.13-1.06)	0.47 (0.16-1.18)	0.58 (0.10-1.23)	0.45 (0.07-1.28)
Mineral soil	0.65 (0.01-2.63)	0.44 (0.02-1.33)	0.59 (0.01-1.72)	0.66 (0.02-2.79)	0.60 (0-2.49)	0.62 (0-2.79)

Table 2.2: Forest floor and mineral soil organic C contents for various Québec stand types (Tremblay *et al*., 2002).

To conclude, the forest floor is a major factor in determining the C stored in boreal ecosystems as indicated by the relative large amounts of C storage in the forest floor and the effect of forest floor thickness, composition and density on vegetation production and decomposition processes by regulating temperature, humidity and nutrient contents and providing a storage for seeds and roots.

3 The importance of fire severity in boreal forest C dynamics

C can be released to the atmosphere through smouldering of the duff layer (Johnson, 1992). Even when fires are infrequent, the potentially large amounts of C released by fire cause the majority of the boreal ecosystems to be only minor sinks of C. The boreal forest's spatial and temporal potential of deep burning and consequent C loss is discussed here.

It is important to consider the severity of the last fire while evaluating forest floor development and C contents as is shown in several studies (Viereck, 1983; Nalder and Wein, 1999; Lecomte *et al*., 2005; Kasischke and Johnstone, 2005). Fire severity influences the forest floor C amounts firstly by the removal of organic matter and secondly by determining postfire dominant overstorey and understorey vegetation C allocation. In addition, there is evidence that fire severity is more important than the re-establishing species as a factor in determining postfire organic matter accumulation (Lecomte *et al*., 2006).

Fire severity shows a legacy that influences the local C stock on long time scales by providing advantages to some vegetation types but disadvantages to others. Despite the fact that fire frequency was not part of the area of research, it should be taken into account here as it determines the maximum period of vegetation development and thus C accumulation pattern resulting from the severity of the last fire. Lecomte *et al*. (2006) demonstrated that forest floor thickness and standing tree biomass values of an old (> 250 years) stand resulting from a severe fire can be similar to that of younger (< 150 years) stands established after a low-severity fire. More generally, stand age has highly significant effects on both vegetation biomass and forest floor C contents (Wang *et al*., 2003). One could conclude from this that fire severity is an important factor in determining long-term C dynamics in boreal forests only if local fire frequencies are sufficiently high to prevent a stand from developing a postfire vegetation structure that is equal after both high- and low-severity fires. However, the large-scale distribution of the Québec boreal biomes is at least partially the result of differing fire regimes, including both fire severity and fire frequency as important components. Payette (1992) concluded that the lichen woodland is a stable biome with characteristic vegetation that can exist as small patches within the closed-crown boreal forest under the same fire regimes and climate conditions. This and the fact that most species that establish within 65 years after fire can remain dominant for 250 years in the absence of fire in various biomes

(Morneau and Payette, 1989; Sirois and Payette, 1991) show that it is important to acknowledge the influence of fire severity, even when fire cycles are long.

Fire severity can be independent of fire frequency, as fire frequencies are primarily determined by climate and weather conditions (Flannigan *et al*., 2001), while fire severity is more dependent on forest floor characteristics (especially bulk density, moisture content and depth of the duff layer) (Miyanishi and Johnson, 2002). So, in theory, sites that burn very infrequently (e.g. fire cycles > 250 years) can burn deeply into the duff layer if this layer has very low moisture contents. However, there seems to be a link between the frequency a site burns and the depth of the remaining forest floor after fire. Sites that burn infrequently, e.g. as found in the Québec closed-crown forest, often possess thick, humid forest floors that prevent deep burning during the next fire (Foster, 1985). In contrast, sites in western boreal Canada have comparably higher fire frequency regimes, and the short period between each fire causes a high probability that a subsequent fire will burn down to the mineral soil, as insufficient time was available to accumulate an organic layer with high moisture content retaining ability. On the other hand, the link between fire severity and fire frequency is perturbed by several other factors, including climate, postfire vegetation species, and mineral soil type, that determine the potential pathways of postfire vegetation establishment, so I conclude that fire severity and fire frequency need to be regarded separately concerning C dynamics (Figure 3.1).

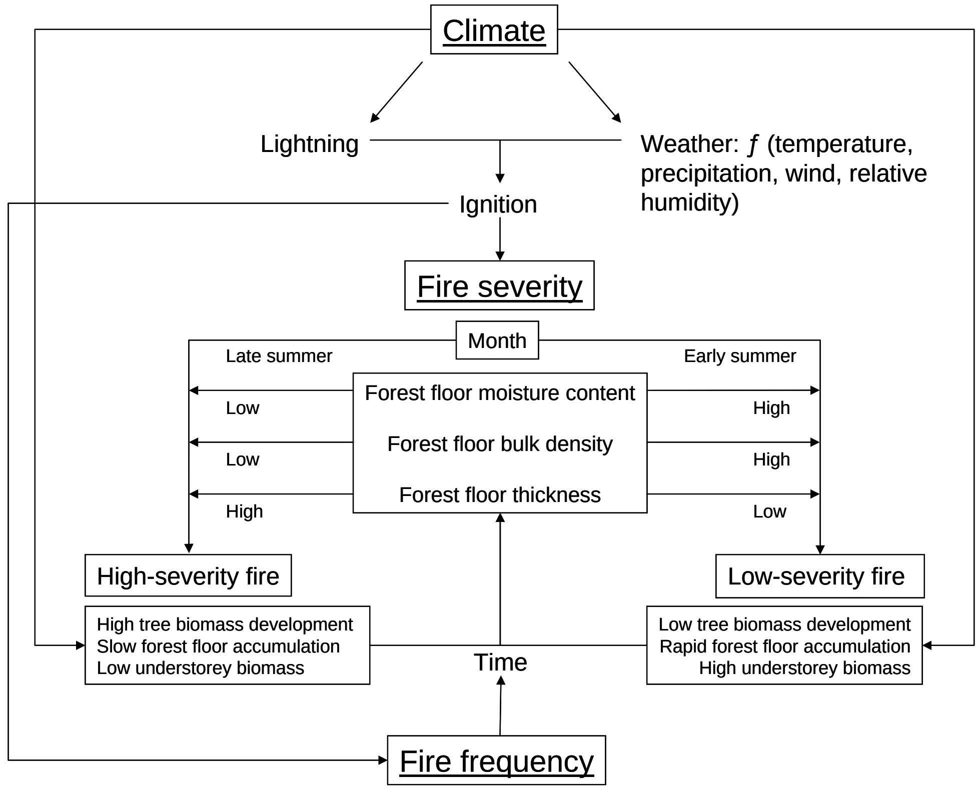

Figure 3.1: Links between climate, stand vegetation and fire frequency and severity effects.

3.1 The use of the Fire Weather Index in distinguishing forest floor moisture patterns

To estimate the effect of precipitation on litter and duff moisture contents, data from the Fire Weather Index (FWI) were studied. The FWI is a system of daily indexes to estimate fuel moisture and fire danger in a uniform fuel type (Flannigan *et al.*, 2001) that is part of the Canadian Forest Fire Weather Index System (CFFWIS). The index is based on the moisture content of the different forest floor horizons, taking into account the effect of wind on fire behaviour (Van Wagner, 1987). The FWI is usually calculated on daily (noon) measurements of temperature, relative humidity, wind speed and rain. The FWI has a good correlation with the "intensity of a spreading fire" (Bergeron and Flannigan, 1995) and it was shown to be a good predictor of the probability of short-duration sustained flaming in various types of fuel (Beverly and Wotton, 2007). Fire intensity indicates the rate at which heat is given off by the flame (Johnson, 1992); a definition that has been interpreted as an indication of the survival of aboveground plant parts (Schimmel and Granström, 1996). Despite the fact that plant death and fire spreading rates depend to a great extent on the intensity of the fire, the FWI can not directly be used in estimating the fire severity, i.e. the depth of burning into the forest floor. This is because the fire severity is related to some specific characteristics of the forest floor as forest floor density and thickness (Miyanishi and Johnson, 2002) that are not linked to the FWI.

The FWI index includes the *fine fuel moisture code* (FFMC) for litter and fine fuel moisture contents, the *duff moisture code* (DMC) for the upper duff layer and the *drought code* (DC) for the more compact lower duff. These are numerical ratings of mean moisture content indicating the potential of fire consumption. The FFMC, DMC and DC show different response rates to the same climatic conditions, taking into account differences in drying rates (Miyanishi and Johnson, 2002). In studying climate effects on fire severity, duff smouldering should be the major interest because of its importance in C and vegetation dynamics. The DMC and DC indirectly reflect the potential for deep smouldering.

Originally, the FWI was developed to represent the moisture content of forest floors in a mature, closed-canopy pine stand (Van Wagner, 1987; Wotton and Beverly, 2007). However, the observed forest floor moisture content varies between different stand types while having the same moisture code values, resulting from variations in forest type, stand density and season (Wotton and Beverly, 2007). In estimating forest floor moisture contents while using the FWI components it is therefore important to regard the specific forest type, stand density and season. FFMC, DMC and DC calibration curves have been constructed that link FWI values with litter, shallow duff and deep duff moisture contents respectively for various biome characteristics: Lawson and Dalrymple (1996) for DC values and Wotton and Beverly (2007) for FFMC values. However, the DC calibration curves from Lawson and Dalrymple (1996) are limited to forest types that are rare or absent in Québec, and this curve could not be used here. Wotton and Beverly (2007) provide a calibration for a variety of stand densities, forest types as well as seasons (Figure 3.2). From the FFMC calibration curves (Wotton and Beverly, 2007) it is clear that, especially when low FFMC values are present (indicating relatively wet weather conditions: e.g. low temperatures and/or high amounts of precipitation), there are large differences in the observed litter moisture contents. Under equal FFMC conditions, dense mixedwood stands at the end of the fire season (including black spruce and deciduous trees), usually found at the southern edge of the closed-crown forest, have high observed litter moisture contents compared to open-canopy pine stands in the spring. Under dry weather conditions (e.g. a FFMC of 70), the observed litter moisture content varies from 23 % to 56 % in pine stands only as a result of varying stand density. This variability has some implications for the expected fire severity in the various stand types in the boreal forest. A decrease in the mean FFMC (indicating more frequent wet conditions) would have a greater impact on dense mixedwood stands late in the fire season, as these stands will retain litter moisture for a relatively long time. In contrast, open pine and spruce stands during the spring, when understorey remains poorly developed, show only a very slight increase in the litter moisture contents with diminishing FFMC, indicating that these stands can not retain high moisture amounts even under humid conditions. The differing moisture contents between stands under equal FFMC values are probably caused by the influence of the underlying duff layer on the litter moisture contents (Wotton and Beverly, 2007). This implies that under a changing climate regime, the fire severity regime could remain more stable in open pine and spruce forests than in dense mixedwood forests.

