# WILDFIRE DYNAMICS OF MIXEDWOOD BOREAL FORESTS IN CENTRAL CANADA

By

Dominic Lafontaine-Senici

A Thesis Submitted in

Partial Fulfillment of the Requirements for the

Degree of Master of Science in Forestry

Faculty of Forestry and the Forest Environment

Lakehead University

August 2009

## ABSTRACT

Lafontaine-Senici, D. 2009. Wildfire dynamics of mixedwood boreal forest in central Canada. 86 pp.

Key Words: boreal forest, ecosystem management, Cox regression, fire cycle, fire frequency, fire history, fire regime, forest management, GIS, natural disturbance, Ontario, stand structure, survival analysis,

Natural disturbances play a key role to forest regeneration, nutrient cycling, wildlife habitat, biodiversity, and climate regulation. A synthesis of literature indicated that boreal forest stand structure is largely a construct of the regional fire regime and time since-stand-replacing fire (TSF). Regional differences in fire frequency exist across the boreal forest based primarily on broad-scale climatic patterns, with local-scale variations dependant on vegetation, fuels, and physiographic features. Fire frequency has also been shown to vary according to human activities, although the magnitude and influence (increase or decrease) is non-uniform, creating multiple instances of spatially mixed fire frequencies. Several gaps in fire history reconstructions based on dendroecological sampling were detected in the central and northwestern boreal shield regions of North America. My objectives were to reconstruct the fire frequency of northwestern Ontario and to investigate multiple spatial or temporal parameters that may be influencing fire frequency at different scales.

Archival sources were used to reconstruct the fire history of an 11, 600 $km^2$ area in the Lake Nipigon ecodistrict of the central boreal forest region of North America. Records allowed us to map the location and extent of (almost) all fires that burned an area greater than 200 ha between 1920 and 2008. Sampling was conducted using a systematic grid system to subdivide the study area into 407 hexagons, each measuring 40 $km^2$. Time-since-fire dates were transferred to hexagons based on the proportion of area burned. Hexes with no TSF determination were sampled in field; canopy tree cores and disks were collected from pioneer species that colonize post-fire and acted as a TSF proxy.

Survival analyses were conducted to investigate temporal and spatial parameters influencing fire frequency. Multiple fire cycles for different time-periods were calculated to check if the fire cycle has remained constant over time. My results indicated that in the past (1820-1920) the fire cycle was longer at 295 years than in recent years (1920-2008) at 96 with a cumulative fire cycle from 1820-2008 to be 150 years. Fire cycle was also found to be shorter (FC; 82 years) in the northern partition of the study area than in the southern partition (FC; 100 years) during the 1920-2008 time period indicating the influence of human activities contributing to spatially mixed fire frequencies.

Fire frequency within the region varies according to latitude, soil order (and associated drainage class), and mean distance to a water body. Sites at the northern limit of the study area have nearly 3 times the burn rate (fire frequency) than sites at the southern limit. Sites of soil order brunisolic and excessively well-drained soil drainage class burn less frequently than sites of soil order podzolic and moderately well-drained soil drainage class. As distance from a water body increases, so too does fire frequency;

compared to sites adjacent to water bodies, those with an average distance of 1 kilometre have roughly 33% increased burn rate. No other differences in fire frequency were observed owing to longitude, surficial deposit, slope aspect, and elevation.

The results of the study suggest that (a) as expected, fire frequency in north western Ontario boreal mixedwood forests respond more to broad-scale 'top-down' factors like climate, than to 'bottom-up' local environmental conditions, (b) fire cycle has changed over time in response to climate and human land-use in the area including landscape fragmentation and fire suppression, and (c) areas with a higher proportion of deciduous species burn less frequently than areas with a higher proportion of coniferous species.

CONTENTS

# TABLES

FIGURES

# ACKNOWLEDGEMENTS

I wish to give my sincere appreciation to my supervisors Dr. Han Chen and Dr. Yves Bergeron for their assistance, support and guidance throughout the development of my thesis. I would also like to thank my committee members Dr. Jian Wang, John MacGillivray, and Michael Gluck and my external reviewer Dr. Sylvie Gauthier. I am grateful towards Dominic Cyr for providing assistance in the laboratory and as a constant source of support. I am grateful towards Lindsey Jupp, Meelis Seedre, and Samuel Bartels for assistance in the field. Funding was provided by grants from the Sustainable Forest Management Network, and the Natural Sciences and Engineering Research Council and Abitibi-Bowater, and are greatly appreciated.

# CHAPTER ONE. GENERAL INTRODUCTION

The boreal forest constitutes the largest forest region of Canada. Composed of four major ecozones (Atlantic Maritime, Boreal Shield, Boreal Plains, and Boreal Cordillera) the boreal forest forms a continuous, transcontinental forest belt from Newfoundland in the east to the state of Alaska in the west between 45° and 70° north latitude. The boreal forest is of variable of species structure and composition, but its overall characteristic imparts a distinctiveness that sets it apart from other world forest regions. Climate and disturbance regimes, locally and regionally, characterize the boreal region more than anything else. Extremely cold long winters and short warm summers are the norm, with mean annual temperatures ranging from a low of -8°C at northern latitudes to 6°C in warmer and moister southern latitudes (Rowe 1972). Multiple natural disturbances are common, with fire being a critical ecosystem process organizing forest plant communities, successional patterns, and ecosystem structure across the boreal landscape (Johnson 1992; Weber & Stocks 1998).

Boreal forest fires are the result of specific weather and fuel conditions and an ignition source. The boreal forest, now and in the past, is characterized by the fire regime, or regional expected pattern and behaviour of fire. The fire regime is comprised of six components: fire occurance, size, intensity, seasonality, type and severity (Heinselman 1973; Malanson 1987; Whelan 1995). In addition, discussion of the fire regime typically includes the cause of ignition, which in the boreal forest are either lightning and human-initiated. Fires in the boreal forest occur frequently during the principle fire season (April through mid-October) and are generally small in size affecting few hectares. It is the infrequent fires occurring during severe fire weather that burn anywhere from one hundred to one hundred thousand hectares of forest that characterize the boreal fire regime (Johnson 1992). Within the boreal region long-term average annual area burned is 1.3 million ha, with extreme fire years burning over 7 million ha in a single fire season (Weber & Stocks 1998); fire regimes are highly variable, changing in response to various temporal and spatial factors.

Application of management strategies based on emulating natural disturbance regimes is constrained by limited knowledge of the disturbance dynamics that characterize a given region. Estimates of the disturbance rate by fire enter into calculations of predicted wood and habitat supply used in Ontario's forest management planning process (Bridge 2001). Given that the state of knowledge the northwestern Ontario fire regime is fragmented (Woods & Day 1977; Suffling & Molin 1982), and our understanding that the fire regime is highly variable and spatially dependent, it is inappropriate to generalize existing results over whole landscapes. With the advent of new spatial fire data that has become available, and the use of dendroecological sampling and advanced statistical techniques (Johnson & Gutsell 1994), this thesis strives to quantify the variability inherent in the regional fire regime and provide fire cycle estimates to forest managers for a large area of commercial forest in northwestern Ontario for use in the forest management planning process.

The three objectives of this thesis were to (1), synthesize the published but fragmented knowledge on temporal and spatial factors influencing current and future boreal forest fire regimes with the goal of understanding the state of knowledge and identifying gaps and uncertainties; (2) to reconstruct fire history and investigate whether or not fire frequency has changed in a large cross-section of boreal forest in northwestern Ontario; and (3) determine if fire frequency is affected by various spatial parameters common to this region of boreal forest. It is my hope that the findings of this thesis will help forest managers make informed decisions in maintaining an ecologically sound forest composition and structure.

# CHAPTER TWO. WILDFIRE DYNAMICS OF NORTH AMERICAN BOREAL FORESTS: A REVIEW

## INTRODUCTION

Covering an excess of 5.9 million square kilometres in a continuous belt from northern Newfoundland to western Alaska, the boreal forest of North America is one of the largest forested areas in the world. Climate varies throughout, ranging from dry and cold with minimum mean annual temperature of -8°C and mean annual precipitation of 300 mm to warm and moist with a maximum mean annual temperature of 6°C and mean annual precipitation of 1350 mm (Rowe 1972). The boreal forest is characterized by the predominance of coniferous trees of multiple genera like spruce (*Picea*), pine (*Pinus*), fir (*Abies*), and larch (*Larix*), interspersed with deciduous hardwoods of aspen (*Populus*), birch (*Betula*), alder (*Alnus*), and willow (*Salix*). The current mosaic of boreal forest stand composition, specifically species assemblages and population dynamics, are a result of the complex interactions between climate, species ecology, anthropogenic land use, and natural disturbances (Bonan & Shugart 1989).

Trees, the most visible and main structural element of the boreal forest ecosystem, are mainly regulated by natural disturbance. Disturbance (as defined in this paper) refers to any natural event that disrupts a forest ecosystem by weakening or killing trees thereby initiating changes in stand structure over time. These changes in forest stand structure over time, or forest stand dynamics (Oliver & Larson 1990; Chen & Popadiouk 2002), are driven by the severity, frequency, magnitude and concurrence of multiple natural disturbances operating at multiple scales. Common natural disturbances within the boreal forest include: forest fire, insect outbreaks, disease, wind throw, and ice storms. Of these disturbances, forest fires, specifically, stand-replacing high intensity crown fires are the dominant natural disturbance that characterizing boreal forest ecosystems (Bonan & Shugart 1989; Johnson 1992).

For the purpose of this paper, the review of fire frequency influencing North American boreal forest composition will include an examination of (1) the boreal fire regime, it's primary components, drivers, and causes, (2) how fire frequency varies temporally and spatially within the boreal forest, (3) the impact of human activities on fire frequency, and (4) the impact of climate change on future fire regimes and frequency within the North American boreal forest.

## FIRE REGIME: THE DOMINANT NATURAL DISTURBANCE IN THE NORTH AMERICAN BOREAL FOREST

Fire has been the dominant natural disturbance in the boreal forest since the last Ice Age. The aspect and character of the boreal forest is not defined by any individual fire, but rather, as a function of the fire regime, where fire regime defines the expected regional fire activity and behaviour. Recurring fire is a critical process for the persistence of pioneer boreal species such as pine and aspen, and is responsible for shaping the observed mosaic of vegetation in the landscape. Past, present, and future forest stand structure and composition is influenced by regional fire regimes (Heinselman 1971). Fire regime is highly variable across a continental longitudinal gradient (Malanson 1987; Flannigan & Harrington 1988; Bonan & Shugart 1989; Bergeron & Flannigan 1995); however, a general trend towards increasing fire frequency and increasingly larger areas burned has been detected in recent decades (Stocks 1991; Flannigan et al. 1998; Soja et al. 2007). Generally the boreal forest of

North America is characterized by infrequent but high intensity crown fires occurring during the principal fire season (April through mid-October) that consume thousands of hectares of standing vegetation (Rowe & Scotter 1973; Bonan & Shugart 1989). Historically, large crown fires represent only 3% of the total number of fires recorded, but account for 97% of the total area burned (Stocks et al. 2002).

Describing the fire regime typically involves a discussion of six primary components: occurrence, intensity, seasonality, severity, size and type (Whelan 1995). Occurrence refers to the number of fires occurring per unit time (typically annually) in a given area. Depending on the recurrence interval, or time it takes for a fire to return within a given stand, selection pressures will favour organisms able to take advantage of the interval length. Fire intensity refers to the rate of heat energy released and is strongly correlated with fuel loading, daily weather conditions, and regional climate; additionally the intensity of a given fire can vary greatly depending on the characteristics of a previous disturbance (e.g. spruce budworm outbreak) or the time-since-last-fire (TSF) in the same location (Malanson 1987; Whelan 1995). The seasonality (time within year) of a fire can strongly influence successional trajectory by moderating vegetative or seed reproductive responses based on the phenological state of the pre-fire vegetation composition (Weber & Flannigan 1997). The season in which fire occurs can also affect fire intensity through seasonal climate influencing moisture on the surface and in the crowns of the trees. Fire severity, a measure of the magnitude of impact including the combined above and below ground biomass loss, strongly directs succession in post-fire communities through direct impacts on residual plant roots, seed bank, nutrient availability and microbial composition (Jayen et al. 2006).

The size of a fire, or total area burned, largely determines the forest mosaic and sizes of different aged patches within the boreal landscape. Fire size also influences the dispersal of propagules from the edges of a fire's boundaries thereby indirectly influencing post-fire regeneration (Malanson 1987; Oliver & Larson 1990). Lastly, fire type describes the class of fire, ground, surface, or crown, based on the fuel layer(s) consumed in the combustion process. Ground fires refer to smouldering deep within the organic layer of forest floor; surface fires burn though grass, shrubs, fallen trees or logs, and needle and leaf litter on the soils surface; and, crown fires burn throughout the crowns of standing trees. Fire type is rarely uniform throughout a burn, differences arising from pre-fire vegetation composition, and physiographic features (e.g. slope). Boreal forest ecosystem dynamics are inextricably linked to these six components of the fire regime which are themselves highly dependent on daily weather conditions and seasonal climate (Flannigan & Harrington 1988; Johnson 1992).

Aside from the fire regime components discussed, several fire history concepts have come to use, which for consistency throughout this manuscript will be defined here. Fire frequency (burn rate) is defined as the annual percent area burned within a study area. The fire cycle (average fire interval) is the inverse of fire frequency, and is the time required to burn an area equal in size to the studied area. Lastly the fire return interval is fire's expected return time within a stand.

The pattern of ecological succession, the appearance and composition of vegetation, in the boreal forest is governed, primarily, by complex interactions with multiple components of the regional fire regime. The boreal landscape is generally characterized by a fire cycle, or the time it takes to burn an area equivalent to the area studied, ranging anywhere from 30 to 500 years (Flannigan *et al.* 1998), and is frequently composed of pure or mixed even-aged stands at different stages of recovery post-fire. Although differences in forest compositions exist between geographic areas and topographic positions, a general pattern of post-fire forest development can be recognized (Figure 2.1).

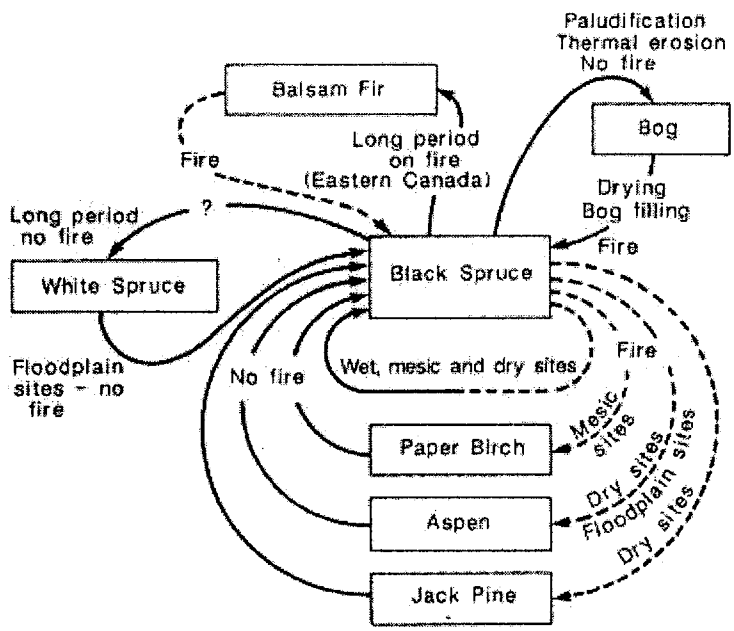



Figure 2.1. Successional relationships among some northern black spruce dominated North American boreal forest types in relation to the occurrence of forest fires (From Vierick 1983).

The post-fire cohort species composition is dependent upon the availability of `propagules` like seeds, vegetative reproductive structures, and persistent seedlings (Fricker et al. 2006; Chen et al. 2009; Ilisson & Chen 2009a; Ilisson & Chen 2009b). Coniferous species with serotinous or semi-serotinous cones, specifically black spruce (*Picea mariana*) and jack pine (*Pinus banksiana*), and deciduous species reproducing via suckering, sprouting, and rapid re-seeding like trembling aspen (*Populus* *tremuloides*) or paper birch (*Betula papyrifera*), are well suited for abundant and rapid post-fire regeneration and frequently dominate the post-fire cohort (Zasada et al. 1992; Bergeron & Dansereau 1993). Dependent on fire severity, growing space and nutrients are relatively abundant and active recruitment of shade intolerant species or rapid regrowth from residual patches occur. After the initial colonization recruitment slows down, and provided that fire return interval is long, longer-lived shade tolerant species, such as white spruce (*Picea glauca*) or balsam fir (*Abies balsamea*), can regenerate under the closed canopy of the initial cohort. It should be noted that given a short fire cycle the landscape will be dominated by young (< 100 years) patches of shade intolerant pioneer species like jack pine.

Fire regimes and their controls vary across a wide range of temporal and spatial scales (Payette et al. 1989; Flannigan & Van Wagner 1991; Johnson 1992). This variation is the result of complex interactions between the primary driver, climate, and vegetation, fuels and physiographic features operating at different scales. At a broad scale, regional climate influences fire regime directly as temperature and precipitation affect the moisture content of fuel, and thus the probability of ignition as well as the size, severity, intensity of the resultant fires. Climate indirectly affects the fire regime by influencing forest composition such as the rate of vegetative growth thereby influencing the amount and arrangement of fuels present in the landscape. In contrast, to broad-scale climatic factors, local variation of physiographic features can affect fuel moisture content and fuel continuity thereby influencing local ignition probabilities, fire size, and rate of spread (Bonan & Shugart 1989). It should be noted that the primary determinant of the fire frequency in the boreal forest is weather variation (influencing fuel moisture) among years rather than fuel variation associated with stand age, structure and composition (Bessie & Johnson 1995; Johnson et al. 2001). That being said, some components of the fire regime, specifically fire intensity and severity, as well as the tendency of a fire to crown, varies among forests owing to differences in vegetation composition and the distribution of fuels within a stand (Bonan & Shugart 1989).

Climate: broad-scale primary driver of the fire regime

Weather and climate are crucial to the occurrence and growth of forest fires. Local, regional and global changes in temperature and precipitation can influence the occurrence, timing, frequency, duration, extent, and intensity of fire (Flannigan & Van Wagner 1991). Weather conditions during fire initiation and the climatic conditions preceding ignition are the central agents determining fire behaviour. Precipitation and thunderstorms influence the likelihood of ignition by influencing fuel moisture, rate of combustion and spread, and initiation from lightning strikes.

Lightning is the result of an electrical discharge from a thunderstorm. The fire season over much of boreal North America is relatively short with the majority of fire activity occurring between May and September, when higher temperatures and atmospheric instability create thunderstorms more frequently. Lightning initiated fires represent about 45% and 15% of all reported fires in Canada and Alaska respectively, yet are responsible for 81% and 90% of the total area burned (Wotton et al. 2003; Soja et al. 2007). Lightning-initiated fires tend to burn large areas because they occur randomly, often in remote locations where detection and subsequent fire management responses are delayed. Lightning-initiation is regulated regionally by abiotic and biotic factors such as weather and forest composition. Broad-scale weather patterns determine the location of lightning events that ignite fires as well as moisture properties of forest fuels resulting from recent weather conditions such as precipitation, temperature, wind, and humidity (Van Wagner 1987). Forest composition, including the presence or absence of particular species (or any species), influences the probability of lightning-initiation. For example, spruce-dominated landscapes have a greater probability of initiation than aspen, owing to the quality and quantity of fuels among them; similarly, harvested areas or areas that have burned recently have a lower probability of initiation due to the relative absence of available fuel (Hely et al. 2000; Krawchuk et al. 2006).

The occurrence of human-caused fires generally depend on persistent climatic and daily weather conditions influencing fine fuel moisture content the moment an ignition source (e.g., smoking embers, recreational, equipment, and miscellaneous) comes in contact with the forest floor. Human-caused fires constitute majority of reported forest fires in the boreal forest, they are however detected early in development and contained quickly resulting in a smaller proportion of total area burned per year (Wotton et al. 2003).

Large forest fires in the North American boreal forest are often associated with the presence of persistent mid-tropospheric (500 hPa) ridging at boreal latitudes (Skinner et al. 1999; Skinner et al. 2002). The location, magnitude and persistence of these prolonged blocking high-pressure systems in the upper atmosphere greatly affect the weather and climate of North America. These systems not only obstruct the normal west-to-east progress of migratory storms, but cause air subsidence in the upper atmosphere, resulting in long periods of sunny, warm days creating dry fuel conditions over hundreds of kilometres (Newark 1975; Johnson & Wowchuk 1993; Skinner et al.

1999). Increased incidences of convective activity causing lightning strikes have been linked to the accumulation of significant moisture or the breakdown of these high-pressure systems (Nash & Johnson 1996).

The frequency of drought has been identified as an important factor controlling occurrence of forest fire (Girardin et al. 2004a; Girardin et al. 2006b). Persistent drought influences fire behaviour by influencing fuel moisture content through the effects on precipitation frequency, relative humidity, air temperature, wind speed, and lightning (Flannigan & Harrington 1988; Flannigan & Van Wagner 1991). Short droughts of up to one month can have significant effects on regions with drier climates, while longer duration droughts are normally required to affect wetter regions (Beverly & Martell 2005). A prolonged moisture level deficit of a few days to several weeks is often enough to result in increased ignitions over a short time (Johnson & Wowchuk 1993), whereas an increase in annual average area burned is attributed to longer droughts (weeks to months) when fuel moisture over large areas is affected (Flannigan & Harrington 1988).

Fuels and Vegetation: local and broad-scale determinant of the fire behaviour

After initiation, a spreading forest fire is a complex combustion process in which the flaming front ignites unburned woody and herbaceous fuels. Fuels in the boreal forest include all fine and coarse woody debris as well as live grasses, brush, small trees, shrubs, low branches and limbs. The fuel in a region influences the fire regime in multiple ways, specifically fuel load determines the maximum energy available to a fire (intensity); the arrangement of fuel load can affect vertical (e.g. presence of ladder fuels) and horizontal spread (type); finally the size distribution of the fuel load can affect the likelihood of initial ignition and rate of spread (occurrence and size). Fire intensity and fire type vary according to differences in the distribution and arrangement of fuels in stands with different fire histories and with different species compositions. For example, sites with a recent history of fire will be unable to accumulate adequate fuel to support an intense fire, and stands dominated by coniferous vegetation are more susceptible to crown fires than deciduous forests (Cumming 2001). The fuel arrangement in coniferous stands, low dead branches covered in lichens on the trunks of black spruce for example, provide a ladder into the canopy (Heinselman 1981; Vierick 1983), while crown fires that enter into a deciduous canopy generally drop to the ground, and continue to burn as surface fires (Cumming 2001). Of note is the two-way interaction between forest fire and the spatial pattern of vegetation in the landscape. The spread and behaviour of fire is constrained by spatial patterns of vegetation, terrain, and fuel availability, at the same time fire creates spatial patterns of vegetation thereby influencing future occurrence of fires.

Crown fire intensity is rarely uniform; varying intensities within large crown fires create a complex mosaic of burn severity across the landscape. The varying intensities present within crown fires are due to complex interactions of wind variation, topography, vegetation, and natural fire breaks. The spatial heterogeneity of burn severity patterns creates a wide range of local effects that influence successional trajectories such as plant reestablishment and other ecological processes (Turner & Romme 1994). Similarly, not all boreal species have the same resistance to fire or the same recuperative capacity after being burnt. Species responses to fire depend on the life cycle of the tree, the fire regime to which they are subject, and the local post-fire environment. Boreal species survive fire by a variety of means: thick insulating bark, serotinous cones, and belowground systems with dormant buds. Some of the more common boreal species and their adaptations to fire are summarized in Table 2.1.

Table 2.1. Fire response life-history traits of common boreal tree species

(Adapted from, Frank 1990; Nienstaedt & Zasada 1990; Perala 1990; Safford et al. 1990; Viereck & Johnson 1990; Chen & Popadiouk 2002)

| **Species** | **Common Name** | **Presence in post-fire cohort** | **Post-fire reproduction mode** | **Post-fire regeneration speed** | **Shade tolerance** | **Fire resistance** |
|---|---|---|---|---|---|---|
| *Populus tremuloides* | Trembling Aspen | Y | Root suckers | Rapid | Intolerant | low |
| *Betula papyrifera* | Paper Birch | Y | Stump sprouts, seeds | Rapid | intolerant | low |
| *Pinus banksiana* | Jack Pine | Y | Seeds from serotinous cones | Rapid or gradual | Very intolerant | high |
| *Picea mariana* | Black Spruce | Y | Seeds from serotinous cones | Rapid or gradual | Tolerant | medium |
| *Picea glauca* | White Spruce | N | Seeds | Gradual or slow | Very-tolerant | low |
| *Abies balsamea* | Balsam Fir | N | Seeds | Gradual or slow | Very-tolerant | low |

Physiographic features: local-scale determinants of fire behaviour

Although fire regime is largely controlled by climate, landscape features have been shown to influence the regional occurrence and behaviour of boreal forest fires. Topographic and physiographic features in the landscape can influence the local probabilities of initial ignition and burning. The aspect of a slope influences fuel moisture and therefore ignition probabilities, as north and east-facing slopes receive less solar radiation than south and west-facing slopes and therefore tend to be cooler and moister with a lower probabilities of ignition (Barrett 1988). The presence of and distance to natural fuel breaks such as, lakes, rivers, or wetlands in the landscape can not only influence local probabilities of burning due to fuel moisture content, but fragment the continuous forest cover thereby impeding fire spread (Heinselman 1973; Romme & Knight 1981). It is important to note that these landscape features have little influence on local fire behaviour when burning conditions are extreme. Under conditions of severe fire weather like prolonged drought, all fuels across the landscape become highly susceptible to burning and may render the occurrence of large stand-replacing crown fires inevitable despite the presence of physiographic features that might typically impede fire spread under less severe conditions (Fryer & Johnson 1988).

## FIRE FREQUENCY IN THE NORTH AMERICAN BOREAL FOREST

Historical fire frequencies of multiple regions across the boreal forest have been reconstructed over the past 35 years. The methodologies and tools necessary to obtain accurate estimates of time-since-fire (TSF) distributions are widespread and well situated within probability theory and statistics. Trees that survive fires often scar, which allow a reconstruction of the fire history at that location. Additionally, increment cores taken from even-aged canopy trees of pioneer species can be used to estimate fire history by using the age of the cohort as an approximate time since last stand replacing fire (Johnson & Gutsell 1994). For much of the boreal forest fire history reconstructions are limited by the longevity of the tree species and fire frequency (new fires erase evidence of old fires); accurate reconstructions greater than 300 years are rare. TSF distributions are typically calculated from time-since-fire maps consisting of spatially distinct patches within a landscape identified by the year in which they last burned. The

TSF distribution is a survivorship curve that gives the proportion of a region that has survived without a fire to time *t*. Through the use of likelihood estimators, calculations of the burn rate can be derived and used to reconstruct historical fire frequencies from TSF maps and dendroecological sampling (Reed et al. 1998).

Our understanding of the fire frequencies that characterize the boreal forest is incomplete. Detailed forest fire statistics have been archived since 1920 for most of Canada which allow an analysis of fire activity within the boreal forest; however it is recognized that the fire record prior to the early 1970s is incomplete, as remote locations in the boreal forest were not consistently monitored prior to the advent of satellite coverage at that time. Although many fire history studies have been completed in the boreal forest, there are many regions for which fire history is unknown, and considering the well-known variation with the boreal forest, attempting to generalize from regional studies to the entire boreal region is inappropriate. A few general trends in boreal fire frequencies and age class distributions can be observed across longitudinal and latitudinal gradients, Table 2.2 summarizes most fire frequency studies conducted in boreal North America.

The boreal forest burns quite frequently, within the past 200-300 years virtually all of the boreal forest has experienced major fire events, and assuming that the majority of the forest follows a negative exponential distribution (Johnson et al. 1995), possibilities for patches of old-growth forest (>200 years) existing within the landscape is limited to a mere 5-10 %.

Figure 2.2. Location of fire frequency studies in the North American boreal forest

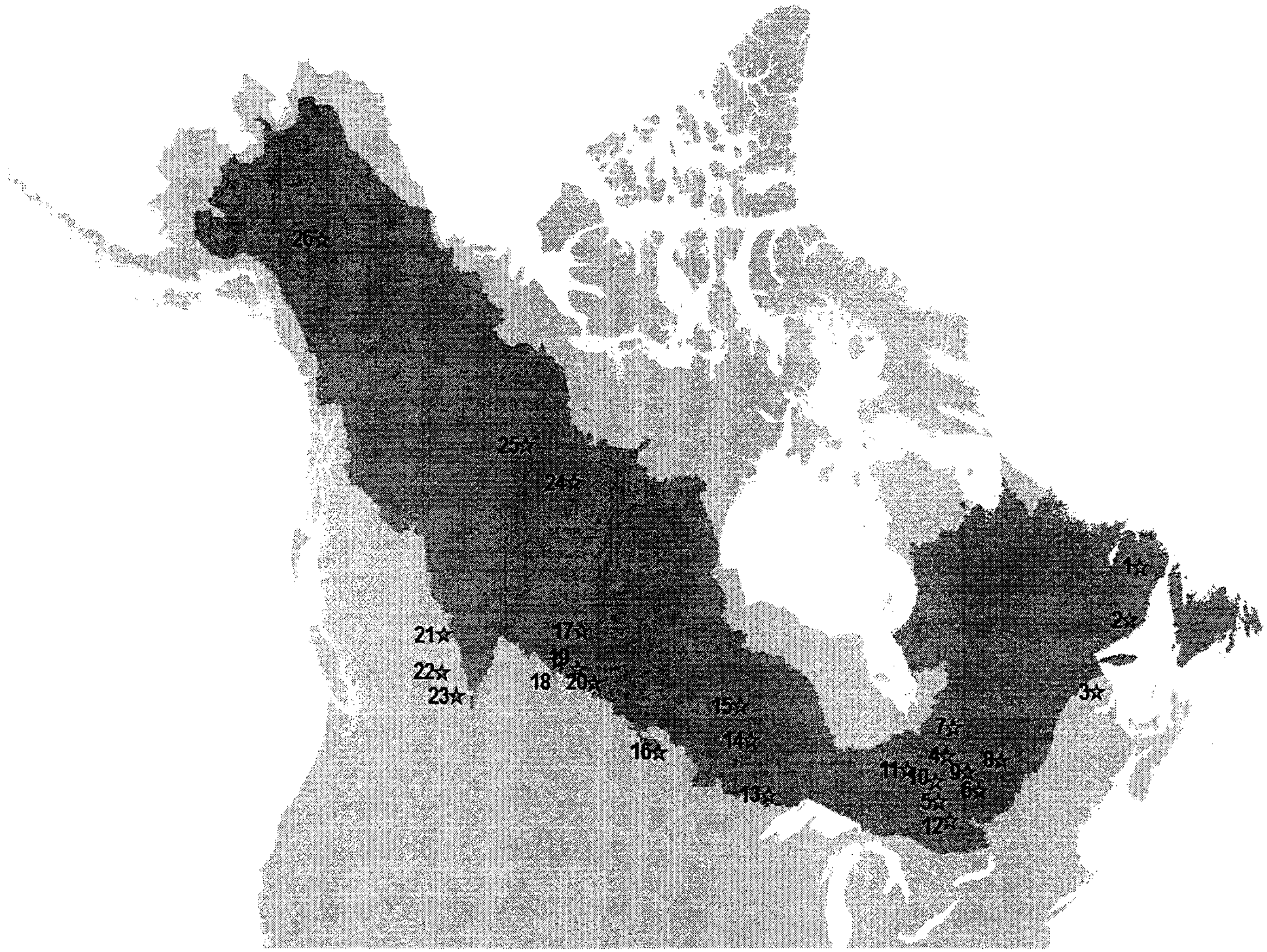


Table 2.2. Fire frequency studies in the boreal forest from east to west to north

| Study | Location | Fire Cycle (years) | Time Period | Probability Distribution | Map Label |
|---|---|---|---|---|---|
| Foster (1983) | Southeastern Labrador, Canada | ~ 500 | < 1983 | Negative exponential | 1 |
| Cyr *et al.* (2007) | North Shore region, Eastern Quebec, Canada | 191 | 1896-2006 | Cox Proportional Hazard | 2 |
| Lauzon *et al.* (2007) | Gaspesie, southeastern Quebec, Canada | 116 | 1920-2003 | Negative exponential | 3 |
| Lauzon *et al.* (2007) | Gaspesie, southeastern Quebec, Canada | 89 | Pre-1850 | Negative exponential | 3 |
| Lauzon *et al.* (2007) | Gaspesie, southeastern Quebec, Canada | 176 | Post-1850 | Negative exponential | 3 |
| Le Goff *et al.* (2007) | Waswanipi, north-central Quebec, Canada | 99 | 1850-1940 | Negative exponential | 4 |
| Le Goff *et al.* (2007) | Waswanipi, north-central Quebec, Canada | 283 | 1940-2000 | Negative exponential | 4 |
| Grenier *et al.* (2005) | Timiskaming region, Southwestern Quebec, Canada | 96 (73-126) | 1890-1948 | Negative exponential | 5 |
| Grenier *et al.* (2005) | Timiskaming region, Southwestern Quebec, Canada | 262 (163-422) | Pre-1890 | Negative exponential | 5 |
| Grenier *et al.* (2005) | Timiskaming region, Southwestern Quebec, Canada | 117 (91-150) | 1850-1948 | Negative exponential | 5 |
| Grenier *et al.* (2005) | Timiskaming region, Southwestern Quebec, Canada | 270 (141-519) | Pre-1850 | Negative exponential | 5 |
| Lesieur *et al.* (2002) | Upper Mauricie region, south-central Quebec, Canada | 147 (116-187) | 1923-1998 | Negative exponential | 6 |
| Lesieur *et al.* (2002) | Upper Mauricie region, south-central Quebec, Canada | 82 (61-111) | Pre-1850 | Negative exponential | 6 |
| Lesieur *et al.* (2002) | Upper Mauricie region, south-central Quebec, Canada | 176 (143-216) | Post-1850 | Negative exponential | 6 |
| Lefort *et al.* (2004) | Western Quebec, Canada | 500 > | 1945-1998 | n/a | 7 |
| Bergeron *et al.* (2001) | Central Quebec, Canada | 273 (183-408) | 1920-1999 | Negative exponential | 8 |
| Bergeron *et al.* (2001) | Central Quebec, Canada | 123 (83-181) | 1850-1920 | Negative exponential | 8 |
| Bergeron *et al.* (2001) | Central Quebec, Canada | 69 (47-102) | Pre-1850 | Negative exponential | 8 |
| Bergeron *et al.* (2001) | Abitibi east, east-central Quebec, Canada | 191 (124-294) | 1920-1999 | Negative exponential | 9 |
| Bergeron *et al.* (2001) | Abitibi east, east-central Quebec, Canada | 86 (56-131) | 1850-1920 | Negative exponential | 9 |
| Bergeron *et al.* (2001) | Abitibi west, western Quebec, Canada | 325 (248-424) | 1920-1999 | Negative exponential | 10 |
| Bergeron *et al.* (2001) | Abitibi west, western Quebec, Canada | 146 (114-187) | 1850-1920 | Negative exponential | 10 |
| Bergeron *et al.* (2001) | Abitibi west, western Quebec, Canada | 132 (98-178) | Pre-1850 | Negative exponential | 10 |
| Bergeron *et al.* (2001) | Lake Abitibi model forest, eastern Ontario, Canada | 521 (370-733) | 1920-1999 | Negative exponential | 11 |
| Bergeron *et al.* (2001) | Lake Abitibi model forest, eastern Ontario, Canada | 234 (171-321) | 1850-1920 | Negative exponential | 11 |
| Bergeron *et al.* (2001) | Lake Abitibi model forest, eastern Ontario, Canada | 132 (98-178) | Pre-1850 | Negative exponential | 11 |

| | | | | | |
|---|---|---|---|---|---|
| Cwynar (1977) | Algonquin Park, Ontario, Canada | 70 | 1939-1974 | n/a | 12 |
| Woods and Day (1977) | Quetico Provincial Park, Ontario, Canada | 200 | 1920-1977 | n/a | 13 |
| Woods and Day (1977) | Quetico Provincial Park, Ontario, Canada | 78 | 1850-1920 | n/a | 13 |
| Suffling *et al.* (1982) | Northwestern Ontario, Canada | 100 | 1920-1982 | Negative exponential | 14 |
| Suffling *et al.* (1982) | Northwestern Ontario, Canada | 50 | 1850-1920 | Negative exponential | 14 |
| Bridge (2001) | Northwestern Ontario, Canada | 300-500 | 1921-1995 | n/a | 14 |
| Bridge (2001) | Western Ontario, Canada | 100-300 | 1921-1995 | n/a | 15 |
| Tardif (2004) | Duck Mountain, Manitoba, Canada | 486 | 1914-2001 | n/a | 16 |
| Weir *et al.* (2000) | Northern partition, Prince Albert National Park, Saskatchewan, Canada | 15 (10-35) | Pre-1890 | Negative exponential | 17 |
| Weir *et al.* (2000) | Northern partition, Prince Albert National Park, Saskatchewan, Canada | 75 (45-150) | 1890-1945 | Negative exponential | 17 |
| Weir *et al.* (2000) | Northern partition, Prince Albert National Park, Saskatchewan, Canada | 1, 745 (285-127, 225) | Post-1945 | Negative exponential | 17 |
| Weir *et al.* (2000) | Southern partition, Prince Albert National Park, Saskatchewan, Canada | 25 (15-40) | 1890-1945 | Negative exponential | 17 |
| Weir *et al.* (2000) | Southern partition, Prince Albert National Park, Saskatchewan, Canada | 645 (200-4, 270) | Post-1945 | Negative exponential | 18 |
| Parisien *et al.* (2004) | Boreal Plain Ecozone, Saskatchewan, Canada | 263 | 1981-2000 | n/a | 18 |
| Parisien *et al.* (2004) | Boreal Plain Ecozone, Saskatchewan, Canada | 288 | 1981-2001 | n/a | 19 |
| Parisien *et al.* (2004) | Boreal Shield Ecozone, Saskatchewan, Canada | 99 | 1981-2002 | n/a | 19 |
| Parisien *et al.* (2004) | Boreal Shield Ecozone, Saskatchewan, Canada | 104 | 1981-2003 | n/a | 20 |
| Parisien *et al.* (2004) | Taiga Shield Ecozone, Saskatchewan, Canada | 114 | 1981-2004 | n/a | 20 |
| Parisien *et al.* (2004) | Taiga Shield Ecozone, Saskatchewan, Canada | 112 | 1981-2005 | n/a | 21 |
| Tande (1979) | Jasper National Park, Alberta, Canada | 74* | 1665-1975 | n/a | 22 |
| Johnson *et al.* (1990) | Glacier National Park, British Columbia, Canada | 80 | 1520-1760 | Negative exponential | 22 |
| Johnson *et al.* (1990) | Glacier National Park, British Columbia, Canada | 110 | 1760-1988 | Negative exponential | 23 |
| Masters (1990) | Kootenay National Park, British Columbia, Canada | > 2700 | 1928-1988 | Negative exponential | 23 |
| Masters (1990) | Kootenay National Park, British Columbia, Canada | 130 | 1788-1928 | Negative exponential | 23 |
| Masters (1990) | Kootenay National Park, British Columbia, Canada | 60 | 1508-1778 | Negative exponential | 23 |
| Johnson (1979) | Northwest Territories, Canada | 37-102 | n.c. | Weibull | 24 |
| Rowe *et al.* (1974) | Mackenzie River Valley, Northwest Territories, Canada | 100 | n.c. | Weibull | 25 |
| Yarie (1981) | Alaska, U.S.A. | 43 | n.c. | Weibull | 26 |

n.c.: Not calculated, insufficient information in the published paper

n/a.: different methods that do not use a probability distribution were used to calculate the fire cycle (often 'burning rate' method)

Among fire frequency studies whether or not the probability of burning increases with stand age is a source of some disagreement. In general, the time-since-fire distributions have been found to best fit a negative exponential or Weibull distribution (Johnson & Van Wagner 1985). If flammability increases with age, the TSF distribution can be found to fit best the Weibull distribution, in which the probability of burning is a power function of time-since-fire. If the probability of burning is independent of the stand age, the fire cycle can be described by the negative exponential distribution (Van Wagner 1978; Johnson & Van Wagner 1985). In real-world scenarios, the assumption of a constant rate is rarely satisfied, however the majority of regional fire frequency reconstructions have been found to 'best fit' the negative exponential model (Weir et al. 2000; Bergeron et al. 2001). The justification for using the negative exponential is that after a relatively short period of low flammability following fire (about 15-20 years, or until crown closure), the boreal forest has an equal burn probability regardless of age (Rowe & Scotter 1973), owing to the relative importance of weather and climate over fuels (Bessie and Johnson 1995).