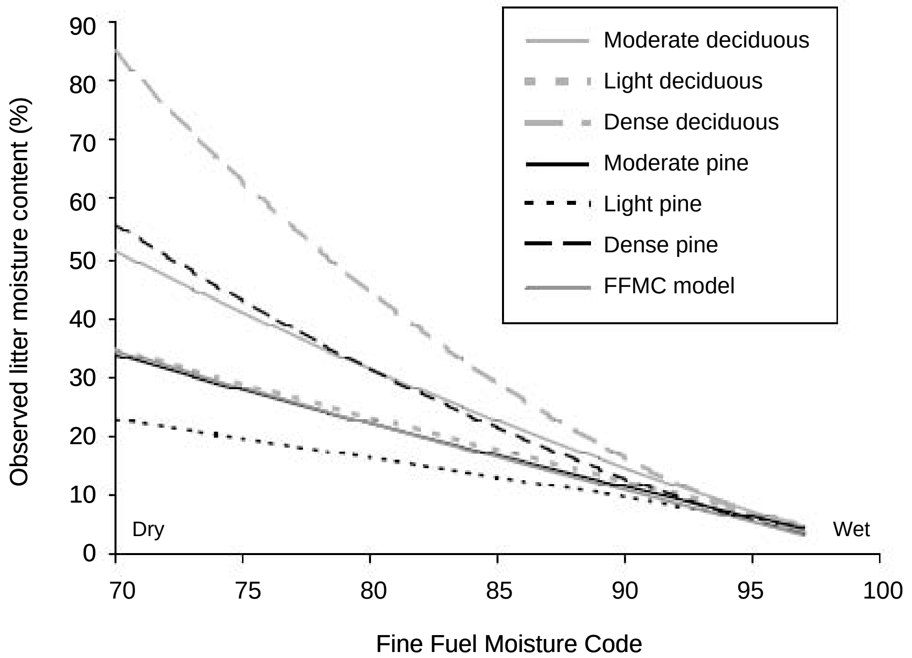

Figure 3.2: FFMC to observed litter moisture content calibration curve for deciduous and pine stands of various density (redrawn from Wotton and Beverly, 2007).

The FFMC in eastern Canada generally diminishes during the fire season, with higher starting values but a larger decrease in the southern boreal forest as compared to the northern part of Québec (Amiro *et al*., 2004; Figure 3.3). Thus, depending on the drying rates of the specific soil types, spring forest floor moisture is generally low but increases during the summer. DMC values fluctuate strongly during the fire season, implying a great sensitivity to weather conditions, however, general highest values, representing low moisture contents, are present in May (southern Québec) and June (northern Québec) (Amiro *et al*, 2004). Finally, the deep duff layer moisture content (represented by the DC) does vary only slightly with weather on the short term (reflected by temperature and precipitation patterns) but has a more or less continuous decreasing trend during the fire season with the lowest moisture contents in August (McAlpine, 1990; Miyanishi and Johnson, 2002; Amiro *et al*., 2004). This indicates an increasing probability of deep burning later in the fire season (Amiro *et al*., 2004). Van Wagner (1987) and Lawson and Dalrymple (1996) reported the importance of the autumn moisture contents on the next spring's DC, depending on winter temperature and precipitation as well as site characteristics.

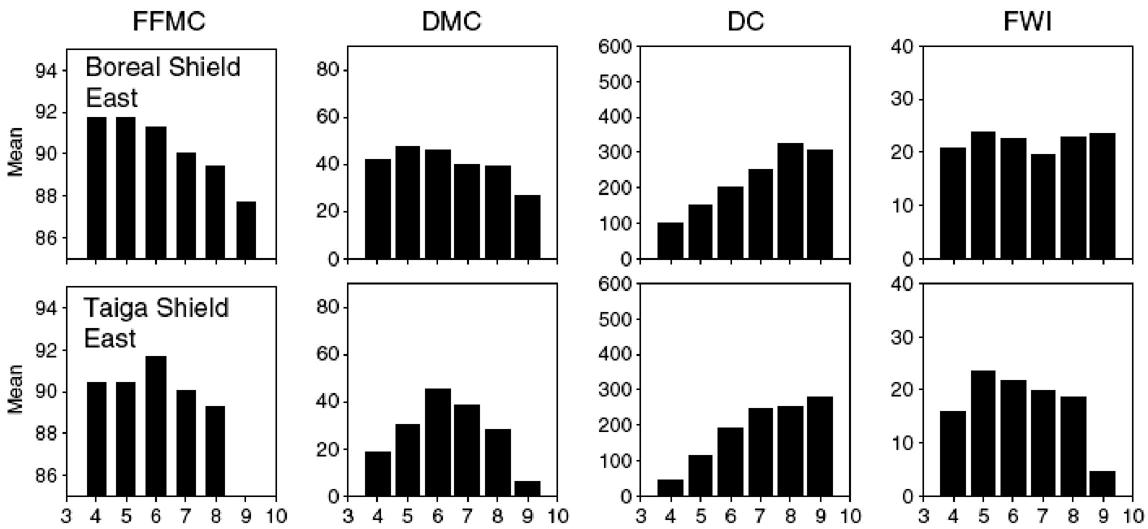

Figure 3.3: Mean monthly values of the FWI components and the FWI (3 = March, 10 = October) in the two eastern Canadian ecozones (redrawn from Amiro *et al*., 2004). For the location of the ecozones, see Figure 2.3.

From the data discussed here, one can conclude that, as the deeper duff layer resembles the general duff characteristics in humid black spruce stands (Miyanishi and Johnson, 2002), these stands are prone to deep smouldering and can release large amounts of C only late in the season. In contrast, shallow duff as found in pine stands is more subjected to precipitation and is less season-dependent as it dries rapidly in the absence of rain. Duff moisture variation is also found at the intra-stand scale: duff generally contains less water if positioned under tree crowns compared to duff in gaps. Miyanishi and Johnson (2002) explained this is as being the result of the interception of precipitation and the lower amounts of dew formed under trees. This is reflected by the findings that duff consumption by fire often shows a patchy distribution, with the largest removal around the tree bases that were alive at the time of fire. However, the moisture content of litter shows a different behaviour: low litter moisture amounts were found in stands with an open canopy, and this effect is stronger under humid conditions (Wotton and Beverly, 2007). The differences between the contrasting relative positions of humid litter and humid duff within a stand may be explained by the origin of the precipitation. Litter moisture is directly recharged by summer precipitation and thus litter moisture benefits from an open canopy, while duff drying is gradual during the summer and may be primarily affected by moisture from snowmelt. This explanation fits with the observations by Miyanishi and Johnson (2002): under trees, snow accumulation is relatively limited (Pomeroy *et al*., 2002) and the underlying duff could therefore receive lower moisture amounts after melting.

These differences in forest floor drying patterns could have important implications for potential duff consumption patterns and high severity fire occurrence. First, pine stands, as well as other type of stands that have shallow forest floors, are expected to have high drying rates of both litter and duff and will therefore be susceptible to deep burning starting early in the fire season when both litter and duff moisture contents are low. The potential for high severity fires is spread relatively uniformly during the fire season as it takes only a couple of days without precipitation to create circumstances prone to high-severity fire. In contrast, black spruce stands, or other stands with thick forest floors and thus a thick duff layer, have duff layers that dry substantially only at the end of the fire season so deep burning is temporally limited to this period. Summer precipitation is likely to be less important than snowmelt for the deep duff moisture contents, however, the drying rates of litter in thick forest floors may be difficult to estimate. The data by Wotton and Beverly (2007) show a slightly higher litter moisture content in black spruce stands compared to equally dense pine stands under equal FFMC and DMC. The differences in depth of burning because of slower drying rates between stands with differing forest floors may be partially obscured by the fact that in the thicker duff layer smouldering can propagate more easily than in the thinner layers.

3.2 Direct forest floor C emission through forest floor burning

The duff moisture content is a major factor in determining potential duff consumption by fire (Van Wagner, 1972); Miyanishi and Johnson (2002) also considered duff thickness and duff bulk density as important variables. The forest floor C content is shown to be strongly correlated to the forest floor bulk density, both increasing with comparable trends with depth (Kasischke *et al*., 2005). This means that as the depth of burn increases, the loss of C to the atmosphere increases nonlinearly. However, potential duff consumption decreases with increases in bulk density due to the limiting effects of a high bulk density on heat transfer during smouldering and the fact that denser duff dries more slowly (Miyanishi and Johnson, 2002). Miyanishi and Johnson (2002) found successful smouldering in sites despite a high duff moisture content due to the corresponding high thickness of the duff layer, which causes a lower convective heat loss. As a result, in two sites with equal relative moisture contents, thick duff layers are potentially burned by smouldering while thin layer smouldering fails.

Most fires in the Canadian boreal forest burn into the canopy, killing the overstorey and understorey, and are therefore identified as crown fires (Wooster and Zhang, 2004). Only the overstorey, understorey and litter are burned by flaming combustion over short periods (Johnson, 1992; Kasischke *et al*., 1995). Crown fires do in almost all cases burn parts of the forest floor as well, although duff consumption occurs predominantly through smouldering which can last for hours to days, primarily because of the high contents of lignin in decomposed duff (Johnson, 1992; Miyanishi, 2001) and less intense aeration (Johnson, 1992). In addition, the high moisture contents of duff, caused by slow drying rates, lower the rate of heating. Two types of duff are distinguished in calculating the FWI (Van Wagner, 1987): loosely compacted moderately decomposed organic matter and deeper, compact, well-decomposed organic matter (Table 2.1). The effect of the differences in bulk density is reflected in the corresponding typical summer drying periods: 15 days for the loosely compacted organic duff and 52 days for the compacted duff at typical summer conditions (temperature 21.1 °C, relative humidity 45 %) as defined by the FWI system (Van Wagner, 1987). The typical drying rates for litter are in the range of 16 hours (Van Wagner, 1987). Duff layers with loosely compacted organic matter are typically found in jack pine stands, while the denser deeper duff that dries more slowly resembles the duff layers in black spruce stands (Miyanishi and Johnson, 2002). Even when the litter is dry enough to be consumed by flaming, the duff may well be too moist, acting as a heat sink and thus preventing local burning (Miyanishi, 2001). However, despite the fact that in the FWI system litter and duff moisture patterns are calculated independently, the upper duff moisture was found to actually influence the litter moisture content to a certain degree (Wotton and Beverly, 2007).

A mean surface fuel consumption (including soil organic matter, coarse woody debris and ground vegetation but excluding crown fuels) of 2.1 kg m^{-2} was reported for the entire Canadian boreal region over a 40-year period (1959-1999) (Amiro *et al.*, 2001), corresponding to ~ 1.1 kg C m^{-2}. As fire C release varies highly both spatially (between and within stands) and temporally depending on stand fuel characteristics, climate conditions and mineral soil type, distinguishing patterns of C release and accumulation across the boreal biomes is extremely difficult.

3.3 Fire severity effects on vegetation re-establishment

Besides regulating C emissions by direct release during a burning event, fire severity determines the thickness of the postfire forest floor and thus the potential postfire re-establishment of vegetation. Generally, organic soils limit postfire vegetation growth, i.e. a higher burn severity increases the potential range of postfire vegetation types within the potential of the site (Johnstone and Chapin, 2006). However, the influence of fire severity on tree recruitment changes with landscape characteristics, including drainage and pre-fire organic layer depth. Because of the important role of burn severity, postfire vegetation and forest floor development are discussed here related to fire severity. The terms high- and low-severity fire are used here merely as a means of comparison.