In studies able to reconstruct the historical fire frequency for a sufficient length of time (typically 200-300 years) major temporal changes in fire cycle have been detected. The most common change detected occurs between the end of the Little Ice Age (LIA) c.1850 and early 20$^{th}$ century (1900-1920) with the majority of studies observing a shift from a shorter fire cycle in the past to a longer cycle in recent decades despite climate warming since the end of the LIA. The gradient of documented fire frequencies is partly the result of regional climate and vegetation type (Payette et al. 1989), as such the change in fire frequency is attributed to a combination of climate changes and the onset of Euro-colonization, including landscape fragmentation, timber harvesting, and fire suppression, which directly affect fire frequency.

Fire frequencies vary in the boreal forest from east to west according to climate. Studies conducted in eastern Quebec (Gaspésian peninsula) and southeastern Labrador have found that their respective maritime and subarctic climates limit the spread of fire resulting is long fire cycles (Foster 1983; Lauzon et al. 2007). Fire cycles present in central Canada (Ontario and Quebec) are highly variable from region to region, although a decrease in fire frequency can be observed from central Quebec to eastern Ontario; fire frequency then increases from eastern to Northwestern Ontario (Cwynar 1977; Woods & Day 1977; Suffling & Molin 1982; Bergeron et al. 2001; Bridge 2001; Lesieur et al. 2002; Lefort et al. 2004; Grenier et al. 2005; Cyr et al. 2007; Le Goff et al. 2007). Fire cycles in the boreal forest of western Canada's prairies provinces (Manitoba, Saskatchewan, Alberta) show widest range of variability from a low of 15 years to a high of 1, 745 years among different time periods (Tande 1979; Weir et al. 2000; Parisien et al. 2004; Tardif 2004). The western and northern boreal forests of British Columbia, Northwest Territories, and the state of Alaska, have relatively short fire cycles ranging between 37 and 110 years. These humid mountainous and cold arctic climates have low rates of fire occurrence, but fires are typically more destructive when they do occur, burning hundreds of thousands of hectares (Masters 1990; Bessie & Johnson 1995). It is interesting to note that studies conducted in national and provincial parks (Johnson et al. 1990; Masters 1990; Weir et al. 2000; Tardif 2004), have observed a significant lengthening of the fire cycle beginning early to mid 19$^{th}$ century attributed to human activity, specifically landscape fragmentation and aggressive fire suppression policies.

The wildfire situation in boreal Alaska is similar to that of the eastern Canadian boreal, forest fire statistics in Alaska are available for the past sixty years, and generally indicate that although fire occurrence has increased, there has been a small decreasing trend in annual area burned. The changes observed in Alaska are attributed to increased accessibility, while road and rail access increased human forest usage resulting in increased fire occurrences, enhanced fire detection capabilities decreased fire suppression response time resulting in lower fire frequencies (Stocks et al. 2001).

Fire frequency studies following the statistically rigorous techniques established in the literature (Johnson and Gutsell 1994, Reed et al. 1998) are limited by the availability of good quality data. They require accurate maps that show the date of the time since last fire for each stand within the study area, which is time-consuming, expensive, and difficult to collect; as such there are many regions in the North American boreal forest in which historical fire frequency is unknown particularly at high latitudes within the central and western boreal shield. Considering the limitations of dendroecological reconstructions, new methodologies for reconstructing fire history are necessary to understand the legacy of past fires in present ecosystems, as well as the role of fire with projected climate changes as a result of increased $CO_2$ emissions in the future.

Changes in the Fire Cycle in response to human activities

The effects of detected temporal changes in fire frequency attributed to human activities are mixed. Landscape fragmentation caused by the removal of timber from continuous forest has been found to significantly decrease fire frequencies in central Saskatchewan boreal forest (Weir & Johnson 1998; Weir et al. 2000). In contrast, in Quebec colonization and its associated logging and agricultural activities appear to have contributed to spatially mixed fire frequencies, with higher fire frequencies in areas adjacent to development (Lesieur et al. 2002; Lefort et al. 2004; Grenier et al. 2005). The change in fire frequency may be due to increased landscape access through the colonization process (e.g. roads), creating conditions for additional ignitions from human activity in regions traditionally dominated by infrequent lightning strikes (Cwynar 1977; Weber & Stocks 1998; Lefort et al. 2003). These studies reiterate the need for caution when extrapolating results from an individual study to make inferences over an entire geographic regions or even the entire boreal forest.

The effectiveness of fire suppression in the boreal forest is hotly contested. Public fire management agencies expend great effort to suppress boreal forest fires, particularly when communities, infrastructure, or commercially valuable forests are endangered; because of this the belief that fire suppression has substantially reduced the average annual area burned is often thought to be self-evident. Studies have argued that this general presumption of effectiveness lacks adequate empirical support in the scientific literature (Johnson et al. 2001; Miyanishi et al. 2002). The impact that effective fire suppression may have on average annual area burned is important because inferences made from fire frequency studies implemented in sustainable forest management practices are key determinants of allowable cut harvest limits. When operating under the assumption that forest harvesting supplants wildfire as the dominant disturbance, provided that fire suppression is effective, harvesting can occur at rates close to the natural fire cycle without increasing the overall rate of disturbance. If fire suppression is not effective, fire and human induced forest loss is occurring at rates outside of the natural variability, and as a result forests are not being managed sustainably.

Arguments against the effectiveness of fire suppression come primarily in two forms: (1) criticism of data quality and inferences drawn from studies based on provincial (primarily Ontario) fire records (Miyanishi & Johnson 2001; Miyanishi et al. 2002; Bridge et al. 2005), and (2) time-since-fire studies.

The case for effective fire suppression rests on a number of comparisons between boreal forest regions within the province of Ontario, Canada (Stocks 1991; Ward & Tithecott 1993; Martell 1994), in which fire frequencies were found to be lower and the occurrence of small fires were higher in regions of aggressive fire suppression, from which the effectiveness is inferred. Arguments suggest that a level of bias exists in that the level of suppression among detected fires is variable, with most operational fire management agencies in Canada prioritizing fires according to perceived values-at-risk. This practice results in numerous fires in low-priority (high latitude) areas being allowed to burn without active suppression and those located adjacent to human settlements aggressively suppressed. This policy has become widely accepted by fire agencies as total fire exclusion is an economic and operational impossibility and it is not ecologically desirable as several boreal species rely on fire for reproduction.

Johnson et al (2001) argues that the data used in comparisons of the average annual area burned between areas with and without aggressive fire suppression policies is biased by the fact that small fires are virtually unreported in remote areas without aggressive suppression policies. Considering that the number of lightning initiated fires is consistent in areas with and without aggressive suppression policies, the smaller average fire size in areas with aggressive policies a likely consequence of increased reporting confounding the regional comparisons.

The numerous time-since-fire studies show a change in fire cycle at the beginning of the $20^{th}$ century which is associated with large-scale climatic factors. Critics cite that given that there has been no observable change in the fire cycle since the completion of most euro-colonization settlements in the boreal forest c.1920, nor an observable change following the implementation of improved suppression policies and the introduction of water bombers c.1950, fire suppression has not been effective in reducing the average annual area burned. The majority of area burned each year is due to a small number of fires occurring under severe weather conditions (Newark 1975; Flannigan & Harrington 1988; Stocks et al. 2002). It has been argued that if suppression cannot affect these few large fires, which their continued occurrence seems to indicate, then it cannot impact the annual average area burned. Recently Cumming (2005), analyzed the operational effectiveness of the practice of initial attack (IA), whose purpose is to limit large fires, in boreal northeastern Alberta, Canada. The conclusion from that study is that fire suppression by IA has had an important impact on area burned in recent decades, and that in the absence of IA fire suppression efforts, average annual area burned would increase. It is likely that the effectiveness of fire suppression is as variable as the fire regime itself, with regional variations arising from various provincial and state fire management policies and resources.

A conclusion we can draw from these fire frequency and fire suppression studies is that fire cycles have changed multiple times in the past 300 years in response to climate and human activities. The concept of a `natural` fire cycle for most of the boreal forest no longer exists, as even at high latitudes where regions are generally unpopulated and fires are allowed to run their course, they still reflect climatic variability of which humans may bear some responsibility in recent decades. Boreal fire cycles are dynamic, carrying with them the memory of past cycles and influencing future cycles (Johnson et al. 1998).

## CLIMATE CHANGE AND FOREST FIRES

There is a general consensus that human activities within the past century are responsible for recent changes in climate (IPCC 2007b). Increases in radiatively active gases such as carbon dioxide, methane, and chlorofluorocarbons in the atmosphere are causing a significant warming of the earth`s surface. Potentially, climatic driven changes in boreal fire regimes may lead to substantial landscape changes, in terms of both the characteristic vegetation mosaic and species composition. General agreement exists that under current climate change scenarios, multiple aspects of the boreal fire regime, specifically fire frequency and area burned are expected to increase (Weber & Flannigan 1997; Flannigan et al. 1998). These increases are likely to be non-uniform; significant variability is expected due to the complex interactions between climate and various spatial and temporal factors driving regional fire regimes (Flannigan et al. 1998).

Many studies investigating the impact of climate change on the forest fire season in North American boreal forests use simulation results from general circulation models

(GCMs). These GCMs are used to build climate change scenarios by predicting increases in surface air temperatures across the boreal forest region in response to increasing concentrations of greenhouse gases (mostly $CO_2$) and other pollutants within the atmosphere. GCMs provide the best means available to estimate the impact of future climate changes on the fire regime at a broad scale (Flannigan et al. 2005a). In general, most models predict warming at high latitudes during winter months across the boreal forest, with many models suggesting an increased moisture deficit in the summer months in the central boreal region. It is worth nothing that the uncertainty surrounding climate change influencing the incidence of extreme dry events could have a significant impact on boreal fire activity as large fires are highly correlated with extreme fire weather (Flannigan et al. 2001).

Several studies (Flannigan & Van Wagner 1991; Wotton & Flannigan 1993; Stocks et al. 1998), have also used GCMs to calculate elements of the Canadian Forest Fire Weather Index (FWI) (Van Wagner 1987), under scenarios that typically use $CO_2$ levels that are double or triple those of the present (Bergeron & Flannigan 1995; Flannigan et al. 2005a). The FWI is a weather-based system that models the effect of fuel moisture and wind on fire behaviour. Results generally show increasingly severe fire weather across most of the western boreal forest, and increases in the occurrence of extreme fire danger (Timoney 2003). For example, the forests of Alberta under a $2 \times CO_2$ scenario (relative to a baseline $1 \times CO_2$) show a projected 12.9% increase in area burned by 2050; under a $3 \times CO_2$ scenario (year 2080) area burned is projected to increase 29.4% (Tymstra et al. 2007). Some studies, in stark contrast to suggesting that climate change will result in increases boreal fire activity, suggest that regional fire weather severity in the eastern Canadian boreal forests will remain unchanged or even experience a decrease under 2 × $CO_2$ scenarios by AD 2100 (Bergeron & Flannigan 1995; Flannigan et al. 2001). The modeled FWI anomalies suggest that despite warming, less severe fire weather is expected over much of eastern Canada; the decrease being attributed to increases in precipitation and humidity (Flannigan et al. 1998; Flannigan et al. 2001).

The numerous studies documenting the likely impacts of climate change on fire disturbance in boreal North America reveal that the potential changes are spatially dependent, and, for some areas the projections are not in agreement (Flannigan et al. 2005a; Girardin & Mudelsee 2008). Likely reasons for the disagreement among studies include the $CO_2$ emission scenario selected, the models selected (empirical vs. process-based), the predictor variable sources, ignition agents (lightning vs. human-caused), the area and scale studied, and the time period(s) for which the simulations are prepared and compared. In the case of eastern Canada, projections range from decreasing fire activity (Bergeron & Flannigan 1995; Bergeron et al. 2004a) to large increases in area burned (Flannigan et al. 2005b), depending on the $CO_2$ scenario used (2 × $CO_2$ vs. 3 × $CO_2$), the source of predictor variable sources (monthly anomaly vs. daily data), and the period with which comparisons are made.

Future climate change could have significant repercussions for lightning and human-caused fires. Lightning-initiated and human-initiated fires are expected to increase in response to climate warming (Price & Rind 1994; Wotton et al. 2003). Using a 2 × $CO_2$ GCM, a 30% increase in global cloud-to-ground lightning activity is predicted, with a 72% increase over continental regions (Price and Rind 1994).

Krawchuk et al. (2009), predicted lightning-initiated fire under 1 × $CO_2$ (1975-1985), 2 × $CO_2$ (2040-2049), and 3 × $CO_2$ (2080-2089) conditions simulated by the Canadian Regional Climate Model (CRCM) and found a 1.5-fold and a 1.8-fold increase of lightning-initiated fires by 2040-2049 and 2080-2089 relative to the 1975-1985 conditions due to changes in fire weather. These results imply that as global climate warms changes in moisture balance and thunderstorm activity will occur in the North American boreal forest; there is some evidence that lightning-initiated fires in the boreal forest are currently occurring with greater frequency (Stocks 1991). Increases in human-initiated fires are also predicted in response to climate change. In Ontario, the total number of human-initiated fires could increase by approximately 18% by 2020-2040 and 50% by the end of the 21st century (Wotton et al. 2003).

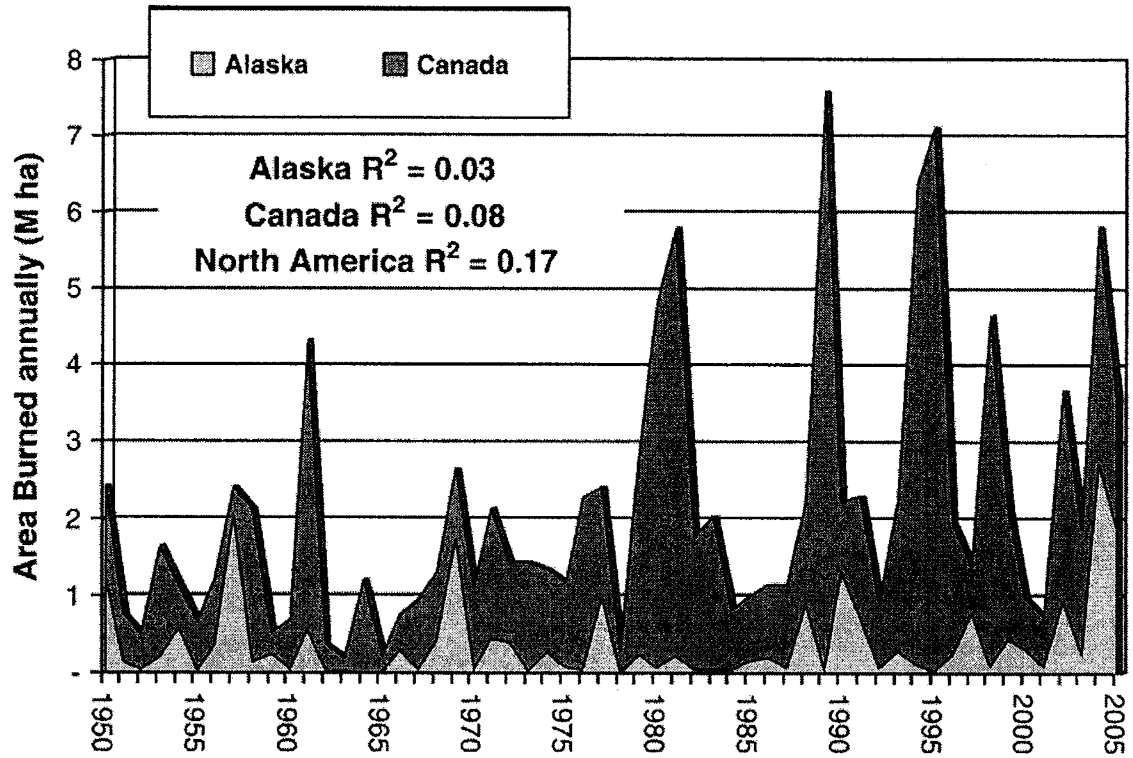



Figure 2.3. Area burned annually across North America reported in millions of hectares. Area burned is shown separately for Canada and Alaska and the cumulative area burned for North America is also shown. Note the recent increase in extreme fire years. (From Soja et al. 2007)

To understand if global climate change will increase the area burned in the boreal forest, it is essential that we understand the linkages between climate and vegetation to area burned. Current climate change projections (IPCC 2007a), along with recent assessments of climate change impacts on forest fire activity (Flannigan et al. 2005a; Soja et al. 2007) indicate that increases in fire weather severity is resulting in larger areas experiencing extreme fire danger in the boreal forest. The area burned by fire in the boreal forest has shown a pronounced upward trend over the past three decades which is consistent with predictions that area burned and fire season length will increase as global temperatures rise (Stocks et al. 2002; Podur et al. 2002; Gillett et al. 2004). Increases in temperatures at boreal latitudes alone do not explain increases in area burned, as stated earlier, studies on fire frequency in the eastern boreal forests of Quebec show that, despite significant increases in temperature since the end of the Little Ice Age (approx 1850), the frequency of fires has in fact decreased (Flannigan et al. 1998; Lesieur et al. 2002; Le Goff et al. 2007). Similarly, the increasing trend in area burned over recent decades (Podur et al. 2002; Gillett et al. 2004; Kasischke & Turetsky 2006) is not correlated with any trend in fire weather severity (Amiro et al. 2004; Girardin et al. 2004a). At the very least, increasing temperatures in the past 150 years have not uniformly altered fire weather severity, fire frequency, and average annual area burned across the boreal forest, and as a result we should expect large regional variations in future fire activity due to climate change.

Gillet et al. (2004) predicts higher temperatures and increased evapotranspiration over North America will result in changes in the length of time needed to effectively dry fuel. Active fire weather correlates strongly with atmospheric circulation particularly the prolonged blocking high pressure systems (Skinner et al. 1999). These strong midtroposphere anomalies may result in severe extended drought events, which in turn can lead to increased fire activity (Fauria & Johnson 2006; Xiao & Zhuang 2007). Similarly, Girardin et al (2006a) using correlation analyses with regional fire statistics showed that drought estimates are accurate enough to approximate fire activity and, hence, the estimates are relevant for the study of climate change impacts on Canadian boreal forests. Future fire regimes in the boreal forest will likely depend on drought patterns under current global climate change scenarios.

Climate change indirectly influencing vegetation composition through the fire regime may be more important than the direct effects of climate change on species distribution, migration, substitution, and extirpation (Weber & Flannigan 1997). If predictions of increased forest fire activity occur as a result of a trend toward warmer and drier conditions, we can expect a shift in the age-class distribution towards younger forests as a result of a shorter fire cycle (Stocks 1993; Kasischke et al. 1995). Additionally the species composition of the boreal forest will shift towards a greater abundance of shade-intolerant pioneer species that colonize rapidly post-fire, in particular aspen and birch as fire promotes these species (Chen et al. 2009; Ilisson & Chen 2009a; Ilisson & Chen 2009b).

In addition to the local effects of climate change on boreal forest ecosystem structure it has been suggested (Kurz & Apps 1995) that increasing boreal fire activity and accompanying species composition shifts will create a positive feedback loop, with increasing amounts of $CO_2$ being released into the atmosphere exacerbating global warming. Altered fire regimes are liberating carbon from boreal vegetation and substrates at rates exceeding accumulation; a retrospective analysis of carbon fluxes (Kurz & Apps 1999) found that in recent decades the Canadian forest sector has become a net source of atmospheric carbon due to an increase in natural disturbance. Suggested efforts to influence the global carbon balance through boreal forest management such as: increasing harvest rotation lengths, reducing regeneration delay, or increasing stocking densities, may be overwhelmed by the predicted changes in future fire frequency (Kurz et al. 2008).

Overall evidence of the transformation of landscapes due to changes in climate is mounting throughout the boreal forest of North America. Given the increases in temperature, fire activity (occurrence and frequency), and the increased frequency of severe fire seasons in North American boreal forest regions over the last few decades suggests that the observed landscape changes are in agreement with modeled predictions of fire and climate change. Having identified a host of predictions and observed changes in boreal ecosystems which have potential to feedback to the climate system, I suggest that the maintenance of ecosystem integrity and stability in response to a dynamic and uncertain future should therefore be an essential goal of forest management in the North American boreal forest to prevent a scenario that includes positive biospheric feedback of ongoing climate change.

SUMMARY

1. Fire is a major stand replacing disturbance in the North American boreal forest. Stand structure is largely a reflection of time since fire which has important implications for forest ecosystems including stand species composition, nutrient cycling, and carbon storage

2. Fire regime defines the expected pattern of fire frequency, size, intensity, seasonality, type, and severity expected in a given region. Boreal forest fire regime is variable along longitudinal and latitudinal gradients.

3. Fire regime is driven primarily by climate, which controls the occurrence, timing, frequency, duration, extent, and intensity of fire by altering fuel moisture levels. The few fires responsible for the majority of annual area burned are typically initiated by

lightning strikes when fuels are dry, which are intrinsically linked to broad-scale climatic patterns.

4. In some regions fire activity can be influenced by vegetation, landscape fragmentation, fuel type and physiographic features, though these factors are more typically associated with site-specific fire behaviour. Vegetation can play an important role controlling fuel amount, continuity, moisture, arrangement, and structure. Physiographic features such as slope, aspect, and fire break distance can significantly influence fire behaviour.

5. Fire cycles in the boreal forest range anywhere from 30 years to over 500. The majority of studies link major changes in the fire frequency of the boreal forest to large-scale climatic shifts following the end of the LIA. In general, over the past century, fire frequency has decreased over much of the boreal forest, although in recent decades annual area burned is increasing.

6. Human activities since the early 1920s have changed the fire cycle in some regions of the boreal forest. A general consensus of the impact of human activities on the fire cycle is mixed; landscape fragmentation impedes fire spread from adjacent areas limiting fire size, but increased accessibility has increased human-initiated fires. Similarly, the effectiveness of fire suppression remains contested.

7. The climate of the North American boreal forest is highly variable across its geographic range, with fire cycles varying accordingly. It is unclear how predicted climate change scenarios will affect regional fire cycles, although the general consensus is a predicted increase in fire activity over most of the boreal forest.

# CHAPTER THREE. TEMPORAL AND SPATIAL VARIATIONS IN THE FIRE FREQUENCY OF MIXEDWOOD FORESTS IN THE CENTRAL BOREAL SHIELD

## INTRODUCTION

Emulating the landscape patterns and ecological effects of natural disturbances is considered central to sustainable boreal forest management (Hunter 1999). In the North American boreal forest, fire is the most dominant natural disturbance (Johnson 1992; Stocks 1993), playing a key role in maintaining biodiversity and essential forest ecosystem functions (Johnson et al. 1995; Bergeron et al. 2002). The fire regime, as defined by several variables including frequency, size, intensity, seasonality, fire type and severity (Rowe & Scotter 1973; Weber & Flannigan 1997), is a key factor affecting age-class distribution, stand composition and structure in the boreal forest (Bergeron & Dansereau 1993; Weber & Stocks 1998). An improved understanding of the natural fire regime is essential to implement sustainable management strategies in the boreal forest (Rowe & Scotter 1973).

Variation in the boreal fire regime has been ascribed to temporal factors including changes in temperature and precipitation generated from climatic warming (Bergeron 1991; Weir et al. 2000; Bridge 2001), and spatial factors such as latitude, longitude, or aspect (Payette et al. 1989; Lefort et al. 2004; Cyr et al. 2007). Broad-scale climatic conditions created by upper level longwave ridging at boreal latitudes (Skinner et al. 1999; Skinner et al. 2002), and human activities (colonization, fire suppression, and harvesting) are considered to have the greatest influence on the fire regime (Johnson 1992; Stocks 1993; Le Goff et al. 2007), predominantly in relation to fire occurrence, and fire frequency.

In addition to broad scale temporal factors, the influence of local physiographic features on regional fire regimes are well documented (Johnson et al. 1999; Weir et al. 2000; Kasischke & Turetsky 2006). Fire regime components are affected by local environmental conditions such as soil and vegetation types, number of water bodies (fuel breaks), and topography (Rowe & Scotter 1973; Bergeron & Dansereau 1993; Krawchuk et al. 2006; Cyr et al. 2007), which operates at a smaller scale, creating variations in microclimate and fuel load (Barton et al. 2001; Brown et al. 2001). For example, changes in aspect and slope have been shown to drive spatial variation by influencing fuel moisture and humidity via solar energy reception, thereby influencing the local probability of fire ignition and spread (Rowe & Scotter 1973; Whelan 1995).

Multiple fire regimes and historical fire frequencies in both the eastern (Bergeron et al. 2006) and western (Johnson et al. 1998; Weir et al. 2000) boreal forest regions of North America have recently been reconstructed. The central boreal shield region represents the transition between the eastern and western boreal forests and contains one of the highest plant diversities in the North American boreal forest (Qian et al. 1998). Despite the region's relative importance ecologically and economically, little information is known regarding the temporal and spatial factors influencing fire frequency. In this study, we sought to understand how the fire frequency of a large mixedwood forest in the central boreal shield varies as a result of broad-scale temporal factors (climate and human manipulation), and physiographic features. We hypothesized that the fire frequency of the central boreal shield should be intermediate to those documented in the eastern and western boreal regions because of its intermediate climate and associated vegetation. Our objectives were two-fold: (i) to reconstruct and describe the fire frequency of a large cross section of the central boreal shield region; (ii) to assess how the (mean) fire frequency is affected by temporal and spatial factors.

## DATA AND METHODOLOGY

### Study Area

The study was conducted north of Thunder Bay, Ontario in mixedwood boreal forest. The study area (49-50°N; 88.5-90.5°W) lies in the northwestern commercial forests of Ontario and covers approximately 11, 600 km$^2$ (Figure 3.1).

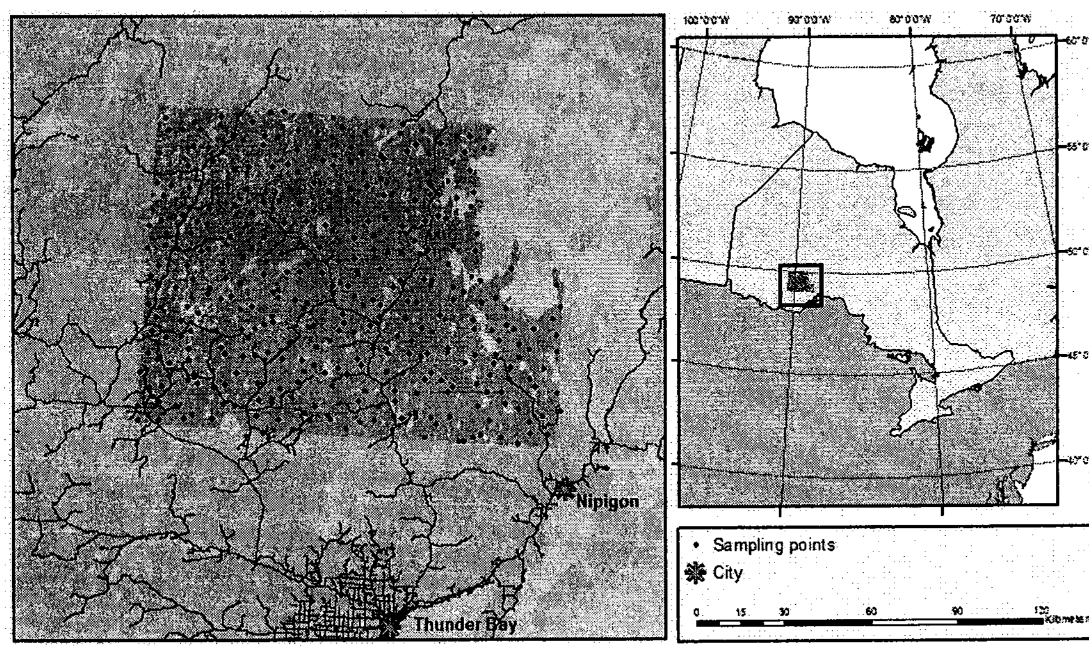



Figure 3.1. Location of the study area within the central boreal shield of Ontario

The study area is located within Ontario's Crown forest, which is divided into geographic planning areas, known as forest management units, for the purposes of forest management. The study area crosses the administrative boundaries of the Black Sturgeon, Dog River-Matawin, English River, Lakehead, and Spruce River Forests (appendix 1), all currently managed by Abitibi-Bowater under a Sustainable Forest Licence (SFL).

The area is within the Moist Mid-Boreal (MBx) ecoclimatic region (Ecoregions Working Group 1989), characterized by a strong continental climate with long cold winters and short hot summers. The mean summer temperatures within the Lake Nipigon ecoregion is typically 14°C (ranging from 11°C - 16°C), with the mean winter temperature at -13°C. Mean annual precipitation ranges from 700-800 mm. The area is subject to relatively cool summer, and warm winter temperatures, due to the moderating lake effect of Lake superior and Lake Nipigon (Ecoregions Working Group 1989). Regional climate is heavily influenced by large air masses in the upper atmosphere. These persistent blocking systems are prone to causing air subsidence, resulting in long stretches of sunny warm days influencing the frequency of dry events (Bessie & Johnson 1995; Skinner et al. 1999; Skinner et al. 2002). Additionally, when these high pressure systems have sufficient moisture or begin to break down, there is strong convective activity leading to a high incidence of lightning strikes (Fauria & Johnson 2006).

Glacial erosion has profoundly modified the landscape features of the region, resulting in an undulating topography with significant relief around the Lake Nipigon basin, with less severe relief in other regions. Surficial deposits are also highly variable ranging from areas of poorly drained flats of granitic sand to large areas of clay deposits caused by the spillway for glacial Lake Agassiz. The nature of soil development in the region is dependent on local combinations of climate, drainage, parent material, terrain, and vegetation over time. Podzols and brunisols are the dominant soil orders, with primary concentration of humo-ferric podzol in the central and southeastern portion and dystric brunisol occurring in the northern portion particularly in the extensive sand and gravel deposits of the plateau surrounding the Lake Nipigon basin. Sites of humo-ferric podzol developed under coniferous forest stands in moderately well-drained sites on coarse-textured, stony, glacial tills and outwash deposits on acidic parent material (Agriculture and Agri-food Canada. 1996; Baldwin et al. 2001). Dystric brunisols are closely associated with acidic bedrock and loamy to sandy acidic glacial till, with outwash and lacustrine material (Baldwin et al. 2001) on sites of excessively to very well-drained drainage classes (Agriculture and Agri-food Canada. 1996).

Post-fire successional trajectories in the ecoregion begin with pioneer species such as jack pine (*Pinus banksiana*), trembling aspen (*Populus tremuloides*), and paper birch (*Betula papyrifera*), regenerating immediately post-disturbance, with slow growing shade tolerant species like black spruce (*Picea mariana*), white spruce (*Picea glauca*) and balsam fir (*Abies balsamea*) replacing them over time (Chen & Popadiouk 2002; Hart & Chen 2008; Brassard et al. 2008). Forest composition is primarily mixedwood with a high percentage of conifer species; lowlands are dominated by black spruce, while uplands are dominated by mixedwoods (Chen & Popadiouk 2002).

The region has a long history of human activities including forest harvesting and management, forest fire suppression, and recreational activities. Timber harvesting began in the early 20th century (1910-1920) in the southern half of the study area encroaching northward over time; however, harvesting was not necessarily limited to the southern portion during 1910-1950 as harvesting based on river access occurred. In the past few decades recent forestry activity has shifted towards the northern and northwestern areas with the main concentration happening beyond the northern limit of the study area's boundary.

Time-since-fire map

Various archival sources have been used to reconstruct the fire history of the study area. Digital resources used include: Ontario's Forest Fire History a record of the location and spatial extent of most large fires that have burned in Ontario between 1921 and 1995; and the Canadian Forest Service's Large Fire Database (LFDB), that documents ignition points, location and spatial extent of fires that burned an area greater than 200 ha between the period of 1959-1997 (Stocks et al. 2002). Aerial photography and satellite imagery were interpreted in order to date and map recent fires not present in existing data sets. Several maps drawn in 1977 were supplied by Abitibi-Bowater Inc. for portions of the landscape situated within the extant boundary of the study area. These maps showed the shape, stand composition, and age of canopy trees as interpreted from aerial photographs taken during 1962-1968. All accessible hexagons were visited and to confirm the accuracy of TSF approximations from the map.

An initial time-since-fire map of the study area was compiled, showing the occurrence of the most recent fire in the landscape. The map was constructed by transferring the fire dates and refined fire boundaries from the analysis of archival data sources to the appropriate topographic map in a Geographic Information System (GIS).

ESRI ArcGIS 9.2 (2006) was used to create the TSF map and subsequent stratification and analysis.

Sampling Methodology

The data set was compiled by using systematic grid system to stratify the region assigning extant TSF burn perimeters and identifying areas requiring for dendroecological sampling. A hexagonal overlay was selected to stratify the study area, thereby creating a gapless grid that is consistent and minimizes edge effects that are present in other polygonal grid systems. The study area was subdivided into 407 hexagons, each measuring an area of 40 $km^2$. The methodology used for assigning TSF dates to each hexagon can be found in Appendix 4. If a hexagon had burn perimeters greater than 200 ha within its boundaries, the total terrestrial land area and the area consumed by each fire that burned within the hexagon's boundary was calculated. Time-since-fire dates were then assigned by assigning statistical weight based on the ratio of area consumed to terrestrial land to each fire burning (> 200 ha) within a hexagon (present per cent coverage). A hexagon could thereby have multiple TSF dates reflecting the dynamics that have created the present stand composition (Figure 3.2.); for example, a hexagon with two known TSF dates would be assigned three data entries each with different weight, one for each known fire and one for the remainder area left unburned, with a value 1.00 being the sum of weights. Assuming the majority of fire boundaries present within the TSF map were complete for the 1920-2008 time period, a censored TSF date of 89 years was assigned to any remainder unburned area within each hexagon. Hexagons without TSF dates derived from defined map boundaries, when

TSF dates were assigned by dendroecological sampling, FRI map and aerial and satellite image interpretation, a hexagon was assigned a standard weight of 1.00.

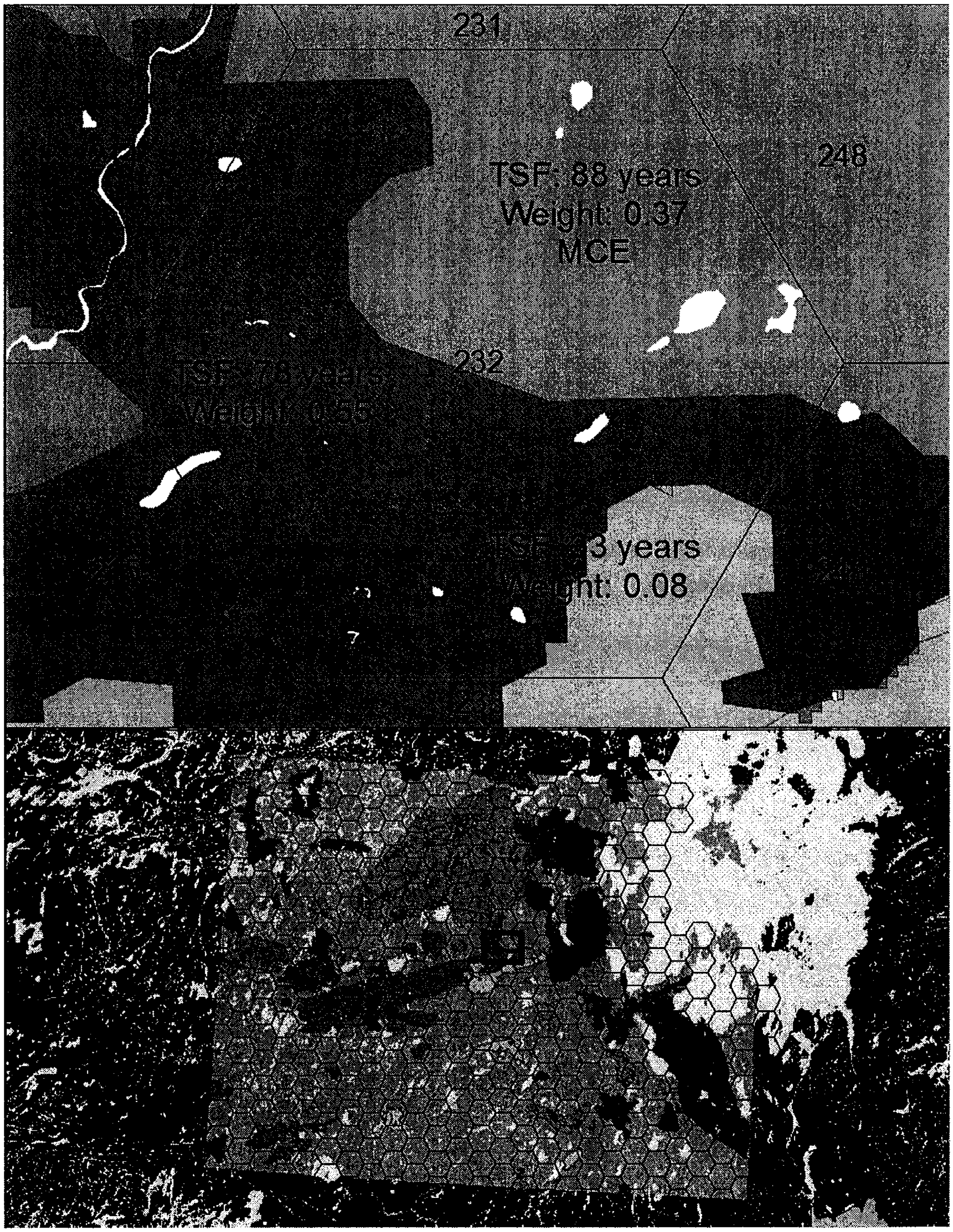



Figure 3.2. TSF map with grid overlay and an example of TSF determination within a hexagon with defined fire boundaries

Hexagons with no TSF determination and located greater than 2 km from road access were considered inaccessible (21% of the data); for these hexagons TSF was extrapolated from the Abitibi-Bowater permanent plot network if sufficiently close (< 2 km) to provided censored estimate of TSF from the oldest tree documented accounting for 4% of the data. The remaining inaccessible hexagons with no TSF determination, stand ages were interpreted from Abitibi-Bowater FRI maps and aerial photography and were used as a minimum censored TSF estimate accounting for approximately 17% of the data. Hexagons with no TSF determination (located outside of mapped burn areas), located within 2 km from road access, were visited and sampled for dendroecological analysis. Barring those that were inaccessible, all hexagons were visited to confirm the fire boundaries from the TSF map, and spatial parameters derived from archival sources.