3.3.1 Forest floor and vegetation re-establishment patterns after a high-severity fire

A high-severity fire is defined here as a fire that consumes most of the forest floor, leaving a thin residual organic matter layer and potentially exposing the mineral soil. Shortly after a high severity fire, site conditions are characterized by high amounts of light, warmer and drier forest floors (if not consumed) and mineral soil and slightly higher amounts of nutrients (Bonan and Shugart, 1989; Lecomte *et al.*, 2005). The exposure of mineral soil provides an additional good basis for rapid tree re-establishment and corresponding closure of the canopy, as the mineral soil provides relatively constant water levels through a high thermal heat capacity and wicking from deeper layers favouring seedling recruitment (Johnstone and Chapin, 2006). Independent of the dominating species, tree biomass production is generally high (Foster, 1985; Lecomte *et al.*, 2006) resulting in even-aged stands (Gower *et al.*, 1997;

Smirnova *et al*., 2008) with high canopy cover and high stand density (Johnstone and Chapin, 2006). The rapid tree growth may be enhanced on sites with a coarse-textured mineral soil that impedes stagnation of water and a corresponding quick accumulation of the forest floor biomass (Lecomte *et al*., 2005). Generally the first regenerating tree species after a high-severity fire is jack pine, at least at sites where this species is present in the pre-fire stand. The serotinous cones of jack pine need intense heat to open and release the seeds, while seeds are tolerant to burning for short periods (Cayford and McRae, 1983). In stands that lack a presence of jack pine, black spruce is likely to establish; in other cases, aspen has the potential to disperse on the remaining mineral from adjacent areas (Bourgeau-Chavez *et al*., 2000) Jack pine starts producing seeds early in its development and seedlings grow fast. However, it is shade intolerant, the seeds need intense heat to be released and its regeneration is strongly negatively affected by thicker forest floors resulting from low-severity fires (Chrosciewicz, 1974; Johnson, 1992). Therefore, jack pine regenerates badly in the absence of fire, even in its own stands (Yu *et al*., 2002). Jack pine stands are generally replaced by more tolerant species in the absence of fire after ~ 100 years (Cayford and McRae, 1983; Smirnova *et al*., 2008), which is a relevant time scale in the eastern Canadian boreal forest: in some parts of the Québec coniferous forests fire cycles equivalent to a 400-year period from 1920 to 2000 (i.e. over a period of 80 years, on average an equivalent of 20 % of the studied area burned) and the general fire frequency showed a decreasing trend since the end of the Little Ice Age (LIA) around 1850 (Bergeron *et al*., 2004), being the result of a reduction in the frequency of drought events (Bergeron and Archambault, 1993).

The accumulation of a forest floor increases with time after fire but at moderate rates while the overstorey develops (Lecomte *et al*., 2006). The forest floor thickness generally increases slowly due to a rapid closure of the canopy and a lower light availability. However, while forest floor thickens, the potential for tree regeneration diminishes, thus inhibiting a re-establishment of the canopy after the postfire tree cohort dies (Taylor *et al*., 1987; Lecomte *et al*., 2006). This is probably the result of a high tree moisture demand together with diminishing water contents in the tree rooting zone due to increases in the forest floor thickness (Taylor *et al*., 1987). In contrast, black spruce stands develop much better in developing forest floors and black spruce presence usually increases relative to other tree species in ageing stands (Lecomte and Bergeron, 2005).

The understorey vegetation, which includes shrubs, herbs as well as moss/lichen, is largely consumed during a high-severity fire. The vascular understorey regeneration is strongly affected by the removal of the forest floor after a high-severity fire as resprouting from underground organs is impeded due to the death of the rhizomes. Generally, bud banks decrease with increasing fire severity (Schimmel and Granström, 1996). As a result, the re-establishment of the understorey after high-severity fires will depend to a greater extent on seeds buried deeply in the soil bank as well as seeds dispersed on the burned surface (Schimmel and Granström, 1996). However, with a further increasing depth of burn, understorey re-establishment will depend uniquely on seeds present by dispersal from invading species (Schimmel and Granström, 1996; Wang and Kemball, 2005). Shortly (up to 3 years) after a high-severity fire, shrubs regenerate very poorly (Johnstone and Chapin, 2006), in contrast to bryophytes (Wang and Kemball, 2005). However, in ageing stands the eventual opening of the canopy enhances an increase in shrub biomass (e.g. *Rhododendron groenlandicum*). Lecomte *et al*. (2005) reported an advantage for evergreen species (*Ericaceae* sp.) in this process, as these are less dependent on the forest floor nutrient stock (Wardle *et al*., 1997). Smirnova *et al*. (2008) found a diminishing understorey biomass in old forest (> 118 years) after the replacement of the postfire stand due to a re-closure of the canopy.

The growth of an understorey assemblage associated with shaded conditions (e.g. *Ceratodon purpureus* (Hedw.) Brid. and *Polytrichum* sp.) is favoured during the first years after fire, with lichens (e.g. *Cladonia* sp.) arriving shortly after (Foster, 1985; Gower *et al*., 1997; Johnstone and Chapin, 2006). While the canopy increases, the lichens are replaced by bryophytes as *Pleurozium schreberi* (Brid.) Mitt. In case the canopy remains open, e.g. resulting from low temperatures limiting tree regeneration as is common in the lichen woodland, lichen may continue to dominate. However, lichen growth may be favoured on well-drained sites directly after fire even before tree establishment (Payette, 1992; Payette *et al*., 2000). On a centennial scale, an assemblage preferring wet conditions (e.g. *Sphagnum* sp.) can establish while canopy opens after death of the postfire tree cohort (Taylor *et al*., 1987; Lecomte *et al*., 2005).

3.3.2 Forest floor and vegetation re-establishment patterns after a low-severity fire

A low-severity fire is defined here as any fire that leaves part of the forest floor intact, while still potentially killing trees. The conservation of a part of the forest floor leaves constraints on the development of the vegetation and the forest floor and thus vegetation regeneration patterns differ from regeneration after severe burning.

The incomplete removal of the forest floor leaves a dark, porous layer which creates a recruitment surface with high variations in forest floor temperature and moisture contents that limit seed germination compared to mineral soil seedbeds (Duchesne and Sirois, 1995; Charron and Greene, 2002). In addition, forest floors that remain after a low-severity fire will start accumulating organic matter at higher rates than when the forest floor is consumed completely (Lecomte *et al*., 2006). This is probably the result of a reduced decomposition rate by an earlier occupation by bryophytes trough lowering the forest floor temperature and reducing litter quality (Zoltai *et al*., 1998; Yu *et al*., 2002). The eventual establishment of *Sphagnum* sp. is mentioned as being a key factor in controlling the thickness of the forest floor (Fenton *et al*., 2005) due to the capability of maintaining low pH and cold conditions thereby further reducing decomposition rates (Oechel and Van Cleve, 1986; Payette and Rochefort, 2001).

Because of the rapidly developing forest floor after a low-severity fire and delayed seedling recruitment in thicker forest floors (Johnstone and Chapin, 2006), tree biomass accumulates at much lower rates. In addition, a multi-aged stand usually develops with an open canopy. The presence of a postfire forest floor influences to a large extent the species that will occupy the site on short as well as long time scales (Johnson, 1992; Schimmel and Granström, 1996), the rate of growth (Lecomte *et al*., 2006) and thus C contents of the ecosystem as a whole. After low-severity fires that leave a layer of burned organic matter, black spruce is more probable to establish than jack pine (Chrosciewicz, 1974; Duchesne and Sirois, 1995; Lecomte *et al*., 2006). The growth of deciduous trees on low-severity burned sites is very limited because of their inability to establish on organic soils (Johnstone and Chapin, 2006). The general presence of feather moss (e.g. *Pleurozium schreberi*) after fire limits the possibilities of tree recruitment, especially when compared to mineral soils and lichen seedbeds (Payette *et al*., 2000). During long periods of fire absence, black spruce predominantly regenerates by layering (Morneau and Payette, 1989).

Shallow burns are advantageous to an understorey vegetation that possesses a forest floor propagule bank (Figure 3.4), e.g. *Rhododendron groenlandicum* as well as *Sphagnum* sp., as these types of vegetation have the ability to resprout from tissues buried in the deep duff layer (Clymo and Duckett, 1986).

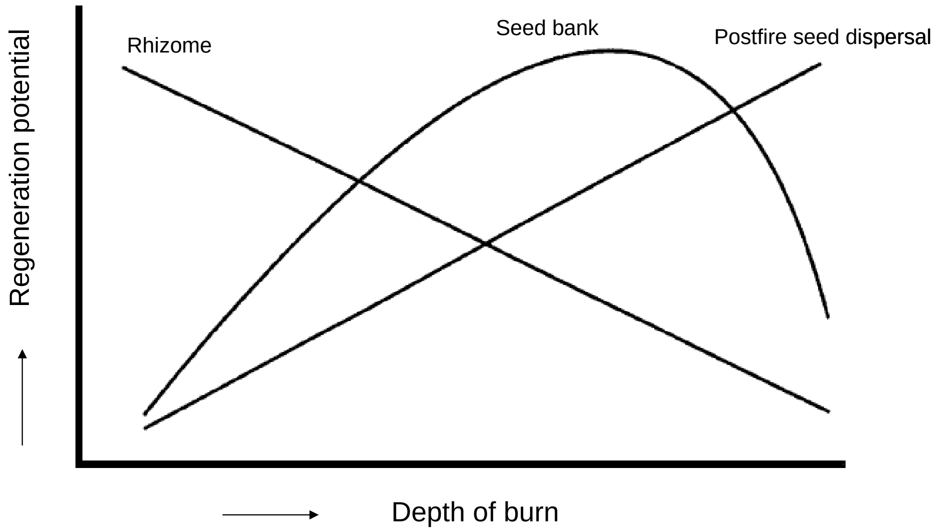

Figure 3.4: A schematic model of the relationship between depth of burn and regeneration potential for three reproduction strategies (redrawn from Schimmel and Granström, 1996).

4 Climate change scenarios and fire severity ratings for Québec

4.1 Québec climate change scenarios for 2050 - 2075

The data from the Canadian Atmospheric Environment Service's General Circulation Model (GCM) is the major source of information for climate and fire regime scenarios in Canada for the 21st century (Bergeron and Flannigan, 1995; Stocks *et al*., 1998; Flannigan *et al*., 2001; Flannigan *et al*., 2005), partially due to its use of daily data. In areas with no large variations in altitude, including Québec, GCMs provide confident data, whereas the use of Regional Climate Models (RCM) is preferable in mountainous areas (Flannigan and Wotton, 2001). Recent studies use RCMs; e.g. the Canadian Regional Climate Model (Sushama *et al*., 2006; Plummer *et al*., 2006). I compared the climate scenarios for the mid-21st century period used by Flannigan *et al*. (1998; 2001) to data from to other studies, being Price *et al*. (2001), Plummer *et al*. (2006) and IPCC (2007). The doubled atmospheric CO_2 concentration (~ 660 ppm) could represent either the 2040 – 2060 period (Flannigan *et al*., 2001) or the situation around 2075 (IPCC, 2007). Temperature and precipitation values were calculated for the fire season only, running from May 1 to August 31 in Québec, which comprises around 99 % of the area burned (Harrington, 1982).

GCMs and RCMs show an expected increase in mean daily maximum temperature during the fire season in south-western Québec of around 4.0 °C while a 3.0 °C increase is expected for northern and eastern Québec and Labrador (Flannigan *et al*., 2001). This study did not provide quantitative data on precipitation changes, but Flannigan *et al*. (1998) found an increase in the 2050 - 2075 fire season precipitation of about 10 % at the border between Québec and Ontario and around the 55 °N parallel to no increase in south-eastern Québec. No differences in wind speed were found between the two periods (Bergeron and Flannigan, 1995).

Attention should be paid to the fact that a percentual increase is by definition relative to the reference value, which is, in case of the northern Québec precipitation, very low. This means the absolute increase in precipitation for this region will be less dramatic (Price *et al*., 2001). Corresponding with the data used by Bergeron and Flannigan (1995), Price *et al*. (2001) found the changes in monthly mean summer wind velocity will be insignificant, except for a small region in north-central Québec (increase of 0 to 10 %), however, the authors have some reservations on the credibility of these data.

The more recent studies (Plummer *et al.*, 2006; IPCC, 2007) show the same tendencies as Flannigan *et al.* (1998; 2001) (Tables 4.1 and 4.2), thereby reinforcing the credibility of the FWI scenarios discussed below. The sole major difference between the climate scenarios concerns the amount of winter precipitation: IPCC (2007) expect an increase of 26 to 38 % for 2080 - 2099 while Price *et al.* (2001) and Plummer *et al.* (2006) announced increases varying from 1 to 25 % over the ensemble of the Québec biomes.

	Period of change	Change in FWI (%)			Change in mean summer temperature (°C)			Change in mean winter temperature (°C)		
		50°N 74°W	53°N 74°W	56°N 74°W	50°N 74°W	53°N 74°W	56°N 74°W	50°N 74°W	53°N 74°W	56°N 74°W
Flannigan et al. (2001) CGCM	1960 - 1980 to 2040 - 2060	- 8	- 12	- 18	+ 3.8	+ 3.2	+ 3.0	-	-	-
Price et al. (2001) CGCM	1961 - 1990 to 2041 - 2070	-	-	-	+ 2.5	+ 3.2	+ 3.2	+ 3.1	+ 4.0	+ 6.0
Plummer et al. (2006) CRCM	1971 - 1990 to 2041 - 2060	-	-	-	+ 2.4	+ 2.6	+ 2.7	+ 2.5	+ 4.0	+ 5.5
IPCC (2007) AOGCM	1980 - 1999 to 2080 - 2099	-	-	-	+ 3.5	+ 3.3	+ 3.2	+ 6.0	+ 6.8	+ 7.5

Table 4.1: FWI and mean temperature change at various latitudes according to different studies.

	Period of change	Change in FWI (%)			Change in mean summer precipitation (%)			Change in mean winter precipitation (%)		
		50°N 74°W	53°N 74°W	56°N 74°W	50°N 74°W	53°N 74°W	56°N 74°W	50°N 74°W	53°N 74°W	56°N 74°W
Flannigan et al. (1998) CGCM	1960 - 1980 to 2040 - 2060	- 8	- 12	- 18	+ 3	+ 9	+ 15	-	-	-
Price et al. (2001) CGCM	1961 - 1990 to 2041 - 2070	-	-	-	+ 2	+ 13	+ 15	+ 1	+ 13	+ 12
Plummer et al. (2006) CRCM	1971 - 1990 to 2041 - 2060	-	-	-	+ 7	+ 10	+ 12	+ 4	+ 15	+ 25
IPCC (2007) AOGCM	1980 - 1999 to 2080 - 2099	-	-	-	+ 2	+ 7	+ 10	+ 26	+ 32	+ 38

Table 4.2: FWI and mean precipitation change at various latitudes according to different studies.

4.2 Québec FWI scenarios for 2050 - 2075

Flannigan *et al*. (1998; 2001) calculated the ~ 2050 FWI values based on daily weather outputs from the Canadian GCM. From the comparison of different climate scenarios we conclude that the data used by Flannigan *et al*. (1998; 2001) to calculate the future (2050 - 2075) FWI values should be reliable, especially in the northern boreal region of Québec. Therefore, the FWI values as calculated by Flannigan *et al*. (1998; 2001) are used here to estimate the climate effects on future fire severity.

The FWI was expressed as a ratio of the 2 × CO_2 and 1 × CO_2 FWI values, with values larger than 1 indicating an increase and values lower than 1 a decrease in FWI. Lower FWI values indicating less severe fire weather are expected in Québec for 2050 - 2075, resulting from an apparent dominance in the influence of precipitation increase as compared to the expected increase in temperature (Flannigan *et al*., 1998; 2001). Québec and a part of eastern Ontario show expected FWI values of 0.8 to 1.0, whereas the central and western parts of Canada generally show values higher than 1.0, indicating an expected increase in fire weather severity (Figure 4.1).

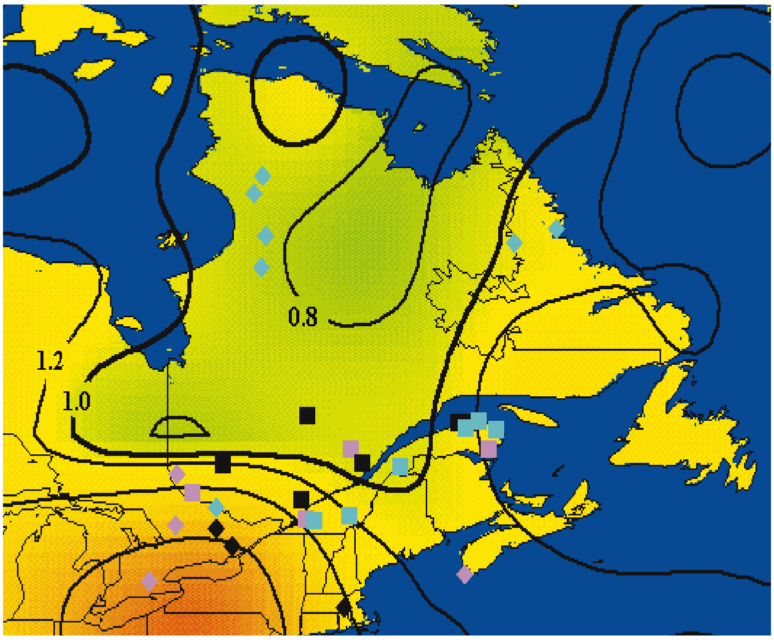

Figure 4.1: The 2 × CO_2 / 1 × CO_2 FWI in eastern Canada showing values < 1 in central and northern Québec (redrawn from Flannigan *et al*., 2001).

The FWI trend in Québec corresponds to the fire history of the recent past, when fire frequency decreased while temperatures gradually rose since the end of the LIA (~ 1850), independent of human fire suppression (Bergeron, 1991; Girard *et al*., 2008) and supposedly driven by a reduced drought frequency (Bégin and Payette, 1988; Archambault and Bergeron, 1992; Bergeron and Archambault, 1993; Girardin *et al*., 2006). Despite the fact that Bégin and Payette (1988) observed an increase in post-LIA winter precipitation as opposed to summer precipitation, the snowfall in winter may well partially influence the DC (Lawson and Dalrymple, 1996; Miyanishi and Johnson, 2002) and the length of the fire season (Wotton and Flannigan, 1993).

5 The effect of changes in fire severity on Québec boreal biome dynamics and C contents

The eastern Canadian boreal forest consists of a southern zone of closed-crown forest, a central zone with open-crown forest or lichen woodland and a northern forest-tundra zone (Rowe, 1972; Payette, 1992). Latitudinal differences in the boreal forest types are the result of zone-typical postfire response, with the various fire regimes initially driven by climate conditions (Payette, 1992).

In a changing climate, spatial and temporal temperature and precipitation patterns are changing, as well as the occurrence and severity of fire. Thus, a spatial change in vegetation patterns is likely; however, it is important to evaluate to what extent different vegetation patterns can exist under the influence of changing climate circumstances. Jasinski and Payette (2005) concluded that vegetation patterns are primarily determined by the disturbance history, and that edaphic conditions do not play a role, as different forest types can exist under the same climatic and environmental conditions. This is an important point, which is supported by the findings that changing vegetation zones on a small scale did indeed not seem to be limited to specific edaphic conditions (Payette *et al*., 2000).

The effect of a changing climate will most probably cause a latitudinal shift in biome dispersion in correspondence with changing fire regimes. In identifying the effect of a lower FWI on fire severity, it is critical to estimate the change in forest type, as fire severity is dependent on the forest-type specific forest floor moisture contents. In other words, the specific moisture codes may be substantially lower (implicating more humid forest floor layers) in the changing climate of 2050 - 2075, but if the importance of each forest component (e.g. overstorey, understorey and forest floor) changes, litter moisture contents may well become lower, thereby increasing local potential fire severity (Wotton and Beverly, 2007). However, as previously stated, a change in forest structure can not be predicted without taking into account future fire severity.

The problem is shown more in detail by an hypothetical example, where fire severity and climate are viewed as two variables that are linked only to the FFMC, the moisture code for the litter layer. In the hypothetical case a substantial warming is announced without any change in precipitation, the FFMC values will generally increase as the weather conditions will dry out the forest floor. Increasing FFMC values generally indicate a higher potential (expressed either as intensity or frequency) for litter burning. At the same time, the increase in temperature causes an increase in tree regeneration potential or tree production (Bergeron, 1998), which corresponds to the large-scale boreal forest tree production pattern on the latitudinal gradient if fire regime effects are kept out of consideration. As a result, tree stand density increases. From this point, two effects are possible (Figure 5.1):

1. If the increase in tree density due to the warming is more important than the increase in FFMC, the observed litter moisture contents will increase, causing a decrease in potential fire severity;
2. If the increase in FFMC is more important than the increase in tree density due to the warming, the observed litter moisture contents will decrease, causing an increase in potential fire severity.

In situation 1, the decrease in fire severity will have its own effect on tree density. One could hypothesize that the decrease in fire severity on the long term will lead to thicker forest floors and an opening of the forest canopy, which would eventually result in a decrease in observed litter moisture contents if climate (and thus the mean FFMC) remains stable. In situation 2, an increase in fire severity could result in increased tree productivity with closed canopies. While climate and FFMC remain stable, the litter layer would become more humid. Situation 1 could be typical for an ecosystem where tree growth is currently strongly limited by temperature, as in the northern forest-tundra (Payette, 1992; Lloyd *et al*., 2005). Situation 2 may be found in the southern boreal forest, where tree reproduction is already high and canopy closure is close to being complete.

These examples show that on a large scale, a change in climate and FWI can eventually lead to a typical fire severity status with a stable vegetation pattern, but the pathways and stable configuration may vary per biome.

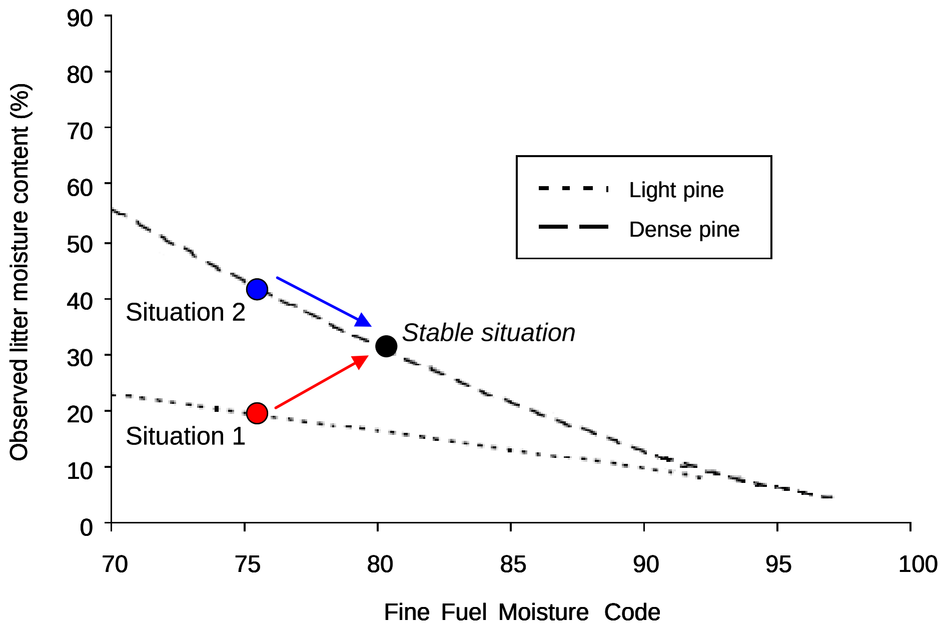

Figure 5.1: FFMC – Observed litter moisture calibration diagram for dense and light pine stands showing the potential pathways resulting from a shift to a warmer climate. Situation 1 represents a shift in stand density resulting from an enhanced tree growth, while the stand in situation 2 has a stable vegetation density.