A total of 102 hexagons with no TSF determination made up the final set that necessitated field reconnaissance. Random sampling points were generated within the terrestrial land area in each hexagon and assigned GPS coordinates to assist location in field. Efforts were made to identify evidence of past fire or harvesting at each sampling point. On-site searches for fire-scarred trees were conducted along with examining the mineral soil for charcoal. In general, finding fire-scarred boreal trees is difficult due to their relative scarcity, and that they often date low intensity surface fires that are not characteristic of the boreal fire regime; fire scarred trees were observed at two sites whereas charcoal was near universally observed at the base of the humus layer, confirming a widespread extant of wildfire within the study area. Sampling points were also inspected for evidence of harvesting by searching for cut stumps; sites where stumps were present were discarded and a new sampling point within the hexagon was generated. If no accurate TSF determination was possible (from TSF map, sampling, aerial and satellite, FRI maps, or permanent plots) due to a long history of harvesting, the hexagon was eliminated from the data set. Of the original 407 hexagons assigned, 32 were eliminated accounting for about 8% of the original hexagon count.

Within each sampling point, a 200 m transect was laid out according the point-centered quarter method (PCQ) (Mueller-dombois & Ellenberg 1974). Increment cores were extracted from ten of the largest canopy trees in each transect, with one increment core or disk taken per tree depending on the diameter and species of tree (if increment bore was of insufficient length to penetrate the pith). Increment cores or disks were extracted from the base of the trunk as low as possible through the pith of the tree to minimize aging problems (Phipps 1986). Sampling was conducted within each transect with preference given to the largest canopy trees of pioneer species that regenerate immediately post-fire. The preferred species are (in order): jack pine, paper birch, and trembling aspen; in the absence of those species, black spruce and other species were sampled.

Cores and disks were packed and transported to the laboratory for dendrochronological analysis. Sampled increment cores were mounted on constructed mounts and disks were transversely cut, all samples were sanded to make rings visible. Rings were then counted under a dissecting microscope until the same count was obtained three times (Phipps 1986; Brassard et al. 2008).

The TSF for a hexagon was estimated by using the sampled canopy trees, taking the age of the oldest tree as the TSF when at least 60% of the sampled individuals were established within the same 10 year period if the sample composition was > 80% jack pine and 20 year period for all other species (Lesieur et al. 2002; Bergeron et al. 2004b). These differential time periods were selected to account for missing rings and post-disturbance regenerative delay that may be present in shade-tolerant species that could cause us to underestimate the true age (Vasiliauskas & Chen 2002; Girardin et al. 2006b). When samples did not meet the ≥ 60% requirement, the TSF for the hexagon was determined by using the age of the oldest tree sampled as a minimum censored estimate in subsequent statistical analyses.

Statistical Analysis

The fire cycle (FC) was first calculated following the methodology of Heinselman (1973). FC was calculated for the study area by dividing the total terrestrial land area (subtracting the area of all water bodies) by the average annual area burned over the time period covered (1920-2008). This procedure follows the following formula:

[1] $FC = U/B$

where $U$ is the terrestrial land area and $B$ is the average annual area burned. The average annual area burned was calculated as the sum of the area burned from the archives divided by the number of years (88 years) in the time period covered by the TSF map. This 'burn rate' method of calculating the fire cycle as an 'average' gives an accurate estimation of the fire cycle, but it does not consider censored data.

The evaluation of the temporal and spatial parameters required the use of multiple survival analysis techniques. Survival analysis being the name given to a collection of statistical procedures for data analysis for which the outcome variable of interest is time to occurrence of event, and allows the use of censored data (Johnson & Gutsell 1994; Kleinbaum & Klein 1996). Censorship is a statistical method of handling data when the exact survival time is unknown such as in our case a minimum estimate of TSF. This is particularly relevant as there are many areas in which only a minimum age can be assigned due the following reasons: 1) cohort sampling within a hexagon having lower than 60% representation from the same decadal period, or, 2) inaccessibility (sample points > 2 km from road access) necessitating aerial and FRI estimations of TSF. The final data set included 375 hexagons consisting of 675 weighted TSF observations, with about 40% of the data considered censored.

The influence of spatial covariates on fire frequency was estimated using a proportional hazards model taking into account accelerated failure time, more commonly known as Cox regression (Cox 1972). The data were fit using the *coxph* procedure in R (The R Foundation for Statistical Computing, 2008). This procedure fits a Cox proportional hazards regression model where spatial covariates, time dependent covariates, time dependent strata, multiple events per subject, and other extensions can be incorporated. Table 3.1 summarizes the independent covariates that were taken into consideration as factors influencing the fire cycle of the study area.

Predictors included soil order and surficial deposit (coded as dummy variables), latitude and longitude (in decimal degrees), drainage, elevation, aspect, and distance to firebreaks. Soil order, surficial deposit, and drainage class were adapted from *Soil Landscapes of Canada*, a series of GIS coverages that show the major characteristics of soil and land for the whole country (Agriculture and Agri-food Canada. 1996). Firebreak distance was calculated as the mean distance to a waterbody (river or lake), measured in kilometres, along the four cardinal directions and their intermediates on a 1:250, 000 topographic map, and in ArcGIS. To incorporate aspect, a continuous variable, which varies within a circular scale, into the linear context of the Cox regression model, we had to convert the aspect class (north, north-east, etc) into x and y coordinates following the methodology presented in Legendre and Legendre (1998) and applied in a similar context by Cyr et al. (2007).

Table 3.1. Details of environmental covariates considered in the survival model

| | | | | |
|---|---|---|---|---|
| Vegetation | Stand type [a] | Coniferous | Broad scale | Nominal |
| | | Mixed | Broad scale | |
| Geography | Longitude | | Broad scale | Continuous (decimal degrees) |
| | Latitude | | Broad scale | Continuous (decimal degrees) |
| Physiography and Topography | Surficial deposit [b] | Ground moraine | Broad scale | Nominal |
| | | End moraine | Broad scale | |
| | | Outwash deposit | Broad scale | |
| | | Beach and Aeolian deposit | Broad scale | |
| | Soil order [c] | Dystric brunisol | Broad scale | Nominal |
| | | Humo-ferric podzol | Broad scale | |
| | Drainage [c] | Scale, from 1 (very poorly drained) to 7 (excessively well-drained) | Local scale | Ordinal |
| | Slope aspect | West-east axis (x) | Local scale | Continuous (x, y) |
| | | South-north axis (y) | Local scale | |
| | Mean firebreak distance [d] | | Intermediate | Continuous (metres) |
| | Elevation | | Local scale | Continuous (metres) |

[a] Stands were categorized according to greater than 75% basal area composition belonging to either coniferous or deciduous vegetation. Stands with in which neither coniferous nor deciduous vegetation account for 75% or more total basal area were categorized as mixedwood. Collinearity with other covariates were tested, however stand type was not selected as an explanatory variable for Cox regression analysis.
[b] evaluated by adapting the Forest Landscape Ecology Program (1996)
[c] evaluated from data from Agriculture and Agri-Food Canada (1996)
[d] The mean firebreak distance considered the eight nearest waterbody measured along all cardinal directions from the random sampling point in ArcGIS 9.2

Each aspect class was positioned on a trigonometric circle of radius 1 centered at the origin where the angle corresponds to the azimuth of the dominant aspect of the slopes, with the horizontal axis (x) equivalent to the west-east axis, while the vertical axis equivalent to the north-south axis. Coordinates were assigned as follows: north (0, 1), north-east (0.7071, 0.7071), east (1, 0), south-east (0.7071, -0.7071), south (0, -1), south-west (-0.7071, -0.7071), west (-1, 0), and north-west (-0.7071, 0.7071) where $\sin(45°) = \cos(45°) \approx 0.7071$.

In order to assess possible collinearity among the covariates in the data set, Spearman's rank correlation coefficients were calculated. This allowed us to evaluate whether some covariates should be discarded as their probable effects could be explained by another more ecologically appropriate variable. Most of the correlations among covariates are not very strong ($r < \pm 0.30$, see appendix 2), with the exception of longitude and elevation ($r \approx 0.73$); drainage classes, extremely well drained and moderately well drained, and soil order ($r \approx -0.56$ - $0.71$); and drainage class, extremely well-drained and latitude ($r \approx -0.76$). Longitude and elevation are closely related by nature, with elevation gently decreasing along a horizontal axis from west to east towards Lake Nipigon. Similarly sites of soil order dystric brunisol are associated with the excessively well-drained drainage class while sites of humo-ferric podzol were associated with the moderately well-drained drainage class. We found that areas of the excessively well-drained drainage class roughly corresponded with higher latitudes along the northern border of our study boundary and moderately well-drained drainage class with the south and central regions. Based on our correlation matrix we thus eliminated drainage class from our survival model in favour of dominant soil order and which takes into account not only drainage class, but parent material, vegetation type, and climate and does not correlate significantly with latitude.

Fire cycles were then calculated for different temporal epochs using Maximum Likelihood Estimation (MLE) in survival analysis. Survival probabilities and baseline hazard functions were calculated for multiple time periods; three different time periods were selected in this study based on the age of the dendroecological samples. The first time period, from 1820 to 1920, covers the time from the end of the Little Ice Age (LIA) to the beginning of forestry activity in the study area. The second, from 1920 to 2008, covers the fire cycle for the most recent epoch in which we suspect fire frequency has shifted. Lastly, 1820-2008, covers the time period for the length of the reconstruction.

Calculations were made using the *basehaz* function in R to compute the baseline hazard functions for our data set using the Cox proportional hazards model. The hazard function or 'burn rate' to which it is often referred in forest fire literature refers to the instantaneous probability of fire and is statistically equivalent to fire frequency. Calculations of the burn rate allow us to estimate the fire cycle by averaging the fire frequency (average hazard) over a designated time period, and taking its inverse (Johnson & Gutsell 1994). Fire cycles were calculated in this way for each time period (1820-2008, 1820-1920, and 1920-2008) for the entire study area, and for stratifications based on any significant covariates.

## RESULTS

### Total area burned (1920-2008)

The total area burned (including areas burned multiple times) between 1920 and 2008 covers 645, 213 ha, or approximately 55% of the study area. According to the

TSF map, during the 1920-2008 epoch, fires burned an average of 7332 ± 35, 503 ha (mean ± SD) or about 0.6% of forest per year. The fire cycle for the 1920-2008 time period was calculated using Heinselman's burning rate method to be 158 years (Table 3.2). Based on the total area burned between 1920 and 2008, approximately 35% of the landscape is characterized by stands that originated from major fire events occurring in the past three decades (1980-2008) (Figure 3.3).

Table 3.2. Fire cycle calculations for 3 distinct temporal epochs

| | Time period investigated | | |
|---|---|---|---|
| Method | 1820-2008 | 1820-1920 | 1920-2008 |
| Burning rate method | N/A | N/A | 158 |
| Inverse hazard method | 150 (126-188) | 295 (229-499) | 96 (84-111) |

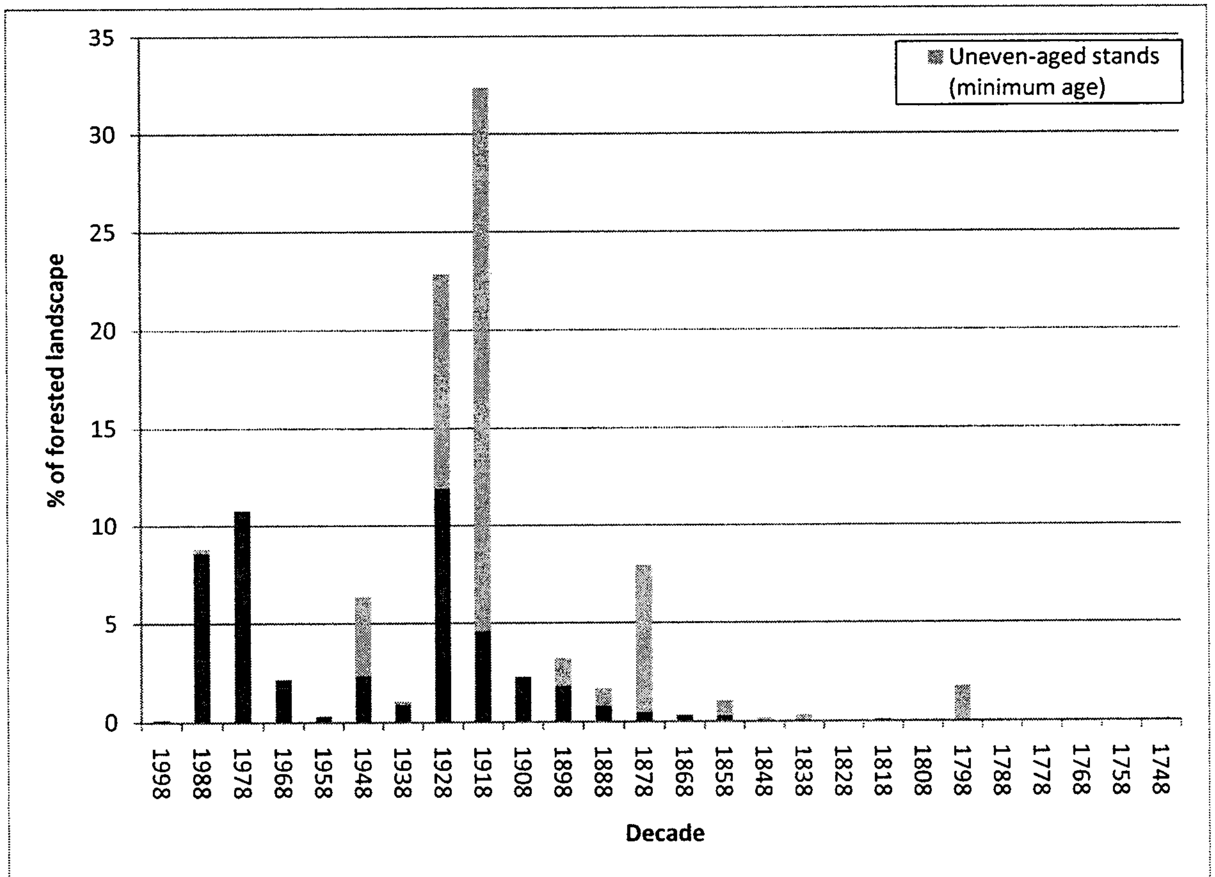



Figure 3.3. Origin of stands by decade 1756-2008

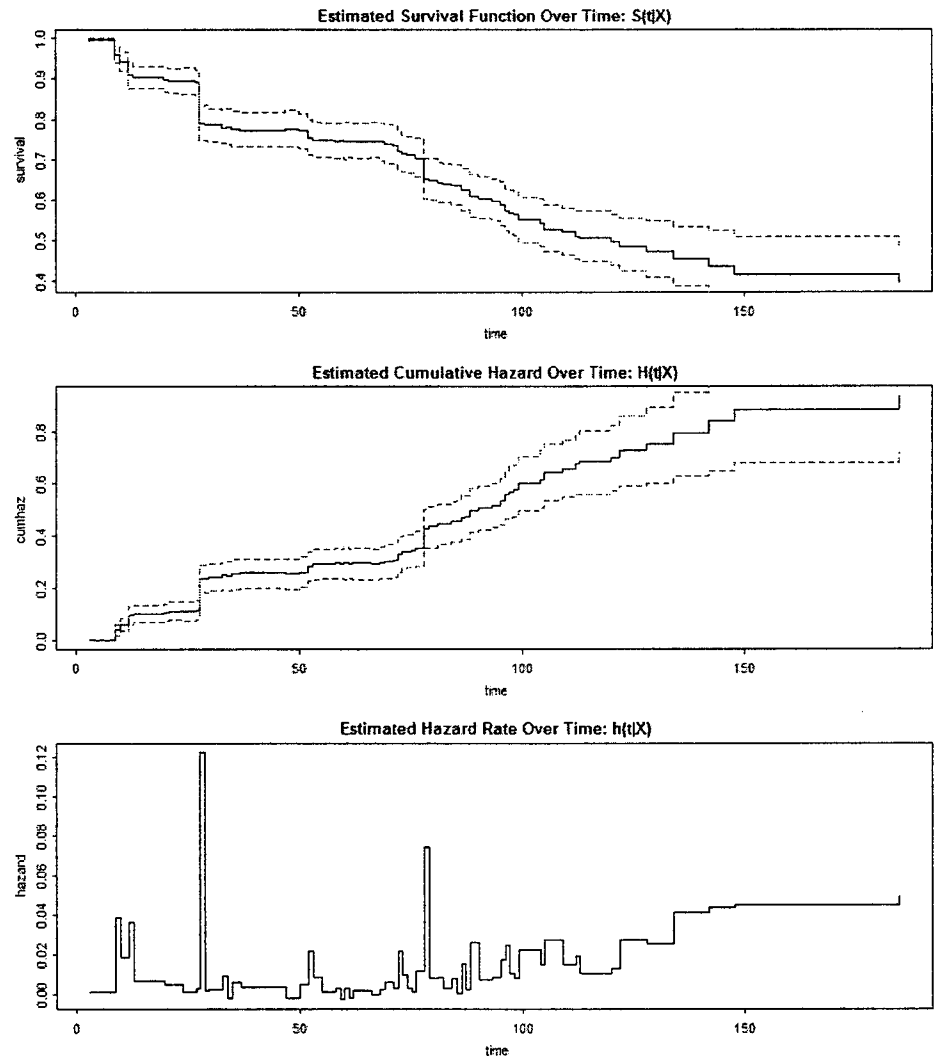



Figure 3.4. Survival function, cumulative hazard, and instantaneous hazard for study area during 1756-2008

Survival Analysis: Temporal Variability

The fire cycle for the study area has not remained constant over time; the frequency of large stand-replacing fires has increased in recent decades decreasing the regional fire cycle (Table 3.2). An examination of the temporal evolution of the burn rate through time (Figure 3.4) reveals a fire cycle of 150 (126-188) years (1820-2008); the most recent epoch (1920-2008) has a short fire cycle of 96 (84-111) years, which stands in stark contrast to the 1820-1920 epoch during which fire cycle was significantly longer (295 (229-499)).

Survival Analysis: Spatial Variability

In the first group of covariates entered into the Cox regression model only dominant soil order and firebreak distance were significantly related to fire frequency (Table 3.3). After incorporating soil order and firebreak distance a statistically significant relationship between latitude and fire frequency can be found. Soil order entered first into the regression model followed sequentially by latitude and firebreak distance after three rounds of backward stepwise selection.

Table 3.3. Summary of the backward stepwise selection of covariates in the Cox Proportional Hazards model

| Covariates | Initial Model | | | Final Model | | |
|---|---|---|---|---|---|---|
| | $\chi^{2\,a}$ | Prob > $\chi^2$ | Fire frequency ratio | $\chi^{2\,a}$ | Prob > $\chi^2$ | Fire frequency ratio |
| **Latitude** | **2.84** | **0.092** | **1.523** | **25.23** | **0.041** | **2.20** |
| Longitude | 0.27 | 0.605 | 0.929 | - | - | - |
| **Soil Order: Humo-ferric Podzol** | **6.76** | **0.009** | **1.533** | **25.23** | **0.004** | **1.78** |
| Surficial Deposit: Beach and Aeolian Deposit | 0.87 | 0.379 | 0.696 | - | - | - |
| Surficial Deposit: Organic Deposit | 0.00 | 0.955 | 0.986 | - | - | - |
| Surficial Deposit: Outwash Deposit | 0.82 | 0.352 | 1.228 | - | - | - |
| Surficial Deposit: Ground Moraine | 0.04 | 0.841 | 0.966 | - | - | - |
| **Mean Firebreak Distance** | **5.73** | **0.016** | **1.299** | **25.23** | **0.011** | **1.38** |
| Aspect (west-east axis) | 0.06 | 0.811 | 0.973 | - | - | - |
| Aspect (north-south axis) | 0.40 | 0.525 | 0.931 | - | - | - |
| Elevation | 0.25 | 0.614 | 0.999 | - | - | - |

[a] The $\chi^2$ values listed above come from the log-likelihood ratio tests from the Cox proportional hazards model described in detail in (Cox 1972)

The fire frequency ratios, also called hazard ratios in survival analysis literature, resulting from the Cox regressions, were used to quantify the influence of each covariate on fire frequency. For indicator (dummy) covariates, the FF ratios were interpreted as the ratio of the estimated burn rate for those with a value of 1 to the estimated burn rate for those with a value of 0 (controlling for all other covariates). For quantitative covariates the FF ratios gives the estimated percent change in the burn rate for each one-unit increase in the covariate. The fire frequency ratio in the final model associated with the dominant soil order humo-ferric podzol (1.78; Table 3.3), indicates the hazard of burning is 78% greater on these soils than sites of other soil orders (dystric brunisol). Similarly, in the final model latitude was found to significantly affect fire frequency (FF ratio 2.20; Table 3.3); the results indicate an increasing gradient in fire frequency from south to north, as for each one-degree increase in latitude the burn rate increases by an estimated 120%. Distance to firebreaks was also significant having a FF ratio of 1.38; the interpretation being that as distance from waterbodies increases so too does burn rate; for each 1 km increase in distance the burn rate increases by 38%.

After incorporating soil order, latitude, and firebreak distance into the final model, it is not possible to detect any other statistically significant relationships at $\alpha = 0.05$, as all other variables were removed from the model. These results suggest that within the study area the effects of longitude, surficial deposit, slope aspect, and elevation were not significantly related to fire frequency at the scale examined. In the final model, the fire frequency ratios of soil order and firebreak distance experience modest increases from the initial model. Conversely, significance and FF ratio of latitude increases significantly (FF ratio: 2.20).

Table 3.4. Fire cycle calculations for spatial stratifications over three distinct time periods

| | Time period investigated | | |
|---|---|---|---|
| Variable Inclusion | 1820-2008 | 1820-1920 | 1920-2008 |
| Humo-Ferric Podzol | 115 (94-150) | 289(196-551) | 68 (59-82) |
| Dystric Brunisol | 209 (142-399) | 216 (136-643) | 202 (154-292) |
| Northern partition | 123 (90-194) | 224 (130-794) | 82 (67-105) |
| Southern partition | 134 (105-185) | 192 (142-357) | 100 (81-130) |

Following spatial analysis and discovery of the significant factors influencing the regional fire regime, fire cycles were re-calculated for stratifications based on the significant covariates for each time period (1820-2008, 1820-1920, 1920-2008) (Table 3.4). Upon stratification we find that the fire cycle calculations for soil order indicate differential patterns over time for both humo-ferric podzol and dystric brunisol (Figure

5). Humo-ferric podzol has a significantly shorter fire cycle in the 1920-2008 time period (68 years) compared with brunisols (202 years), while the reverse is true during 1820-1920 (289 years podzol, 216 years brunisol) although the confidence intervals during this time period overlap indicating relatively minor differences between the soil orders during this time period.

As latitude is a continuous variable, it does not lend itself to stratification without modification. The fire cycle calculations for latitude were conducted by stratifying the study area into a northern and southern half at. The northern partition has a shorter fire cycle (82 years) than the southern partition (100 years) in the most recent epoch 1920-2008. Prior to 1920, the fire cycle was longer in the northern half (224 years) than the southern half (192 years). The fire cycles for these halves indicate that fire cycle has decreased over time from 1820-2008, with the northern half experiencing a larger decrease (Figure 3.5).

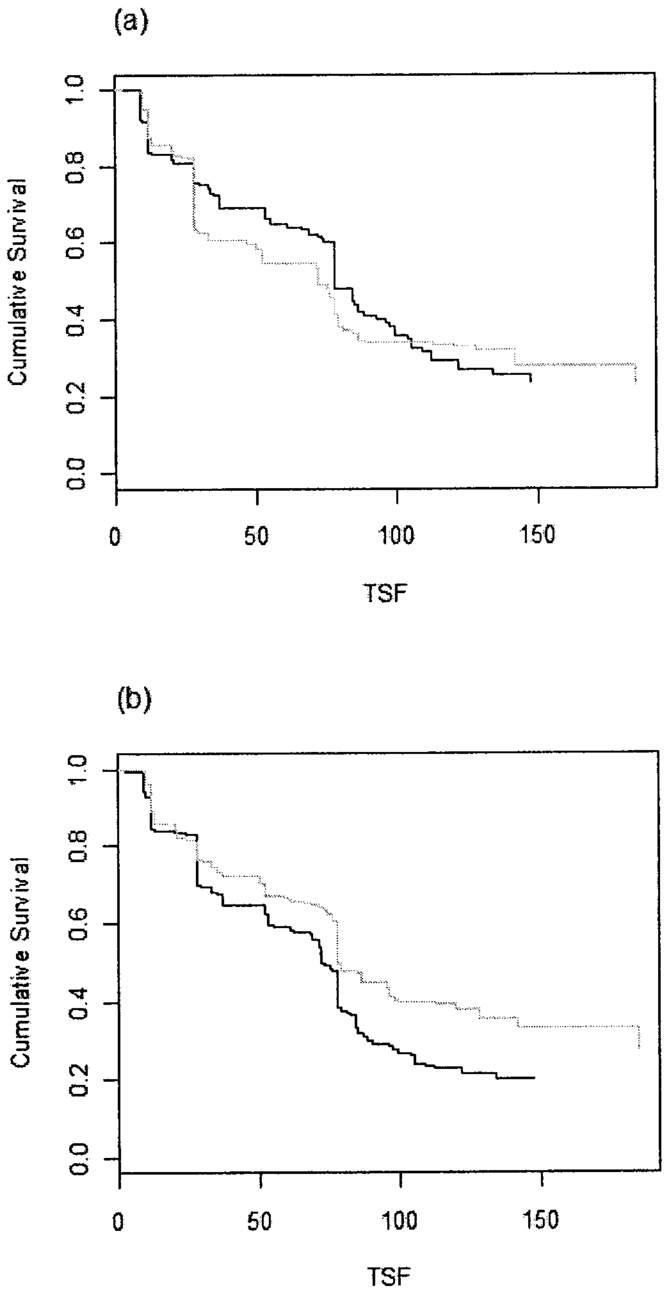



Figure 3.5. (a) and (b) Cox regression survival curves for stratifications by significant spatial covariates for the study region on timescale 1820-2008. (a) Stratification by latitude (northern vs. southern partition) with black representing southern partition and gray representing the northern partition. (b) Stratification by soil order with black representing humo-ferric podzol and gray representing dystric brunisol.

# DISCUSSION

## Temporal Variation

The length of the fire cycle suggests two patterns of fire regime development for the region: many small fires or infrequent but large fires. The results from the fire cycle calculations using Cox regression and the burn perimeters from the time-since-fire map for the region supports the latter pattern of fire regime development that infrequent large fires characterize fire frequency in this region of the central boreal shield. Relative agreement between the fire cycle calculation using burn rate based solely on archival data (158 years; 1920-2008) and survival analysis (150 years; 1820-2008) provide confidence in our determination of a fire cycle situated around 150 years for this region.

The observed shifts from longer fire cycles in the past to a shorter cycles over the last century is likely due to a combination of climate change and human influences. This shift from a longer to shorter fire cycle is consistent with what one might expect following the end of the Little Ice Age (LIA) with cool moist weather during the LIA hindering fire ignition and spread, transitioning into warmer drier conditions in recent decades creating ideal fire conditions. Area burned in Canada has increased in recent decades (Skinner et al. 1999; Skinner et al. 2002; Stocks et al. 2002) our results support this, however, this increase in fire activity stands in stark contrast with a trend of declining fire frequency during the 20$^{th}$ century that has been reported in eastern (Bergeron et al. 2001; Lauzon et al. 2007), and western (Bergeron et al. 2004b; Grenier et al. 2005) Quebec as well as in western Canada (Johnson et al. 1998; Weir et al. 2000). In northwestern Ontario, the increase in fire frequency relates to an increase in temperature since the end of the LIA in central Canada (Gillett et al. 2004), and an important change in the circulation of global air masses that regulate regional climate (Girardin et al. 2004b). Regional drought reconstructions for area denote 1918-22 and 1934-38 (Girardin et al. 2005), these drought periods coincide with extreme fire years. The observed fire cycle decrease in our study area has occurred despite an increase in area under fire management, more efficient fire suppression techniques, and an extensive road network improving access and forest fragmentation, which suggests the fire cycle change was driven primarily by climate, with the aforementioned human activities contributing to spatially mixed fire frequencies.

Spatial Variation

Although the climate change that has occurred since the end of the LIA is known to have had a major impact on boreal fire frequencies (Larsen 1997; Bergeron et al. 2001), our results suggest that human activities have exerted significant effects on the fire regime of our study area. Evaluation of spatial covariates show that the burn rate increases on a latitudinal gradient; considering the northern and southern boundaries of the study area are bound between 49-50° N latitude, the northern limit can be said to have a 120% greater chance of experiencing a stand replacing fire than the southern limit. Colonization and its associated timber harvesting and fire suppression activities appear to have contributed to spatially mixed fire frequencies, with longer fire cycles in areas adjacent to developed sections near the southern boundary and larger annual area burned at increasing latitudes.

Forest fragmentation began during different time periods in the northern and southern portions of the study area. Historically, European settlement in the area occurred in the late 1600s, with the primary industry being the fur trade, little or no commercial exploitation of timber resources occurring in the region until 1910-1920, and then it was limited northward by accessibility

The southern portion of the study area has a dense road network relative to the northern portion, if we assume that roads have the ability to act as firebreaks, inhibited fire spread from adjacent forested regions seems likely. The road network density is related to an earlier history of timber harvesting in the southern portion, which not only fragments the continuous forest cover but also inhibits lightning-initiation due to the lack of forest cover (Krawchuk et al. 2006). While we propose that historical patterns of timber harvesting as an explanation for the increase in burn rate observed on a latitudinal gradient, there is also evidence that suggest fire suppression is linked to the significance of latitude within our study area.

Fire suppression during the past 50 years has contributed to changes in the recurrence of fire in many regions of Canada (Bourgeau-Chavez et al. 2000). The introduction of water bombers combined with improved fire detection, response systems, and communications have improved boreal forest firefighting (Cumming 2005). Prior to the technological advances of the 1950s, fire suppression activities were not as effective in controlling large fires and was geographically confined to the southern half of the region where most commercial timber harvesting occurred, whereas the northern half was governed, primarily, by natural forces due to its relative inaccessibility. The effectiveness of fire suppression is dependent on prompt reporting influencing response time, by allowing fires to burn without suppression for longer periods of time they can grow, potentially, into crown fires; considering the absence of communities, communications, and the distance from fire response centres at higher latitudes, the influence of fire suppression could account for some of the spatial variation associated with a lower fire frequency at lower latitudes. In addition, prompt suppression action is taken more often on fires in the lower boreal zone, while values-at-risk generally dictate whether or not more northerly fires receive suppression, and fire is allowed to assume its natural role. Considering historical patterns of development, regional accessibility, fire management response time, and suppression history, the picture that emerges is one illustrating the geographical impact of human activities on a latitudinal scale in the study area.

Within the study area dominant soil order of a site is a strong indicator of forest fire frequency. The majority of soils within the study area belong to either brunisolic or podzolic soil order, with dystric brunisol and humo-ferric podzol being the dominant great group within each order. Humo-ferric podzol dominates (64%, n=238) the central and southern region, occurring under coniferous and mixed-wood vegetation on moderately well-drained drainage class; dystric brunisol is present in the northwest, northeast, and south-western areas of the region (36%, n=134) occurring under forest vegetation, shrubs, and grass, on the excessively well-drained drainage class. Results show that sites located on humo-ferric podzol soil have an increased (about 79%) chance of burning than sites of dystric brunisol. It is difficult to interpret these results, as the soil orders are largely similar as both are able to support a similar range of vegetation and have similar a pH (acidic), parent material (coarse textured base-poor, e.g. sands and sandy tills), and drainage classes (extremely-well to poorly drained), differing in minor respects, specifically organic matter content of brunisols lacking a developed podzolic B horizon. Additionally the spatial distribution of soils in the region seemingly contradicts the significance of latitude as the primary concentration of brunisol is in the northern partition. The tie that binds soil order and TSF is not readily detectable, although we surmise that it is a probable consequence of the podzolization process underlining the relationship between forest vegetation and soil and local drainage classes.

Podzol development in northwestern Ontario was the natural outcome of soil development following colonization of bare soil after glaciations 9000-12, 000 years ago (Lundstrom et al. 2000); soil development in the region is dependent on four factors: climate, parent material, vegetation, and topography (Jenny 1941; Jauhiainen 1973); of these vegetation and climate are key. Podzolization is encouraged under forest conditions that favour the development of an organic surface layer. Compared to other deciduous litter, coniferous litter is decay resistant, have the highest concentration of lignin, and the lowest concentrations of plant macro nutrients (Flanagan & Vancleve 1983) essential to microbial activity; combined these characteristics reduce forest floor decomposition and nutrient availability under conifer dominated stands. Coniferous forests also facilitate podzolization by providing year-round shade of the forest floor further inhibiting litter decomposition and vascular plant growth creating a mat of organic materials that leach into the soils (Bonan & Shugart 1989; Willis et al. 1995). The historical dominance of black spruce would encourage podzolization on moderately well-drained soils. The increasing FF ratio results for humo-ferric podzol is likely caused by an abundant presence of black spruce, in mixedwood stands, which are prone to crown fires owing to the presence and arrangement of ladder fuels in mature cohorts.

Conversely, dystric brunisol, an immature soil, lacks the degree of B horizon development specified for podzolic order (Soil Classification Working Group 1998). Additionally, brunisolic soils within the study area are highly correlated ($r = 0.57$) with the extremely well-drained drainage class. Deciduous litter undergoes rapid decomposition retarding organic matter accumulation. Mixedwood and deciduous canopies are typically more open then conifer dominated canopies permitting more light on the forest floor, supporting more vascular plant, sapling, shrub and grass vegetation. Thus we explain the decreasing hazard of burning for sites of dystric brunisol, by the presence of deciduous species, such as trembling aspen which has been found to inhibit fire spread (Cumming 2001), comprising a larger portion of stands on these soils. Barring the previous explanations, it may be useful to carefully analyze the effect of local site conditions such as soil moisture, soil texture, and fertility on the rate of forest growth, species composition and fuel types, which are well known to drive some spatial variation in the fire regime (Heyerdahl et al. 2001; Bergeron et al. 2004b).

Mean distance to a firebreak (defined as any river, lake, or stream) was found to positively alter fire frequency (FF ratio 1.38, $p = 0.011$); regionally, firebreaks were relatively effective in limiting the expansion of large fires. There is a high degree of variability, as a fire of sufficient size and intensity may bypass these firebreaks entirely, either by drying out and igniting nearby fuels without direct contact, or by throwing flaming embers over short distances (e.g.: narrow rivers) or long distances (wind permitting) (Vanwagner 1983; Larsen 1997; Cyr et al. 2005). In addition, the efficiency of each firebreak likely varies substantially according to size and density. The average distance to a firebreak for all sites was short, about 3 kilometres, a result of the abundant presence of natural water bodies in the region carved during Pleistocene glaciations. As a general rule however, sites located near firebreaks were generally exempted from fire for longer intervals; caution is urged in this interpretation as a few sites with short firebreak distances have burned recently (TSF = 12), while a few sites with long distances have been spared for more than 250 years. Larsen (1997), suggests that covariations between soil type and waterbreak distance influence dominant forest type and fire frequency as mean waterbreak distance was significantly higher in jack pine and aspen forests than in black or white spruce forests. While it is interesting to hypothesize about a relationship between vegetation type and mean distance to waterbreaks, no significant covariation was found in our data set between these variables (appendix 2), however it is possible that the local site data gathered was inadequate to assess the relationship.

Topographically complex landscapes, presence of firebreaks, and other site-specific characteristics (vegetation) have been identified as factors hindering fire spread thereby promoting local-level deviations from regional fire patterns (Bergeron 1991; Bergeron & Dansereau 1993; Larsen 1997). The Cox proportional hazard regression model used in the examination of spatial covariates estimates by partial likelihood, disallowing rigorous estimations of an individual covariate's contribution to survival, as the variance is not partitioned. However the nature of the covariates and inclusion in the final model indicates that the regional fire regime responds more to top-down controls, such as latitude and broad-scale topographic context (dominant soil order influencing vegetation patterns), than to bottom-up fine-scale mechanisms of fire spread such as surficial deposit, aspect, or elevation.

Aside from firebreak distance, no significant differences in fire frequency owing to local factors of elevation, aspect, or surficial deposit were detected. The forests studied are predominantly mixed wood boreal, consisting of species such as jack pine, black and white spruce, and balsam fir which are conducive to high intensity crown fires influencing a large-scale fire regime. As such, fire frequency is controlled by broad top-down controls (climate), as large fires coincide with relatively infrequent but extreme weather conditions (drought and lightning storms), whereas the ever present local factors are subordinated to a relatively minor role. Additionally, landscape patterns may have little influence on fire behaviour when burning conditions are extreme (Johnson et al. 1990). It is possible that existing relationships between local factors and fire frequency were not detected due, in part, to the grid size (40 $km^2$) used to stratify the area. The grid size was selected to ensure that samples would be taken from pioneer species following large stand-replacing fires, which was our chief sampling objective; this coarse scale sampling many not have been adequate to reveal a statistically significant relationship among those local spatial factors that may be connected more to probability of ignition or to smaller, less lethal fires.

## Forest Management Implications and Conclusion

The mixedwood boreal forest of our study area is infrequently disturbed by large fires, however, fire frequency has been increasing since the turn of the 20$^{th}$ century. Reconstructing the past and predicting the future fire activity is of key for effective implementation of sustainable management strategies. Our results indicate that fire cycle has been decreasing since the early 20$^{th}$ century, and that variation in fire frequency is spatially mixed as a result of climate change and human activities. The recent shift in fire frequency and the projected expected shifts in fire regime in response to global climate change pose formidable challenges to policy makers concerned with socioeconomic aspects of boreal forest management. Predictions based on 2 × $CO_2$ models suggest fire frequency may potentially increase over central and western Canada in the future in response to global climate change (Flannigan et al. 2001; Bergeron 2004). Similarly, predictions indicate that, by 2061-2100, the typical number of large forest fires per year could increase by more than 34% when compared with the past two centuries (Girardin & Mudelsee 2008). Although these future projections become complicated in reality due to the complex linkages among forest disturbances, anthropogenic factors (current logging practices and fire suppression), and the uncertain rate of climate change, if predicted increases of fire activity in, then there will be considerable changes in forest ecosystems and their arrangement within the landscape that managers will need to address in response to a dynamic fire regime.