5.1 Vegetation and C in the southern closed-crown forest

The southern closed-crown forest in Québec is dominated by black spruce – balsam fir stands with co-dominance of white spruce (*Picea glauca* (Moench) Voss.) and paper birch (*Betula papyrifera* Marsh.). The southern edge, located around 49 °N is characterized by balsam fir and paper birch stands and is part of the mixedwood forest. At the transition to the northern closed-crown forest, there is an increasing forest floor moss presence (e.g. *Pleurozium schreberi*) (Bergeron *et al.*, 2004). Large differences in the actual (1920-1999) average fire cycle are reported for this biome, varying from 398 years (Bergeron *et al.*, 2004) to 191 and 521 years in west and central Québec (Bergeron *et al.*, 2001). The mean annual temperature is roughly between 0 and -1 °C and the mean annual precipitation varies between 850 and 1100 mm (Payette and Rochefort, 2001). The length of the growing season is around 155 days (Gerardin, 1980). This region extends roughly from the Québec-Ontario border near Rouyn-Noranda to the north shore of the Gulf of Saint Lawrence between 49 and 50 °N (Figure 5.2).

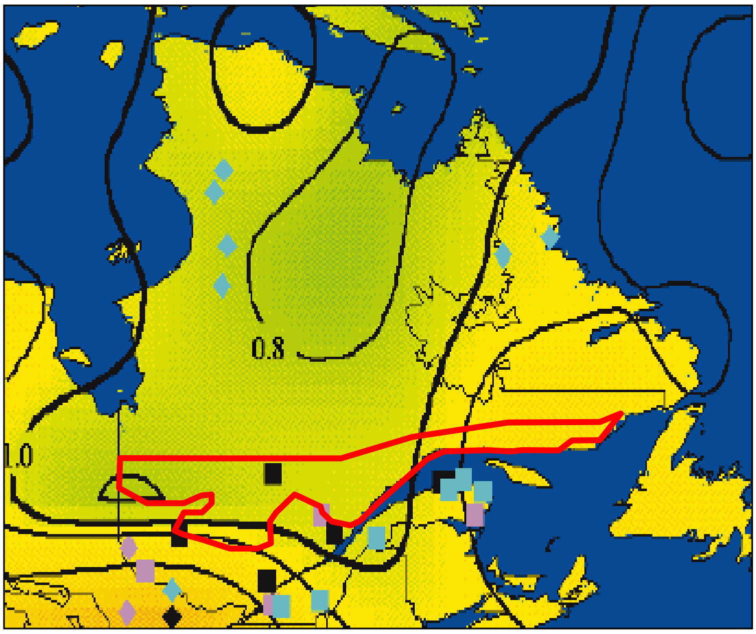

Figure 5.2: Approximate position of the southern closed-crown forest (in red) projected on the 2050 FWI map (Flannigan *et al*., 2001)

The actual location of the transition of the southern closed-crown forest and the mixedwood forest in the south is at least partially determined by factors that will remain stable under changing climate regimes. The presence of lakes and a pronounced topography that influence potential fire size and the abundance of deciduous species that lowers potential fire severity are positively correlated with balsam fir presence in western Québec (Bergeron *et al*., 2004). In the closed-crown forest, where black spruce dominates, balsam fir presence is limited because of generally lower temperatures and a frequent presence of humid and thick forest floors (Harper *et al*., 2003; Messaoud *et al*., 2007). This is acknowledged by Johnstone and Chapin (2006) who found that moist conifer forests have strong reductions in seed germination and seedling establishment in thick postfire forest floors, being the result of a low-severity fire, while lichen woodland species (including spruce) establish more easily on these post-low-severity fire forest floors. They also stated that deciduous species are clearly disadvantaged on organic soils, especially when already absent in the pre-fire stand, because the reproduction from seed will be strongly limited. However, balsam fir stands can establish where trembling aspen and paper birch dominate the postfire stand (Messaoud *et al*., 2007), e.g. after a high-severity fire that removes all or most of the organic soil (Bourgeau-Chavez *et al*., 2000) or on rock outcrops with shallow tills (Bergeron, 1998). On the stand scale, the closed black spruce – balsam fir stands have relatively high amounts of forest floor moisture (Bourgeau-Chavez *et al*., 2000). High amounts of deep duff moisture in spring cause a low potential for severe burning, but in the late summer potential fire severity increases due to continuous drying of the deep duff.

5.1.1 Changes in climate, fire regime and C stocks in the southern closed-crown forest

The expected FWI values for 2050 - 2075 show values that are just below 1, indicating a slight decrease in fire spreading intensity (Flannigan *et al*., 2001; Figure 5.2).

Given the fact that in the actual situation balsam fir can not establish in the closed-crown forest due to a disadvantage in establishing on thick forest floors, it is not likely that this species will move northward due to an increase in the average stand age and thus the average forest floor thickness. As a result, local paludification processes may become more important. The status of the canopy, however, may show different tendencies. First, the reduced opening of the canopy can occur due to a general ageing of the stands. However, higher summer temperatures can increase production rates (Bergeron, 1998) and may cause a general closure of the overstorey. A reduced opening of the canopy may further cause an increase in understorey vegetation; however, if paludification is present, this understorey may be composed of relatively homogenous *Ericaceae* sp. (Taylor *et al*., 1987). If the canopy remains closed, understorey development may be limited and a forest floor cover of feather moss is favoured.

Due to the closed canopy, the litter layer of the southern closed-crown forest can remain humid for a long time even when FFMC values are relatively high. In addition, a decrease in mean FFMC around 2050 - 2075 would imply a disproportional increase in observed litter moisture contents (Wotton and Beverly, 2007). The lower amounts of winter precipitation and the higher winter, spring and summer temperatures will cause a general earlier start of the fire season. In the southern boreal region, the litter moisture contents are low in spring but increase clearly during the fire season (Amiro *et al*., 2004). In the 2050 – 2075 situation, the litter moisture contents could increase resulting from an increase in summer precipitation, but a 2 to 5 % increase in summer precipitation may be insufficient to compensate the 2.0 to 3.5 °C increase in temperature. In addition, Price *et al*. (2001) and IPCC (2007) announced a lower increase in both temperature and precipitation for 2050 - 2075 compared to Flannigan *et al*. (2001), causing an additional source of uncertainty. The duff layer is generally humid in

May and dry in August. In drier sites the deep duff moisture contents may be negatively influenced by the lower amounts of winter precipitation and higher winter temperatures that determine the spring DC values (Lawson and Dalrymple, 1996). Here, the expected decrease in winter precipitation may cause a higher potential burn severity earlier in the fire season, after snow melts but before the understorey develops. These contrasting patterns of the increase in litter moisture and a decrease in duff moisture contents are confusing and make it difficult to estimate contemporary fire severity regimes, also because of the uncertainty in the development of the canopy.

I estimate that the occurrence of fire may remain low, as the litter may be too humid to sustain frequent burning and the relatively closed canopies will keep the forest floor from rapid drying. However, if both litter and duff dry substantially, which may still occur during summer in the future climate regime, fires may become severe. The increase in summer precipitation that will keep litter humid may induce paludification in parts of the closed-crown forest and may consecutively cause a further local increase in forest floor humidity; the extent of paludification depending on the importance of the changes in temperature and precipitation. A shift of the southern closed-crown forest under a diminishing fire frequency regime (Flannigan *et al*., 2001) towards a slightly more open black spruce forest with thicker forest floors that may well show a frequent occurrence of paludification processes will turn these boreal biomes into potentially larger C sinks. However, as winter precipitation is expected to remain relatively stable, the duff layer will dry earlier and to a greater extent, leaving a greater potential for deep burning with a largest change early in the season. Thus, total closed-crown forest C contents may increase until 2050 - 2075 due to thicker forest floors and local paludification under a less frequent burning, but this process is less probable to result from an increase in the occurrence of low-severity fires.

5.2 Vegetation and C in the northern closed-crown forest

The northern closed-crown forest has a vegetation of black spruce with a forest floor cover of feather moss (e.g. *Pleurozium schreberi*). Despite the term closed-crown, tree cover varies between 40 and 80 % in mature stands. Understorey vegetation is present in the more open parts, and contains *Rhododendron groenlandicum*, *Vaccinium angustifolium* Ait. and *Kalmia polifolia* Wang. (King, 1987). The mean annual temperature is between 0 and -2.5 °C and the mean annual precipitation varies from 700 to 1000 mm with the higher values in the east of Québec (Payette and Rochefort, 2001). The growing season length is about 145 days (Gerardin, 1980). The northern part of the closed crown forest occupies the region between 50 and 52 °N from the southern part of the James Bay region until the eastern Labrador coast (Figure 5.3). The northern closed-crown forest is characterized by a patchy presence of lichen woodland vegetation, predominantly on well-drained sites (Payette, 1992; Girard *et al.*, 2008; discussed below).

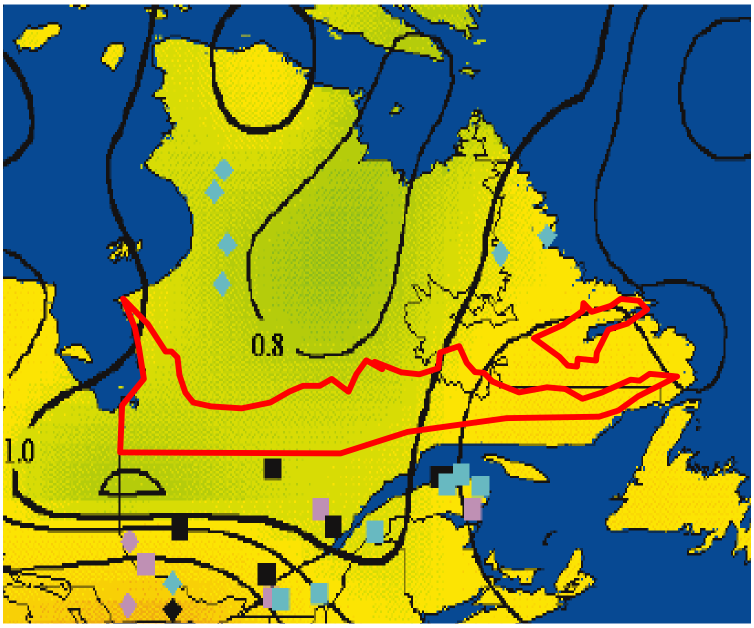

Figure 5.3: Approximate position of the northern closed-crown forest (in red) projected on the 2050 FWI map (Flannigan *et al.*, 2001)

5.2.1 Changes in climate, fire regime and C stocks in the northern closed-crown forest

The models used by Flannigan *et al*. (2001) show 2050 - 2075 FWI values for the northern closed-crown forest of close to 0.95 in western Québec to values attaining 0.85 in central Québec, indicating less intense fire weather.