The spatially mixed fire frequencies in our results are considerable, and likely a permanent feature of the region having been linked to the physical environment (soil order and associated drainage, and firebreak distance). There are potential implications from both management and ecological perspectives. A better understanding of how the landscape features influence regional fire frequency and the effective spatial scales at which they operate can increase our ability to predict future fire occurrences within the study area. This knowledge may be useful for not only fire management, but by incorporating spatially mixed rates of natural depletion (fire frequencies) affecting stand attributes and spatial distributions and identifying regions, the information can be used to optimize the forest management planning process within the region.

The anticipation of future climate changes may contribute to the development of forest management adaptation strategies to future fire regimes driven by climate variability. The adaptation options most recommended are to increase suppression effort, salvage logging, and regeneration enhancement, so as not to increase the probability of interference between fire disturbance and silvicultural management (Le Goff et al. 2005). The characteristics of the new regional fire regime must be reasonably well understood with the potential interactions made clear if adaptive management strategies are to succeed in mitigating/minimizing the potential changes. In some cases these options may be undesirable depending on the values at stake or are currently practiced at the limits of socioeconomic feasibility. Future management will have to balance the tangible economic realities like timber supply and fire management costs with the somewhat intangible long-term ecological values at risk. These analyses of the significant temporal and spatial factors that have influenced the regional fire regime are available so forest practitioners can compose improved estimates for use in their timber and habitat supply modelling which should help improve the assessment of sustainability of forest management plans and natural disturbance emulation. Unease with the inherent uncertainty that surrounds the future of the regional fire regimes should not be an impediment to improved management practices based on current knowledge if the assumptions are made explicit and the unknowns and uncertainties are addressed, with the continual improvement of this knowledge a goal.

# CHAPTER FOUR. GENERAL CONCLUSIONS

The findings of my thesis confirm that the fire cycle of the study area has shown considerable spatial and temporal variations. A summary of my research findings, conclusions, management implications, and recommendations are as follows:

(1) Our results show that, within the studied region, there is a considerable decrease in the length of the fire cycle since the early 1920s that is likely driven by climate change following the end of the LIA. Moreover, the fire cycles reported here are shorter than those generally reported for the eastern boreal forest, and closer to those reported in western boreal forest. Fire frequency and annual area burned increase on a south-north latitudinal gradient suggesting that landscape fragmentation (harvesting and road construction) and fire suppression have decreased fire frequency owing to temporal differences in the onset and execution of these practices.

(2) The environmental factors controlling the spatial heterogeneity of the regional fire regime vary according to the spatial scale (top-down or bottom-up) considered. Fire frequency is the study area is mainly affected by top-down control; the environmental factors described on broad spatial scales had an impact on the regional fire frequency. Fire frequency is higher on moderately well-drained soils of humo-ferric podzol, and lower on extremely well-drained soils of dystric brunisol. From combined properties of dominant soil order and its associated drainage class we infer the influence of vegetation type on the regional fire regime. The conditions necessary for the development of a podzolic B horizon, coupled with moderately well drained soils, suggest that sites of humo-ferric podzol have likely been dominated by coniferous black spruce which are prone to large crown fires under the severe weather conditions that characterize the

boreal forest fire regime. Dystric brunisol, an immature soil that is excessively well-drained (within the study area), likely supports stands comprised of deciduous hardwoods, such as trembling aspen that are not prone to crown fires, and jack pine which have some degree of fire resistance.

(3) Fire frequency increases as mean distance (km) from a waterbody increases. Water bodies can reduce local fuel moisture levels altering the probability of a successful ignition. Additionally water bodies act as natural fuel breaks in the landscape inhibiting fire spread.

(4) Local landscape features, usually classified as bottom-up controls, like surficial deposit, slope aspect, and elevation were not significantly related to fire frequency at the scale examined. Fire frequency did not vary across a longitudinal gradient within the study area.

(5) The spatially mixed fire frequencies suggest substantial intraregional heterogeneity, with potential opportunities for management to optimize sustainable forestry management plans. The increase in fire frequency in recent decades and the projected increase in fire activity over much of boreal northwestern Ontario presents many challenges to forest managers to ensure sustainable forest management in the future.

**References**

Agriculture and Agri-food Canada. 1996. Soil Landscapes of Canada, Version 2.2, National Soil Database. Research Branch, Agriculture and Agri-Food Canada, Ottawa.

Amiro, B.D., Logan, K.A., Wotton, B.M., Flannigan, M.D., Todd, J.B., Stocks, B.J., and Martell, D.L. 2004. Fire weather index system components for large fires in the Canadian boreal forest. International Journal of Wildland Fire **13**(4): 391-400.

Baldwin, D.J.B., Desloges, J.R., and Band, L.E. 2001. Physical Geography of Ontario. *In* Ecology of a Managed Terrestrial Landscape: Patterns and Processes of Forest Landscapes in Ontario. UBC Press, Vancouver.

Barrett, S.W. 1988. Fire suppression's effects on forest succession within a central Idaho wilderness. Western Journal of Applied Forestry **3**: 76-80.

Barton, A.M., Swetnam, T.W., and Baisan, C.H. 2001. Arizona pine (Pinus arizonica) stand dynamics: local and regional factors in a fire-prone madrean gallery forest of Southeast Arizona, USA. Landscape Ecology **16**(4): 351-369.

Bergeron, Y. 1991. The Influence of Island and Mainland Lakeshore Landscapes on Boreal Forest-Fire Regimes. Ecology **72**(6): 1980-1992.

Bergeron, Y. 2004. Is regulated even-aged management the right strategy for the Canadian boreal forest? For. Chron. **80**(4): 458-462.

Bergeron, Y., Cyr, D., Drever, C.R., Flannigan, M., Gauthier, S., Kneeshaw, D., Lauzon, E., Leduc, A., Le Goff, O., Lesieur, D., and Logan, K. 2006. Past, current, and future fire frequencies in Quebec's commercial forests: implications for the cumulative effects of harvesting and fire on age-class structure and natural disturbance-based management. Can. J. For. Res. -Rev. Can. Rech. For. **36**(11): 2737-2744.

Bergeron, Y. and Dansereau, P.R. 1993. Predicting the Composition of Canadian Southern Boreal Forest in Different Fire Cycles. Journal of Vegetation Science **4**(6): 827-832.

Bergeron, Y., Flannigan, M., Gauthier, S., Leduc, A., and Lefort, P. 2004a. Past, current and future fire frequency in the Canadian boreal forest: implications for sustainable forest management. Ambio **33**: 356-360.

Bergeron, Y. and Flannigan, M.D. 1995. Predicting the Effects of Climate-Change on Fire Frequency in the Southeastern Canadian Boreal Forest. Water Air and Soil Pollution **82**(1-2): 437-444.

Bergeron, Y., Gauthier, S., Flannigan, M., and Kafka, V. 2004b. Fire regimes at the transition between mixedwood and coniferous boreal forest in Northwestern Quebec. Ecology **85**(7): 1916-1932.

Bergeron, Y., Gauthier, S., Kafka, V., Lefort, P., and Lesieur, D. 2001. Natural fire frequency for the eastern Canadian boreal forest: consequences for sustainable forestry. Can. J. For. Res. -Rev. Can. Rech. For. **31**(3): 384-391.

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

Bessie, W.C. and Johnson, E.A. 1995. The Relative Importance of Fuels and Weather on Fire Behavior in Sub-Alpine Forests. Ecology **76**(3): 747-762.

Beverly, J.L. and Martell, D.L. 2005. Characterizing extreme fire and weather events in the Boreal Shield ecozone of Ontario. Agricultural and Forest Meteorology **133**(1-4): 5-16.

Bonan, G.B. and Shugart, H.H. 1989. Environmental-Factors and Ecological Processes in Boreal Forests. Annual Review of Ecology and Systematics **20**: 1-28.

Bourgeau-Chavez, L.L., Alexander, M.E., Stocks, B.J., and Kasischke, E.S. 2000. Distribution of Forest Ecosystems and the Role of Fire in the North Americal Boreal Region. *In* Fire, Climate Change, and Carbon Cycling in the Boreal Forest. Springer, New York. pp. 111-131.

Brassard, B.W., Chen, H.Y.H., Wang, J.R., and Duinker, P.N. 2008. Effects of time since stand-replacing fire and overstory composition on live-tree structural diversity in the boreal forest of central Canada. Can. J. For. Res. -Rev. Can. Rech. For. **38**(1): 52-62.

Bridge, S.R.J. 2001. spatial and temporal variations in the fire cycle across ontario. OMNR, Northeast Science & Technology. NEST TR-043 41p.

Bridge, S.R.J., Miyanishi, K., and Johnson, E.A. 2005. A critical evaluation of fire suppression effects in the boreal forest of Ontario. Forest Science **51**(1): 41-50.

Brown, P.M., Kaye, M.W., Huckaby, L.S., and Baisan, C.H. 2001. Fire history along environmental gradients in the Sacramento Mountains, New Mexico: Influences of local patterns and regional processes. Ecoscience **8**(1): 115-126.

Chen, H.Y.H. and Popadiouk, R.V. 2002. Dynamics of North American boreal mixedwoods. Environ. Rev. **10**: 137-166.

Chen, H.Y.H., Vasiliauskas, S., Kayahara, G.J., and Ilisson, T. 2009. Wildfire promotes broadleaves and species mixture in boreal forest. Forest Ecology and Management **257**: 343-350.

Cox, D.R. 1972. Regression Models and Life Tables (with Discussion). Journal of the Royal Statistical Society, Series B **34**: 187-220.

Cumming, S.G. 2001. Forest type and wildfire in the alberta boreal mixedwood: What do fires burn? Ecological Applications **11**(1): 97-110.

Cumming, S.G. 2005. Effective fire suppression in boreal forests. Can. J. For. Res. -Rev. Can. Rech. For. **35**(4): 772-786.

Cwynar, L.C. 1977. Recent Fire History of Barron Township, Algonquin Park. Canadian Journal of Botany-Revue Canadienne De Botanique **55**(11): 1524-1538.

Cyr, D., Bergeron, Y., Gauthier, S., and Larouche, A. 2005. Are the old-growth forests of the Clay Belt part of a fire-regulated mosaic? Can. J. For. Res. -Rev. Can. Rech. For. **35**(1): 65-73.

Cyr, D., Gauthier, S., and Bergeron, Y. 2007. Scale-dependent determinants of heterogeneity in fire frequency in a coniferous boreal forest of eastern Canada. Landscape Ecology **22**(9): 1325-1339.

Ecoregions Working Group 1989. Ecoclimatic regions of Canada, first approximation.

Fauria, M.M. and Johnson, E.A. 2006. Large-scale climatic patterns control large lightning fire occurrence in Canada and Alaska forest regions. J. Geophys. Res. -Biogeosci. **111**(G4): 17. doi: G04008 10.1029/2006jg000181.

Flanagan, P.W. and Vancleve, K. 1983. Nutrient Cycling in Relation to Decomposition and Organic-Matter Quality in Taiga Ecosystems. Can. J. For. Res. -Rev. Can. Rech. For. **13**(5): 795-817.

Flannigan, M., Campbell, I., Wotton, M., Carcaillet, C., Richard, P., and Bergeron, Y. 2001. Future fire in Canada's boreal forest: paleoecology results and general circulation model - regional climate model simulations. Can. J. For. Res. -Rev. Can. Rech. For. **31**(5): 854-864.

Flannigan, M.D., Amiro, B.D., Logan, K.A., Stocks, B.J., and Wotton, B.M. 2005a. Forest Fires and Climate Change in the 21st Century. Mitigation and Adaptation Strategies for Global Change **11**: 847-859.

Flannigan, M.D., Bergeron, Y., Engelmark, O., and Wotton, B.M. 1998. Future wildfire in circumboreal forests in relation to global warming. Journal of Vegetation Science **9**(4): 469-476.

Flannigan, M.D. and Harrington, J.B. 1988. A Study of the Relation of Meteorological Variables to Monthly Provincial Area Burned by Wildfire in Canada (1953-80). Journal of Applied Meteorology **27**(4): 441-452.

Flannigan, M.D., Logan, K.A., Amiro, B.D., Skinner, W.R., and Stocks, B.J. 2005b. Future area burned in Canada. Clim. Change **72**(1-2): 1-16.

Flannigan, M.D. and Van Wagner, C.E. 1991. Climate Change and Wildfire in Canada. Can. J. For. Res. -Rev. Can. Rech. For. **21**(1): 66-72.

Foster, D.R. 1983. The History and Pattern of Fire in the Boreal Forest of Southeastern Labrador. Canadian Journal of Botany-Revue Canadienne De Botanique **61**(9): 2459-2471.

Frank, R.M. 1990. Abies balslamea (L.) Mill. - Balsam Fir. *In* Silvics of North America. Vol 1. Conifers. U.S.D.A. Forest Service, Washington, D.C. pp. 26-35.

Fricker, J.M., Chen, H.Y.H., and Wang, J.R. 2006. Stand age structural dynamics of North American boreal forests and implications for forest management. International Forestry Review **8**(4): 395-405.

Fryer, G.I. and Johnson, E.A. 1988. Reconstructing fire behavior and effects in a subalpine forest. Journal of Applied Ecology **25**(3): 1063-1072.

Gillett, N.P., Weaver, A.J., Zwiers, F.W., and Flannigan, M.D. 2004. Detecting the effect of climate change on Canadian forest fires. Geophysical Research Letters **31**(18) doi: L18211 Artn l18211.

Girardin, M.P., Tardif, J.C., Flannigan, M.D., and Bergeron, Y. 2005. Synoptic-Scale Atmospheric Circulation and Boreal Canada Summer Drought Variability of the Past Three Centuries. Journal of Climate **19**: 1922-1947.

Girardin, M.P., Tardiff, J.C., Flannigan, M.D., and Bergeron, Y. 2006a. Forest Fire-Conducive Drought Variability in the Southern Canadian Boreal Forest and Associated Climatology Inferred from Tree Rings. Canadian Water Resources Journal **31**(4): 275-296.

Girardin, M.P., Bergeron, Y., Tardif, J.C., Gauthier, S., Flannigan, M.D., and Mudelsee, M. 2006b. A 229-year dendroclimatic-inferred record of forest fire activity for the Boreal Shield of Canada. International Journal of Wildland Fire **15**(3): 375-388.

Girardin, M.P. and Mudelsee, M. 2008. Past and future changes in Canadian boreal wildfire activity. Ecological Applications **18**(2): 391-406.

Girardin, M.P., Tardif, J., Flannigan, M.D., Wotton, B.M., and Bergeron, Y. 2004a. Trends and periodicities in the Canadian Drought Code and their relationships with atmospheric circulation for the southern Canadian boreal forest. Can. J. For. Res. -Rev. Can. Rech. For. **34**(1): 103-119.

Girardin, M.P., Tardif, J., Flannigan, M.D., Wotton, B.M., and Bergeron, Y. 2004b. Trends and periodicities in the Canadian Drought Code and their relationships with atmospheric circulation for the southern Canadian boreal forest. Can. J. For. Res. -Rev. Can. Rech. For. **34**(1): 103-119.

Grenier, D.J., Bergeron, Y., Kneeshaw, D., and Gauthier, S. 2005. Fire frequency for the transitional mixedwood forest of Timiskaming, Quebec, Canada. Can. J. For. Res. -Rev. Can. Rech. For. **35**(3): 656-666.

Hart, S.A. and Chen, H.Y.H. 2008. Fire, logging, and overstory affect understory abundance, diversity, and composition in boreal forest. Ecological Monographs **78**(1): 123-140.

Heinselman, M.L. 1971. The natural role of fire in the northern conifer forests. *In* Fire in the northern environment. USDA Forestry Service, Portland. pp. 64-72.

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

Heinselman, M.L. 1981. Fire and succession in the conifer forests of northern North America. *In* Forest Succession: Concepts and Applications. Springer-Verlag, New York. pp. 374-405.

Hely, C., Bergeron, Y., and Flannigan, M.D. 2000. Effects of stand composition on fire hazard in mixed-wood Canadian boreal forest. Journal of Vegetation Science **11**(6): 813-824.

Heyerdahl, E.K., Brubaker, L.B., and Agee, J.K. 2001. Spatial controls of historical fire regimes: a multiscale example from the Interior West, USA. Ecology **82**: 660-678.

Ilisson, T. and Chen, H.Y.H. 2009a. Response of six boreal tree species to stand replacing fire and clearcutting. Ecosystems **12**: 820-829.

Ilisson, T. and Chen, H.Y.H. 2009b. The direct regeneration hypothesis in northern forests. Journal of Vegetation Science **20**(4): 735-744.

IPCC 2007a. Climate change 2007: impacts, adaptation and vulnerability. Cambridge University Press, New York, USA.

IPCC 2007b. Climate change 2007: the physical science basis. Cambridge University Press, New York, USA.

Jauhiainen, E. 1973. Effect if climate on podzolisation in southwest and eastern Finland. Commentat. Phys. Math. **43**: 214-242.

Jayen, K., Leduc, A., and Bergeron, Y. 2006. Effect of fire severity on regeneration success in the boreal forest of northwest Quebec, Canada. Ecoscience **13**(2): 143-151.

Jenny, H. 1941. Factors of Soil Formation. A System of Quantitative Pedology. McGraw-Hill, New York.

Johnson, E.A. 1992. Fire and vegetation dynamics - studies from the North American boreal forest. Cambridge University Press, Cambridge, UK.

Johnson, E.A., Fryer, G.I., and Heathcott, M.J. 1990. The Influence of Man and Climate on Frequency of Fire in the Interior Wet Belt Forest, British-Columbia. Journal of Ecology **78**(2): 403-412.

Johnson, E.A. and Gutsell, S.L. 1994. Fire Frequency Models, Methods and Interpretations. *In* Advances in Ecological Research, Vol 25. pp. 239-287.

Johnson, E.A., Miyanishi, K., and Bridge, S.R.J. 2001. Wildfire regime in the boreal forest and the idea of suppression and fuel buildup. Conservation Biology **15**(6): 1554-1557.

Johnson, E.A., Miyanishi, K., and O'Brien, N. 1999. Long-term reconstruction of the fire season in the mixedwood boreal forest of Western Canada. Canadian Journal of Botany-Revue Canadienne De Botanique 77(8): 1185-1188.

Johnson, E.A., Miyanishi, K., and Weir, J.M.H. 1995. Old-Growth, Disturbance, and Ecosystem Management. Canadian Journal of Botany-Revue Canadienne De Botanique **73**(6): 918-926.

Johnson, E.A., Miyanishi, K., and Weir, J.M.H. 1998. Wildfires in the western Canadian boreal forest: Landscape patterns and ecosystem management. Journal of Vegetation Science **9**(4): 603-610.

Johnson, E.A. and Van Wagner, C.E. 1985. The Theory and Use of 2 Fire History Models. Can. J. For. Res. -Rev. Can. Rech. For. **15**(1): 214-220.

Johnson, E.A. and Wowchuk, D.R. 1993. Wildfires in the Southern Canadian Rocky-Mountains and Their Relationship to Midtropospheric Anomalies. Can. J. For. Res. -Rev. Can. Rech. For. **23**(6): 1213-1222.

Kasischke, E.S., Christensen, N.L.J., and Stocks, B.J. 1995. Fire, global warming, and the carbon balance of boreal forests. Ecological Applications **5**: 437-451.

Kasischke, E.S. and Turetsky, M.R. 2006. Recent changes in the fire regime across the North American boreal region - Spatial and temporal patterns of burning across Canada and Alaska. Geophysical Research Letters **33**(9) doi: L09703 Artn l09703.

Kleinbaum, D.G. and Klein, M. 1996. Survival analysis: a self-learning text. Springer-Verlag, New York.