The openness of the canopy relative to the southern closed-crown forest causes a different reaction of the forest floor moisture contents on a decrease in mean FFMC (Wotton and Beverly, 2007). As the forest floor drying rates are higher under the open canopy, a shift to wetter summer weather conditions will cause a less pronounced increase in litter moisture as compared to the southern closed-crown forest. The precipitation patterns expected for 2050 -

2075 differ substantially between the southern and the northern closed-crown forest, i.e. in the northern biome large increases in precipitation were announced, while the southern biome precipitation patterns show a slight increase (summer) or are not clearly defined (winter). Thus, the expected increase in summer precipitation relative to the present is much higher in the northern part of the closed-crown forest, which may compensate the lower moisture retention capability of the forest floor.

Due to the increase in both summer and winter precipitation, deep burning fires could become less important, which contrasts with the southern closed-crown forest. In addition, fire frequency was announced to diminish (Flannigan *et al*., 2001). A lower general fire severity should be advantageous to species that regenerate from sprouting of underground parts as *Vaccinium angustifolium* and *Rhododendron groenlandicum* (Viereck, 1983; Schimmel and Granström, 1996). The regeneration of species that depend on postfire seed dispersal will be clearly disadvantaged as compared to especially rhizomatous species, thus one could expect a relative decrease in birch species. More generally, tree establishment, probably dominated by black spruce, will be more limited due to a thickening of the forest floor and will thus result in more open stands that enhance the opportunities for a rapid establishment of *Pleurozium schreberi*. In addition, *Sphagnum* sp. may be able to recolonize the site, especially where tissues remain intact present in the deep duff, but may remain absent initially at sites that contain a well-developed shrub layer.

The total ecosystem C stock may increase in the northern closed-crown stands due to a more pronounced development of the forest floor, especially if *Sphagnum* sp. are able to establish and forest floors accumulate rapidly.

5.3 Vegetation and C in the lichen woodland

The lichen woodland zone is found roughly between 52 and 55 °N (Figure 5.4). Lichen woodland vegetation in Québec has a dominance of black spruce, whereas lichen (e.g. *Cladina* and *Cladonia* sp.) dominate the forest floor cover (Payette, 1992). The tree cover ranges from 40 % in the south to 5 % at the transition to the forest-tundra with trees several meters apart; the growing season covers between 110 and 125 days (Gerardin, 1980). The mean annual temperature varies between -2.5 and -5.0 °C and the mean annual precipitation between 650 and 900 mm (Payette and Rochefort, 2001). The lichen woodland is the biome in Québec that is most exposed to frequent burning with an actual fire rotation period of ~ 100 years (Payette *et al*., 1989). Due to the openness of the canopy shrubs are common, with *Rhododendron groenlandicum*, *Betula glandulosa* Michx., *Kalmia angustifolia* L., *Vaccinium angustifolium* and *Vaccinium myrtilloides* Michx. as common shrubs (King, 1987; Payette *et al*., 2000). Lichen (e.g. *Cladonia stellaris* (Opiz.) Pouz. et Vezda) generally cover the ground in stands exceeding 60 years of age, while mosses (e.g. *Pleurozium schreberi*) can be found under trees and in more recent burns (King, 1987). In the southern part of the lichen-woodland, forming the transition to the closed-crown forest, more dense black spruce stands are present, with a preference for poorly drained soils (Girard *et al*., 2008). In contrast, in the northern part where longer fire cycles exist, black spruce populations remain marginal due to less frequent episodes of postfire seedling establishment and the gradual dependence on layering for reproduction.

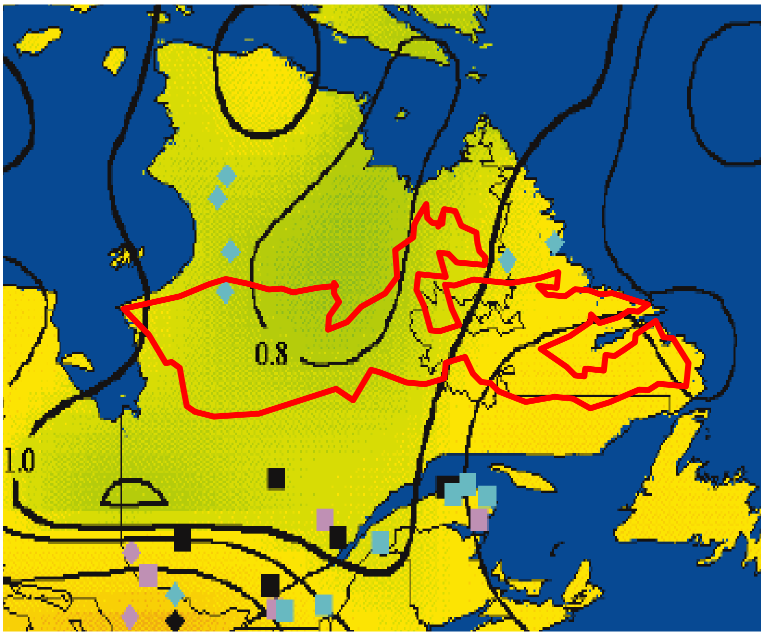

Figure 5.4: Approximate position of the lichen woodland forest (in red) projected on the 2050 FWI map (Flannigan *et al*., 2001)

There has been much debate on the status of the lichen woodland zone in the postfire regeneration process (e.g. Maikawa and Kershaw, 1976; Payette and Morneau, 1993). Maikawa and Kershaw (1976) supported the theory that in the absence of fire, the lichen-woodland would close and the lichen cover would be replaced by moss. However, Payette (1992) and Payette *et al*. (2000) argue that the lichen woodland is maintained in the southern and central part of the boreal forest by a combination of climate conditions and dry and mesic soils favouring high fire frequencies, while Payette and Morneau (1993) stated that the lichen woodland can be regarded as “a stable ecosystem able to maintain its structure for thousands of years at exceptionally undisturbed treeline sites”. Jasinski and Payette (2005) do not mention the influence of soil and hydrology, but acknowledge the higher fire frequencies typical in sites with a microclimate due to local orographic variations combined with frequent spruce budworm (*Choristoneura fumiferana* Clem.) outbreaks, causing stable lichen woodland patches in closed crown forests. After a high-severity fire, lichen woodland can establish if balsam fir is impeded to return by seeding from the unburned edges (Jasinski and Payette, 2005). In addition, lichen will be able to colonize a site when pre-fire shrub populations can be eliminated due to the severe burning of the rhizomes, thus impeding shrub growth from sprouts (Rowe, 1983; Viereck, 1983; Payette, 1992; Schimmel and Granström, 1996). In estimating the effect of a decrease in fire severity and frequency on the lichen woodland biome, it is important to regard the possible pathways of lichen woodland development in the long term in the absence of fire.

5.3.1 Changes in climate, fire regime and C stocks in the lichen woodland

The lichen woodland FWI values for the period around 2050 – 2075 are 1.0 in western Québec at the Hudson Bay and the northern parts of the James Bay coasts, as well as the lichen woodlands at the Québec-Labrador border. The central zone, however, shows values lower than 0.8, thus indicating a large decrease in potential fire intensity (Flannigan *et al*., 2001).

The open nature of the lichen woodland with its lichen ground cover is prone to deeper burning as compared to the closed-crown forest. The litter dries rapidly in absence of humid weather conditions and a more humid climate will therefore cause only a minor increase in litter moisture contents (Wotton and Beverly, 2007). This means that mean potential fire severity is expected to decrease only slightly. In contrast to the closed-crown forest, the forest floor moisture contents show less variation during the season and slightly higher values for the duff layer (Amiro *et al*, 2004). The increase in precipitation will positively influence both the litter and the duff moisture contents, and thus most probably cause a diminishing occurrence of severe burning as well as a diminishing fire frequency, with a possible exception of some of the western and eastern regions (Flannigan *et al*., 2001).

A regime of less severe burning may turn parts of the lichen woodland into other boreal forest types, especially those parts that are actually in proximity of the present closed-crown forest and the forest-tundra. The incomplete burning of the forest floor is advantageous for the understorey vegetation that regenerate by sprouting from the deeper forest floor. As the origin of the lichen woodland is at least partially explained by “the elimination or substantial depletion of the shrub populations during catastrophic fires” (Payette, 1992), the expected decrease in severe fires may cause a shift to a denser shrub layer and a thicker forest floor that are occupied by shade-tolerant moss instead of lichen. If moss indeed dominates, a second period of black spruce seedling establishment will be restricted as the moss cover may create a barrier for seed germination (Morneau and Payette, 1989). However, the black spruce cover may well remain stable due to enhanced layering as a regenerative process as opposed to seedling establishment and higher mean summer temperatures will increase the growth potential. As a result, parts of the lichen woodland may start resembling the northern closed-crown forest. The relatively wet forest floors will remain dominated by black spruce, however, in mesic sites, balsam fir may become dominant while sites presently dominated by jack pine would become less abundant. The combination of an increase in summer precipitation and a milder climate may thus cause a shift from lichen woodland to a biome more resembling the black spruce – *Pleurozium schreberi* stands in the closed crown forest. This same shift has already been documented in the more humid and milder oceanic regions of Labrador (Foster, 1985). In this study, lichen woodlands were replaced by spruce – *P. schreberi* forest in a region with a mean annual temperature of 0 °C and a mean annual precipitation of 1000 to 1100 mm (Foster, 1985; Payette and Rochefort, 2001). This development could form a model for what can be expected for parts of the lichen woodlands in the more continental regions of Québec. At the Hudson Bay coast and at the Québec-Labrador border where FWI values will remain stable, a different development may prevail. Here, the local increase in precipitation may not be able to compensate for the increase in temperature, causing a stable fire intensity regime. Therefore, the lichen woodland may remain stable in these regions.

The higher temperatures in both summer and winter may cause a longer growing season, however, the 10 to 20 % increase in winter precipitation that will accumulate as snow may compensate for an earlier start of potential vegetation growth circumstances.

I conclude that a slight decrease in fire severity may turn some parts of the lichen woodland in a black spruce – *Pleurozium schreberi* forest. However, the changes may be of minor importance as compared to the closed-crown boreal forest, because the actual lichen forest floor is not as sensitive to changes in summer precipitation, causing a relatively stable fire severity regime. In addition, the regional variation in the expected FWI for 2050 - 2075 is great, indicating possibly differing patterns in western and eastern as opposed to central Québec. The lichen woodland C stock will increase in areas with less severe burning where *P. schreberi* growth enhances forest floor development. Tree C may remain relatively constant due to the balance of lower tree seedling establishment and the increase in production due to the increase in mean summer temperature.

5.4 Vegetation and C in the forest-tundra

The forest-tundra shows a northward decrease in black spruce canopy cover until treeless parts in the north. Locally, white spruce and eastern larch (*Larix laricina* (Du Roi) K. Koch) can be dominant (King, 1987). The forest-tundra biome is further characterized by a mosaic structure of lichen – spruce and lichen – *Ericaceae* sp. communities (Sirois and Payette, 1991). Lichen, *Rhododendron groenlandicum* and *Betula glandulosa* are present when black spruce is absent. *Alnus crispa* (Ait.) Pursh. regularly forms the understorey in small white spruce and eastern larch stands, while *P. schreberi* can still be present on the forest floor under a small tree canopy (King, 1987). The northern limit of the forest tundra is defined as the limit of trees with a tree growth form. From the southern to the northern limit of the forest-tundra zone (Figure 5.5), there is a diminishing capability of trees to recover after fire disturbance, because of lower seedling establishment causing a limited success of tree regeneration (Payette, 1992) which results in a shrub coverage that increases with latitude. In the shrub subzone trees are present but grow predominantly as krummholz, defined as small, stunted trees. Because of the severe climate, seedling establishment is rare during fire-free periods. Like in the lichen woodland, black spruce layering is a common means of regeneration in the absence of fire (Payette *et al*., 1985). The mean annual temperature is roughly between -5 and -7 °C and the mean annual precipitation varies from 400 to 600 mm (Payette and Rochefort, 2001). Permafrost is sporadic (< 2 %) around 55 °N and more general (< 50 %) around 57 °N (Allard and Séguin, 1987). The actual fire cycles are around 180 years at the southern forest tundra and increase to 1460 years in the northern forest tundra (Payette *et al*., 1989).

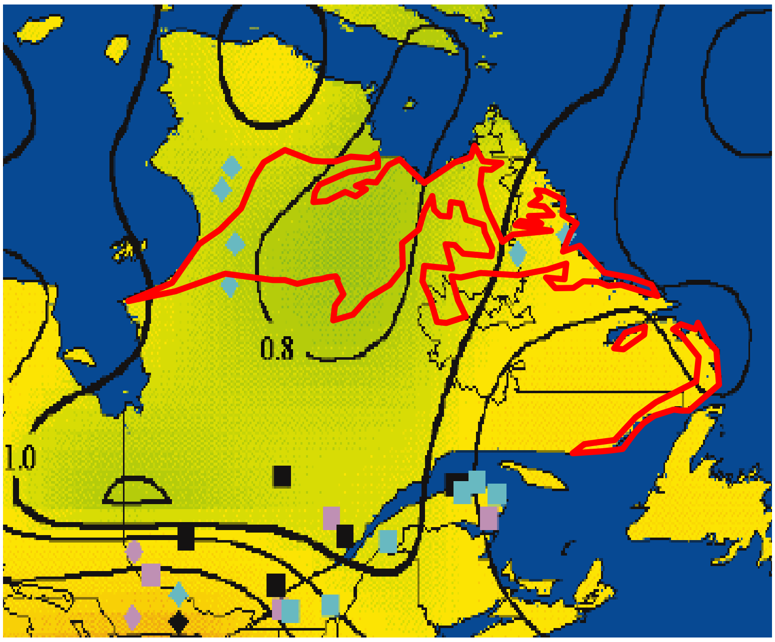

Figure 5.5: Approximate position of the forest-tundra (in red) projected on the 2050 FWI map (Flannigan *et al*., 2001).

Tree regeneration in the forest-tundra depends on several factors. Firstly, regeneration patterns depend on the presence of forest with normal cone development or of krummholz with few or no cones. Second, the presence of fire, combined with local climate variations, can strongly limit postfire tree regeneration, e.g. a fire during a cooling event can cause the replacement of a tree stand by krummholz (Payette and Gagnon, 1985). The gradual opening and creation of the forest-tundra in its present state was dated around 3000 cal. BP (Payette and Gagnon, 1985) and resulted from an increase in fire occurrence combined with a cooler climate (Asselin and Payette, 2005).

5.4.1 Changes in climate, fire regime and C stocks in the forest-tundra

Like most of the other parts of the Québec boreal forest biomes the 2050 - 2075 FWI will decrease relative to the modern values: a FWI of 0.75 to 0.8 was estimated in the central forest-tundra and 0.9 at the Hudson Bay coast; in the east, the values may be close to 1.0 (Flannigan *et al*., 2001).

Independent of soil moisture conditions, pre- and postfire stand tree densities are influenced by latitudinal position as shown by studies on fire effects along a boreal forest – forest-tundra transect (Sirois and Payette, 1991), probably being the result of a lower regeneration potential of black spruce under the influence of colder climatic conditions. This strong dependence on temperature makes the forest-tundra to differ from the southern boreal biomes, where tree growth is not or only slightly limited by temperature. Thus, the expected increase in temperature will affect to a greater extent and more directly vegetation patterns in the forest-tundra, and changes in fire severity might be of minor importance, also because fires are already rare around the treeline (Payette and Gagnon, 1985).

If we follow the line of thought as exposed by Payette and Gagnon (1985), the great warming trend announced for the Québec forest-tundra biome may well cause a densification of the forest, as the temperature constraints on seed production and regeneration will be relieved. A denser pattern was also suggested by Asselin and Payette (2005). However, in the latter research, an increase in fire frequency was announced due to both the higher temperatures and the densification of the forest. The climate argument is partially tackled by Flannigan *et al*. (2001), who showed that the FWI generally decreases in this biome, as the increase in precipitation will be of greater importance than the increase of temperature. The effect of an eventual densification of the forest on fire occurrence, however, was not verified by Flannigan *et al*. (2001). Given the fact that in this region tree reproduction is strongly limited by temperature and that a warmer but more humid climate was announced while actual fire cycles can be extremely long especially in the northern part (Payette *et al*., 1989), it seems more probable that, concerning a change in tree density, the relieve of stress on tree production by a warming climate will be more important than a potential increase in fire frequency. However, in the southern forest-tundra, where actual fire frequencies are around 180 years and mean temperatures are currently less important in limiting tree growth, trees may have a greater difficulty in regenerating when fire cycles increase. The lower FWI values due to a great increase in precipitation may cause a generally more humid forest floor. Fire severity is likely to decrease resulting from the increases in both summer and winter precipitation, which forms another factor positively influencing thicker forest floors. The warmer climate may cause a further expansion of both black spruce and white spruce in the northern sites, the latter being a species that already increases its presence in the Hudson Bay region (Caccianiga and Payette, 2006) as a result of a constant seed release that causes regeneration immediately when climate conditions are favourable. In addition, the limited regeneration of black spruce in unburned conditions may increase the possibilities for white spruce to establish (Parisien and Sirois, 2003; Lloyd *et al*., 2005) as fire frequency decreases.

The increase in winter precipitation in north-western Québec can be interpreted as an increase in the mean snow cover, even when winter temperatures rise substantially. This is because winter temperatures will remain low enough to sustain snowfall. Thick snowdecks and high temperatures will cause a diminishing area influenced by permafrost. In addition, the general increase in tree cover will cause a better snow insulation as the snow will be prevented from drifting (Payette *et al*., 1986). The melting of the permafrost in some regions may induce local shifts from a spruce-lichen vegetation to a dominance of *Sphagnum* sp. and *Carex* sp. vegetation (*cf*. Camill *et al*., 2001). The effect of changes in permafrost abundance and distribution on C stocks depend on the reaction of dominant vegetation, production, decomposition and N mineralization. Camill *et al*. (2001) found increasing rates of C accumulation after permafrost thaw in boreal peatlands in northern Manitoba with increases in both production and decomposition rates. However, the great number of small-scale spatial factors make it difficult to predict C accumulation patterns after permafrost thaw for a specific region.

The increase in tree cover in the northern part of the forest-tundra may cause an increase in the ecosystem C stock primarily due to an increase in the mean temperature, while the decrease in fire severity and frequency in the southern part may induce thicker forest floors and comparatively small changes in tree production, but a potential increase in the forest floor C stock.

6 Discussion

The aim of the study was to investigate the effect of climate change on fire severity in the boreal forest and the consequences for the long-term C stock. Although fire severity was found to be a major factor that not only determines postfire vegetation development but also shows to be strongly influenced by a changing climate, a number of other factors should be regarded while investigating the future's boreal C stock. Firstly, climate itself influences the boreal C stock by regulating vegetation growth rates, forest floor decomposition rates and permafrost development. In some cases, the direct influence of temperature and precipitation were discussed, because of their importance in the aforementioned processes. However, the direct influence of climate was not part of the aim of the study. Second, as already mentioned, fire frequency is an important factor, as the recurrent local presence of fire potentially prevents a full development of the vegetation, independent of fire severity. The stand age distribution across the landscape, which is an important factor determining the boreal C stock, is primarily a function of fire frequency.

The FWI and its components FFMC, DMC and DC have limitations in estimating the potential future fire severity. Primarily, the codes are based solely on meteorological parameters that do not take into account stand-specific vegetation patterns. Wotton and Beverly (2007) provide data to convert FFMC values into litter moisture contents for a large number of stand types, making it somewhat easier to estimate the effect of stand type on potential fire severity under certain meteorological conditions. However, these type of calibrations are limited to a few forest types (e.g. Lawson and Dalrymple, 1996) for the deeper duff layers, whereas these layers can be significantly affected by smouldering while they contain comparatively the largest amounts of C. In addition, no scenarios are known for future FFMC, DMC and DC values individually. The FWI was used in this study in combination with climate scenarios to verify the relative importance of a future increase in precipitation and temperature in determining the 2050 – 2075 fire regimes. All scenarios (Flannigan *et al*., 1998; 2001; Price *et al*., 2001; Plummer *et al*., 2006; IPCC, 2007) indicated a general increase in both mean temperature and precipitation around 2050 – 2075, suggesting a solid base to estimate corresponding changes in the fire regimes. However, recent publications have estimated fire regimes using GCMs during the 2060 – 2100 period with an atmospheric CO_2 concentration 3 times that of 1975 – 1995 (e.g. Flannigan *et al*., 2005;

Girardin and Mudelsee, 2008). These studies show an increase in fire activity for all ecoregions in Canada, showing that the expected rise in precipitation will not be able to compensate for the increase in temperature at the end of the 21st century. The differences between these studies and the studies discussed in this paper are, among others, the effect of different CO_2 emission scenarios, model types, sources of predictor variables and reference periods (Girardin and Mudelsee, 2008). These studies did not take into account the potential variations in the vegetation cover under a changing climate, so that difficulties remain in estimating the general potential for deep burning. In addition, large variations in fire activity among regions are possible and scenarios were obtained only for the May to July period, while severe burning is more probable in July and August.

If the recent and ongoing climate change can be used as a model for the nearby future, the observed changes in vegetation resulting from a shift in fire regime should be indicative for the upcoming shift in the boreal forest vegetation. Bergeron (1998) registered a shift in vegetation resulting from lower fire frequencies since the end of the LIA around 150 years ago. In the balsam fir – paper birch forest, the increase in temperature and the more frequent presence of humid air masses have caused a more rapid tree growth and an increase in balsam fir and white cedar (*Thuja occidentalis* L.) presence and a decrease in jack pine presence through the increase in fire cycles (Bergeron, 1998). This forms another solid indication of the potential of vegetation development in the Québec boreal forests in a changing climate.

In investigating the potential of the 2050 – 2075 boreal forest C stocks, there are a number of other factors that should be considered that were not mentioned here. Human disturbance factors will further influence boreal forest development by fire suppression and harvesting, while other disturbance factors e.g. the spruce budworm (alone or in combination with fire) have already been proven to change stand types (e.g. Jasinski and Payette, 2005).

The publications used in the synthesis provided each a clear idea of natural processes as forest regeneration and burning mechanisms. However, it was proven to be extremely difficult to discern clear patterns by comparing various study sites because the processes observed are often at least partially a reaction on site-specific characteristics. Thus, caution was necessary in estimating general trends concerning these processes.

To estimate the effect of changes in fire severity on the total C storage in the boreal forest, there is a need to quantify C stocks for the components (plant biomass and forest floor) that constitute the boreal forest ecosystem. As the aim of the study was to estimate long-term changes in boreal C stock, the typical boreal forest C stock for each type of biome as obtained from a number of studies from various regions needed to be linked to the expected shifts in biome distribution.

As indicated before, stand C contents at any given moment are influenced by a great number of factors, e.g. stand age (Yu *et al*., 2002), past fire severity (Johnstone and Chapin, 2006), climate (temperature and precipitation), mineral substrate (Bhatti and Apps, 2000), dominant vegetation species and the relative presence of overstorey, understorey and a moss layer. To estimate the change in the boreal C stock resulting from climate change and the shift in fire severity regimes, all these factors need to be taken into account, except for the mineral substrate as it is not related to changes in climate. Gower *et al*. (1997) and Yu *et al*. (2002) concluded that C reservoirs vary more between different stand types in a similar climate than between similar stand types in differing climate zones (Table 6.1).

	Climate characteristics			Mean total stand C content (kg m^{-2})							
	Mean January temperature (°C)	Mean July temperature (°C)	Mean annual precipitation (mm)	Aspen	Black spruce		Jack pine (25-27 yrs)	Jack pine (60-65 yrs)	Scots pine	Norway spruce	Silver birch
					Feather moss	Sphagnum					
53°20' N 105°05' W	- 19.8	+ 17.6	405	15..9 [1]	44.6 [1] 11.3 [2]	10.7 [2]	5.1 [1]	6.9 [1]	-	-	-
55°14' N 97°36' W	- 25.0	+ 15.7	536	17.6 [1]	47.9 [1]	-	7.6 [1]	6.8 [1]	-	-	-
47°38 N 83°15 W	- 16.0	+ 17.0	834	-	-	-	-	16.1 [4]	-	-	-
49°30 N 87°50 W	- 20.2	+ 16.9	784	-	25.1 [4]	-	-	-	-	-	-
62°58' N 27°40' E	- 10.0	+ 18.0	478	-	-	-	-	-	13.2 [3]	26.1 [3]	12.5 [3]
66°58' N 25°40' E	- 11.0	+ 15.0	459	-	-	-	-	-	9.8 [3]	20.2 [3]	10.1 [3]

Table 6.1: Mean total stand C content for various stands from a number of regions. [1]Gower *et al*. (1997); [2]O'Connell *et al*. (2003); [3]Briceño-Elizondo *et al*. (2006); [4]Morrison *et al*. (1993).

6.1 Stand age effects

Tree biomass C increases with time for different tree species according to a number of studies (Gower *et al*., 1997; Nalder and Wein, 1999; Yu *et al*., 2002; Lecomte *et al*., 2006), however a decline in tree production is possible due to a poor regeneration potential after postfire stand break-up (Lecomte *et al*., 2006). The onset of tree C accumulation rates slowing down depends on the initial growth rates (Gower *et al*., 1997) and the previous fire severity (Lecomte *et al*., 2006). The effects of stand break-up on the stand C budget may be limited however, as snags may remain standing for 20 years after fire and fallen logs may decompose only very slowly as they eventually become part of the humid forest floor (Nalder and Wein, 1999). In jack pine stands, moss and lichen C show strong linear increases with time in stands up to 150 years (Nalder and Wein, 1999). In addition, forest floor C contents are highly correlated with stand age (Paré *et al*., 1993; Nalder and Wein, 1999; O'Neill *et al*., 2006). Coarse woody debris is high directly after stand establishment; its importance diminishes shortly after but can return to increase when stands break up or by self-thinning as found in aspen and pine stands, respectively (Yu *et al*., 2002).

Studies by Lecomte *et al*. (2006) have shown that even 700 year-old stands still increase their total biomass, increasing forest floor C amounts while tree biomass C remain relatively constant. Data from Wardle *et al*. (2003) indicated that, regarding stand age on the long term, the absence of fire causes a decrease in decomposition rates that acts before the decline in ecosystem productivity. On the stand scale, Gower *et al*. (1997) found two mature black spruce stands (115-155 years) containing 44.6 – 47.9 kg C m^{-2}, while young (25-65 years) jack pine stands showed values of 5.1 – 7.6 kg C m^{-2}. However, because other factors as anterior fire severity and dominant species vary as well, it is uncertain if the latter findings can be dedicated solely to variations in stand age. Wardle *et al*. (2003) found increasing stand C contents in ageing stands primarily resulting from a reduced forest floor decomposition (Figure 6.1).

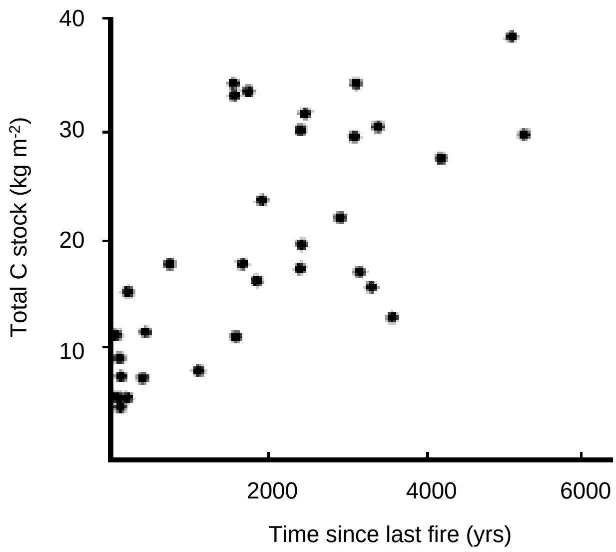

Figure 6.1: Total stand C stock increase with increasing time since the last local fire from a number of island sites (redrawn from Wardle *et al*., 2003).

6.2 Fire severity and species effects

Species effects on C contents in the boreal forest is highly influenced by past fire severity, as fire severity has a strong control on postfire vegetation re-establishment. Forest floor C was found to be higher in trembling aspen stands compared to jack pine stands (Nalder and Wein, 1999) but black spruce stands forest floor C contents are generally higher than those in aspen stands (Yu *et al*., 2002). Jack pine stands often establish after a severe fire that removes all organic material from the forest floor. Despite the fact that Nalder and Wein (1999) did not verify for past fire severity, this indicates that the species effect may actually be a fire severity effect, given the strong link between fire severity and postfire tree species establishment discussed above. This is strengthened by the fact that the studied stands were from the same climatic region. High fire severity causes a high initial C accumulation in the overstorey but an earlier break-up of the tree canopy (Lecomte *et al*., 2006). This break-up may cause a decline in long-term litterfall and a decrease in forest floor C accumulation.

A bryophyte layer that covers the forest floor positively influences C accumulation in the forest floor. In addition, bryophyte covers benefit from large amounts of overstorey C and shrub C, as dense tree covers or a shrub layer cause shading conditions favouring *Pleurozium schreberi* development, after which *Sphagnum* sp. can establish. Nalder and Wein (1999) found a highly positive link between tree C and forest floor C, which was explained by the higher litterfall from developed overstorey.

6.3 Climate effects

From the climatic parameters, temperature and precipitation are the most important in influencing C accumulation patterns. Data from Yu *et al*. (2002) and Nalder and Wein (1999) suggest that, concerning C contents of boreal stands, high temperatures causing high litter production rates are of a greater importance than high precipitation values maintaining low decomposition rates. This was suggested by high C accumulation in relatively warm forest floors in a comparison between three trembling aspen stands of equal age (Nalder and Wein, 1999). However, both Simmons *et al*. (1996) and Gower *et al*. (1997) found higher C amounts stored in stands from a colder climate in Maine and Saskatchewan/Manitoba, respectively, while other factors remained relatively constant. These differences in forest floor C content along climatic gradients may be explained by species-specific differential growth and decomposition response to varying climates.

Wang *et al*. (2003) studied chronosequences of C distribution in wet and dry black spruce stands. When comparing wet and dry black spruce stands in comparable climatic conditions, dry sites show greater vegetation biomass (including overstorey, understorey and ground cover biomass) than wet sites (Wang *et al*., 2003). However, in wet stands, understorey and bryophyte biomass as well as the forest floor C pool are greater. This was confirmed by Nalder and Wein (1999) for trembling aspen stands, as the mean annual temperature was highly positively correlated to forest floor C contents while precipitation was negatively correlated. When considering total postfire biomass development, dry sites accumulate C at higher rates and peak at a higher level than wet sites (Wang *et al*., 2003).

One of the key questions here is: does an stand established after a low-severity fire generally incorporate higher amounts of C than a stand established after high-severity fire, if other factors (e.g. climate) are constant? An increase in ecosystem C after a low-severity fire is possible if the impact of thicker forest floors on stand C sequestration outnumbers the consequent lower values in tree biomass C. Considering the high C contents in *Sphagnum* sp. forest floors or peat accumulation as compared to black spruce stand tree biomass (e.g. Gower *et al*., 1997), this seems indeed probable. This is further illustrated by forest floor C data. Wirth (2005) found an average ~ 2.0 kg C m^{-2} of tree biomass C and ~ 5.0 kg C m^{-2} of organic layer C for a variety of stand types in central Canada while for the same region Bhatti and Apps (2000) reported mean values of ~ 4.5 kg C m^{-2} and 3.7 kg C m^{-2}, respectively. From other studies, black spruce stand tree C contents of 4.9 to 5.7 kg C m^{-2} were found and even the highest living tree C in relatively warm aspen stands did not exceed 9.3 kg C m^{-2} (Gower *et al*., 1997). The highest stand total C contents from Nalder and Wein (1999) were 8.5 kg C m^{-2} of which 33 % was stored in the forest floor. The fact that paludification is a potential process in forests on fine-textured soils that lack disturbance for a long period (Fenton *et al*., 2005) justifies the comparison of potential forest floor C content with peat C contents: 2.3 m of vetical peat accumulation typically contains 133 kg C m^{-2} (Gorham, 1991), indicating the potential for large total boreal stand C stocks after paludification.

In short, old stands that develop thick forest floors with a bryophyte cover and an overstorey that is typically characterized by black spruce (Van Cleve *et al*., 1983) contain large amounts of C even when overstorey is in decline (Wardle *et al*., 1997). In contrast, young stands with a jack pine cover and a shallow forest floor contain less ecosystem C.

7 Conclusion

The influence of fire severity on the C stock of the boreal biomes has two aspects. Primarily, the severity of fire determines to a large extent the amount of C released during fire. Second, there is the indirect influence of fire severity on the various pathways of plant regeneration and thus the potential in determining postfire C accumulation patterns.

The importance of fire severity should be studied while taking into account the frequency of fire. The frequency of fire has the potential to determine the period of C accumulation according to the pattern that was induced by the severity of the last fire. Thus, fire severity is a relevant factor if burning is frequent enough to prevent a stand from developing a uniform C distribution. The stand-specific potential to maintain an initial stand type resulting from a specific fire severity regime in the absence of general disturbance is difficult to estimate, as it may be dependent on a great number of factors. Climate is an important one, and because of the fact that the present-day climate changes rapidly and continuously, stand evolution on the long term will always show a (minor) climate factor.

The general patterns of postfire vegetation re-establishment as a function of fire severity in combination with a number of climate and forest floor moisture scenarios delivered an estimate of the future C stock in the Québec boreal biomes. The combination of an increase in precipitation together with an expected lower fire intensity that outstrip an increase in summer temperature will lower direct C emissions due to shallow burning and will be advantageous to a rapid postfire forest floor C accumulation and disadvantageous to tree C stocks. Due to its efficient suppression of decomposition rates, forest floors C stocks may increase substantially especially if *Sphagnum* sp. are able to establish. Thus, the dominant factors that may cause a change in the future fire severity regime are the change in both summer and winter precipitation relative to changes in temperature and the vegetation type that provides litter and determines forest floor temperature and humidity.

For future research, it is desirable to obtain specific scenarios on the various moisture codes as well as a greater number of moisture code – moisture content calibration curves. In addition, the stand-specific patterns of vegetation development in absence of fire need to be better known to be able to estimate the future boreal C stocks.

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