Krawchuk, M.A., Cumming, S.G., and Flannigan, M.D. 2009. Predicted changes in fire weather suggest increases in lightning fire initiation and future area burned in the mixedwood boreal forest. Clim. Change **92**(1-2): 83-97.

Krawchuk, M.A., Cumming, S.G., Flannigan, M.D., and Wein, R.W. 2006. Biotic and abiotic regulation of lightning fire initiation in the mixedwood boreal forest. Ecology **87**(2): 458-468.

Kurz, W.A. and Apps, M.J. 1995. An analysis of future carbon budgets of Canadian boreal forests.Kluwer Academic Publ. pp. 321-331.

Kurz, W.A. and Apps, M.J. 1999. A 70-year retrospective analysis of carbon fluxes in the Canadian forest sector. Ecological Applications **9**(2): 526-547.

Kurz, W.A., Stinson, G., Rampley, G.J., Dymond, C.C., and Neilson, E.T. 2008. Risk of natural disturbances makes future contribution of Canada's forests to the global carbon cycle highly uncerain. Proceedings of the National Academy of Sciences of the United States of America **105**(5): 1551-1555.

Larsen, C.P.S. 1997. Spatial and temporal variations in boreal forest fire frequency in northern Alberta. Journal of Biogeography **24**(5): 663-673.

Lauzon, E., Kneeshaw, D., and Bergeron, Y. 2007. Reconstruction of fire history (1680-2003) in Gaspesian mixedwood boreal forests of eastern Canada. For. Ecol. Manage. **244**(1-3): 41-49.

Le Goff, H., Flannigan, M.D., Bergeron, Y., and Girardin, M.P. 2007. Historical fire regime shifts related to climate teleconnections in the Waswanipi area, central Quebec, Canada. International Journal of Wildland Fire **16**(5): 607-618.

Lefort, P., Gauthier, S., and Bergeron, Y. 2003. The influence of fire weather and land use on the fire activity of the Lake Abitibi area, eastern Canada. Forest Science **49**: 509-521.

Lefort, P., Leduc, A., Gauthier, S., and Bergeron, Y. 2004. Recent fire regime (1945-1998) in the boreal forest of western Quebec. Ecoscience **11**(4): 433-445.

Legendre, P. and Legendre, L. 1998. Numerical Ecology. 2nd English edition ed. Elsevier Science.

Lesieur, D., Gauthier, S., and Bergeron, Y. 2002. Fire frequency and vegetation dynamics for the south-central boreal forest of Quebec, Canada. Can. J. For. Res. -Rev. Can. Rech. For. **32**(11): 1996-2009.

Lundstrom, U.S., van Breemen, N., and Bain, D. 2000. The podzolization process. A review. Geoderma **94**(2-4): 91-107.

Malanson, G.P. 1987. Diversity, stability and resilience: effects of fire regime. SPB Academic Publishing bv. pp. 49-63.

Martell, D.L. 1994. The impact of fire on timber supply in Ontario. For. Chron. **70**: 164-173.

Masters, A.M. 1990. Changes in Forest Fire Frequency in Kootenay-National-Park, Canadian Rockies. Canadian Journal of Botany-Revue Canadienne De Botanique **68**(8): 1763-1767.

Miyanishi, K., Bridge, S.R.J., and Johnson, E.A. 2002. Wildfire regime in the boreal forest - Reply. Conservation Biology **16**(5): 1177-1178.

Miyanishi, K. and Johnson, E.A. 2001. Comment - A re-examination of the effects of fire suppression in the boreal forest. Can. J. For. Res. -Rev. Can. Rech. For. **31**(8): 1462-1466.

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

Nash, C.H. and Johnson, E.A. 1996. Synoptic climatology of lightning-caused forest fires in subalpine and boreal forests. Can. J. For. Res. -Rev. Can. Rech. For. **26**(10): 1859-1874.

Newark, M.J. 1975. The Relationship between Forest Fire Occurrence and 500 mb Longwave Ridging. Atmosphere **13**: 26-33.

Nienstaedt, H. and Zasada, J.C. 1990. Picea glauca (Moench) Voss - White spruce. *In* Silvics of North America. Vol 1. Conifers. U.S.D.A. Forest Service, Washington, D.C. pp. 204-226.

Oliver, C.D. and Larson, B.C. 1990. Forest Stand Dynamics. 1st ed. McGraw-Hill Inc, New York. pp. 467.

Parisien, M.A., Hirsch, K.G., Lavoie, S.G., Todd, J.B., and Kafka, V.G. 2004. Saskatchewan Fire Regime Analysis. Natural Resources Canada, Canadian Forestry Service North. Rep. NOR-X-394.

Payette, S., Morneau, C., Sirois, L., and Desponts, M. 1989. Recent Fire History of the Northern Quebec Biomes. Ecology **70**(3): 656-673.

Perala, D.A. 1990. Populus tremuloides Michx - Quaking aspen. *In* Silvics of North America. Vol 2. Hardwoods. U.S.D.A. Forest Service, Washington D.C. pp. 555-569.

Phipps, R.L. 1986. Collecting, preparing, crossdating and measuring tree increment cores. U.S. Geological Survey Water-Resources Investigations. Rep. 85-4148.

Podur, J., Martell, D.L., and Knight, K. 2002. Statistical quality control analysis of forest fire activity in Canada. Can. J. For. Res. -Rev. Can. Rech. For. **32**(2): 195-205.

Price, C. and Rind, D. 1994. The impact of a 2xCO2 climate on lightning-caused fires. Journal of Climate 7(10): 1484-1494.

Qian, H., Klinka, K., and Kayahara, G.J. 1998. Longitudinal patterns of plant diversity in the North American boreal forest. Plant Ecology **138**(2): 161-178.

Reed, W.J., Larsen, C.P.S., Johnson, E.A., and MacDonald, G.M. 1998. Estimation of temporal variations in historical fire frequency from time-since-fire map data. Forest Science **44**(3): 465-475.

Romme, W.H. and Knight, D.H. 1981. Fire frequency and subalpine forest succession along a topographic gradient in Wyoming. Ecology **62**: 319-326.

Rowe, J.S. 1972. Forest Regions of Canada. Canadian Forestry Service. Rep. No 1300.

Rowe, J.S. and Scotter, G.W. 1973. Fire in the Boreal Forest. Quat. Res. **3**: 444-464.

Safford, L.O., Bjorkbom, J.C., and Zasada, J.C. 1990. Betula papyrifera Marsh - Paper birch. *In* Silvics of North America. Vol 2. Hardwoods. U.S.D.A Forest Service, Washington, D.C. pp. 158-171.

Skinner, W.R., Flannigan, M.D., Stocks, B.J., Martell, D.L., Wotton, B.W., Todd, J.B., Mason, J.A., Logan, K.A., and Bosch, E.M. 2002. A 500-hPa synoptic wildland fire climatology for large Canadian forest fires 1959-1996. Theoretical and Applied Climatology **71**: 157-169.

Skinner, W.R., Stocks, B.J., Martell, D.L., Bonsal, B., and Shabbar, A. 1999. The association between circulation anomalies in the mid-troposphere and the area burned by wildland fire in Canada. Theoretical and Applied Climatology **63**: 89-105.

Soil Classification Working Group 1998. The Canadian System of Soil Classification. 3rd ed. Agric. and Agri-Food Can. Publ. 1646 (Revised). 187 pp..

Soja, A.J., Tchebakova, N.M., French, N.H.F., Flannigan, M.D., Shugart, H.H., Stocks, B.J., Sukhinin, A.I., Varfenova, E.I., Chapin, F.S., and Stackhouse, P.W. 2007. Climate-induced boreal forest change: Predictions versus current observations. Global and Planetary Change **56**(3-4): 274-296.

Stocks, B.J. 1991. The extent and impact of forest fires in northern circumpolar countries. *In* Global biomass burning: atmospheric, climatic, and biospheric implications. MIT Press, Cambridge, MA.

Stocks, B.J. 1993. Global Warming and Forest-Fires in Canada. For. Chron. **69**(3): 290-293.

Stocks, B.J., Fosberg, M.A., Lynham, T.J., Mearns, L., Wotton, B.M., Yang, Q., Jin, J.Z., Lawrence, K., Hartley, G.R., Mason, J.A., and McKenney, D.W. 1998. Climate change and forest fire potential in Russian and Canadian boreal forests. Clim. Change **38**(1): 1-13.

Stocks, B.J., Mason, J.A., Todd, J.B., Bosch, E.M., Wotton, B.M., Amiro, B.D., Flannigan, M.D., Hirsch, K.G., Logan, K.A., Martell, D.L., and Skinner, W.R. 2002. Large forest fires in Canada, 1959-1997. Journal of Geophysical Research-Atmospheres **108**(D1) doi: 8149 Artn 8149.

Stocks, B.J., Wotton, B.M., Flannigan, M., Fosberg, M.A., Cahoon, D.R., and Goldammer, J.G. 2001. Boreal Forest Fire Regimes and Climate Change. *In* Remote Sensing and Climate Modeling: Synergies and Limitations. Springer Netherlands, pp. 233-246.

Suffling, R. and Molin, J.D. 1982. Estimating past forest age distributions and disturbance rates in Northwestern ontario - a demographic approach. Journal of Environmental Management **14**(1): 45-56.

Tande, G.F. 1979. Fire history and vegetation pattern of coniferous forests in Jasper NAtional Park, Alberta. Canadian Journal of Botany-Revue Canadienne De Botanique **57**: 1912-1931.

Tardif, J. 2004. Fire history in the Duck Mountain Provincial Forest, Western Manitoba. Project Report. Sustainable Forest Management Network.

Timoney, K.P. 2003. The changing disturbance regime of the boreal forest of the Canadian Prairie Provinces. For. Chron. **79**(3): 502-516.

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

Tymstra, C., Flannigan, M.D., Armitage, O.B., and Logan, K. 2007. Impact of climate change on area burned in Alberta's boreal forest. International Journal of Wildland Fire **16**(2): 153-160.

Van Wagner, C.E. 1978. Age-Class Distribution and Forest Fire Cycle. Can. J. For. Res. -Rev. Can. Rech. For. **8**(2): 220-227.

Van Wagner, C.E. 1987. The development and structure of the Canadian forest fire weather index system. Canadian Forest Service, For Tech Rep 35, Ottawa, Ontario.

Vanwagner, C.E. 1983. Simulating the Effect of Forest Fire on Long-Term Annual Timber Supply. Can. J. For. Res. -Rev. Can. Rech. For. **13**(3): 451-457.

Vasiliauskas, S.A. and Chen, H.Y.H. 2002. How long do trees take to reach breast height after fire in northeastern Ontario? Canadian Journal of Forest Research **32**: 1889-1992.

Viereck, L.A. and Johnson, W.F. 1990. Picea mariana (Mill) B.S.P. - Black Spruce. *In* Silvics of North America. Vol 1. Conifers. U.S.D.A. Forest Service, Washington D.C. pp. 227-237.

Vierick, L.A. 1983. The effects of fire in black spruce ecosystems of Alaska and northern Canada. *In* The Role of Fire in Northern Circumpolar Ecosystems. Wiley, New York. pp. 201-220.

Ward, P.C. and Tithecott, A.G. 1993. The impact of fire management on the boreal landscape of Ontario. Queens Printer for Ontario. Rep. Aviation, Flood and Fire Management Branch Publication No. 305.

Weber, M.G. and Flannigan, M. 1997. Canadian boreal forst ecosystem structure and function in a changing clime: impact on fire regimes. Environmental Review **5**: 145-166.

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

Weir, J.M.H. and Johnson, E.A. 1998. Effects of escaped settlement fires and logging on forest composition in the mixedwood boreal forest. Canadian Journal of Forest Research **28**: 459-467.

Weir, J.M.H., Johnson, E.A., and Miyanishi, K. 2000. Fire frequency and the spatial age mosaic of the mixed-wood boreal forest in western Canada. Ecological Applications **10**(4): 1162-1177.

Whelan, R.J. 1995. The Ecology of fire. Cambridge University Press, New York.

Willis, K.J., Sumegi, P., Braun, M., and Toth, A. 1995. The Late Quaternary Environmental History of Batorliget, Ne Hungary. Palaeogeography Palaeoclimatology Palaeoecology **118**(1-2): 25-47.

Woods, G.T. and Day, R.T. 1977. A fire history study of Quetico Provincial Park. Report No. 4, Fire Ecology Study, Quetico Provincial Park. Ontario Ministry of Natural Resources.

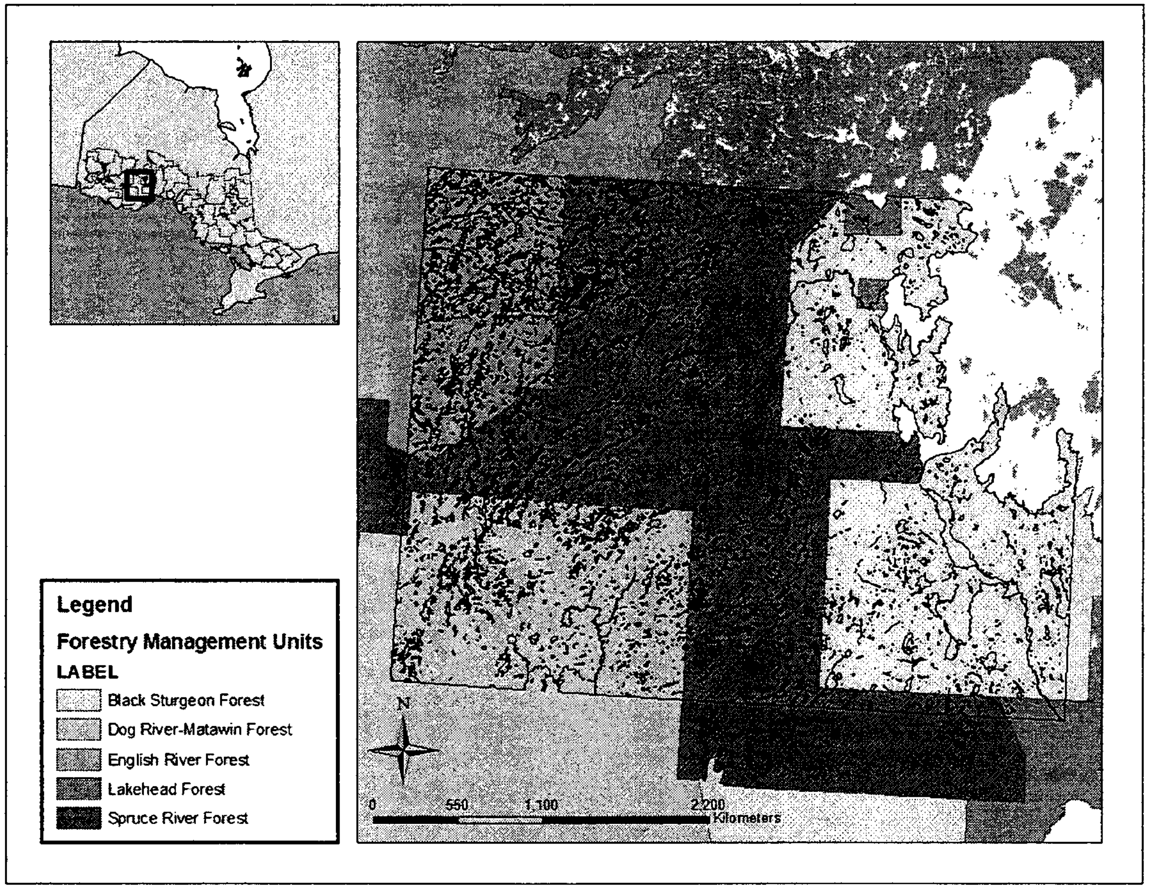
Wotton, B.M. and Flannigan, M.D. 1993. Length of the Fire Season in a Changing Climate. For. Chron. **69**(2): 187-192.

Wotton, B.M., Martell, D.L., and Logan, K.A. 2003. Climate change and people-caused forest fire occurrence in Ontario. Clim. Change **60**(3): 275-295.

Xiao, J.F. and Zhuang, Q.L. 2007. Drought effects on large fire activity in Canadian and Alaskan forests. Environmental Research Letters **2**(4).

Zasada, J.C., Sharik, T.L., and Nygren, M. 1992. The reproductive process in boreal forest trees. Cambridge University Press, Cambridge, U.K. pp. 85-125.

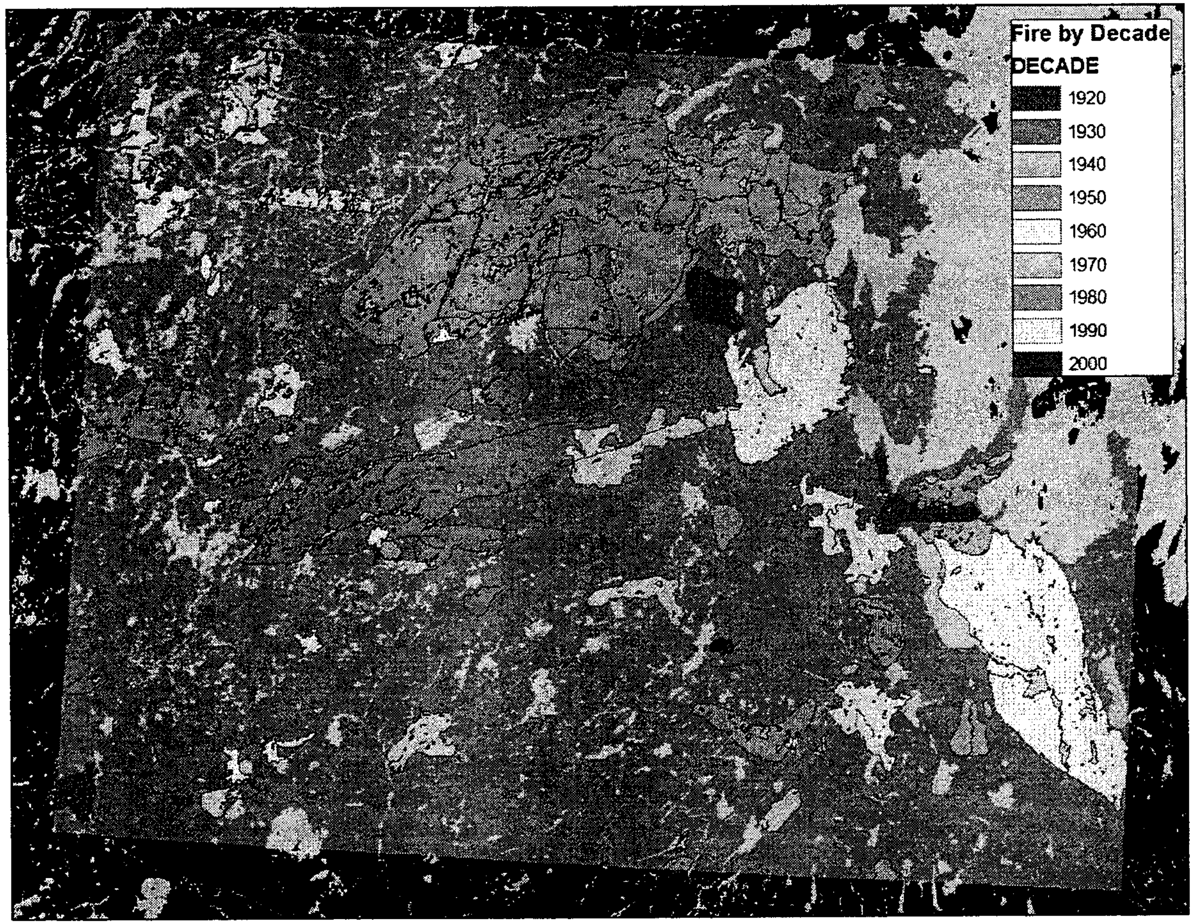
APPENDIX 1 Ontario Forestry Management Units (FMUs) in relation to the study area



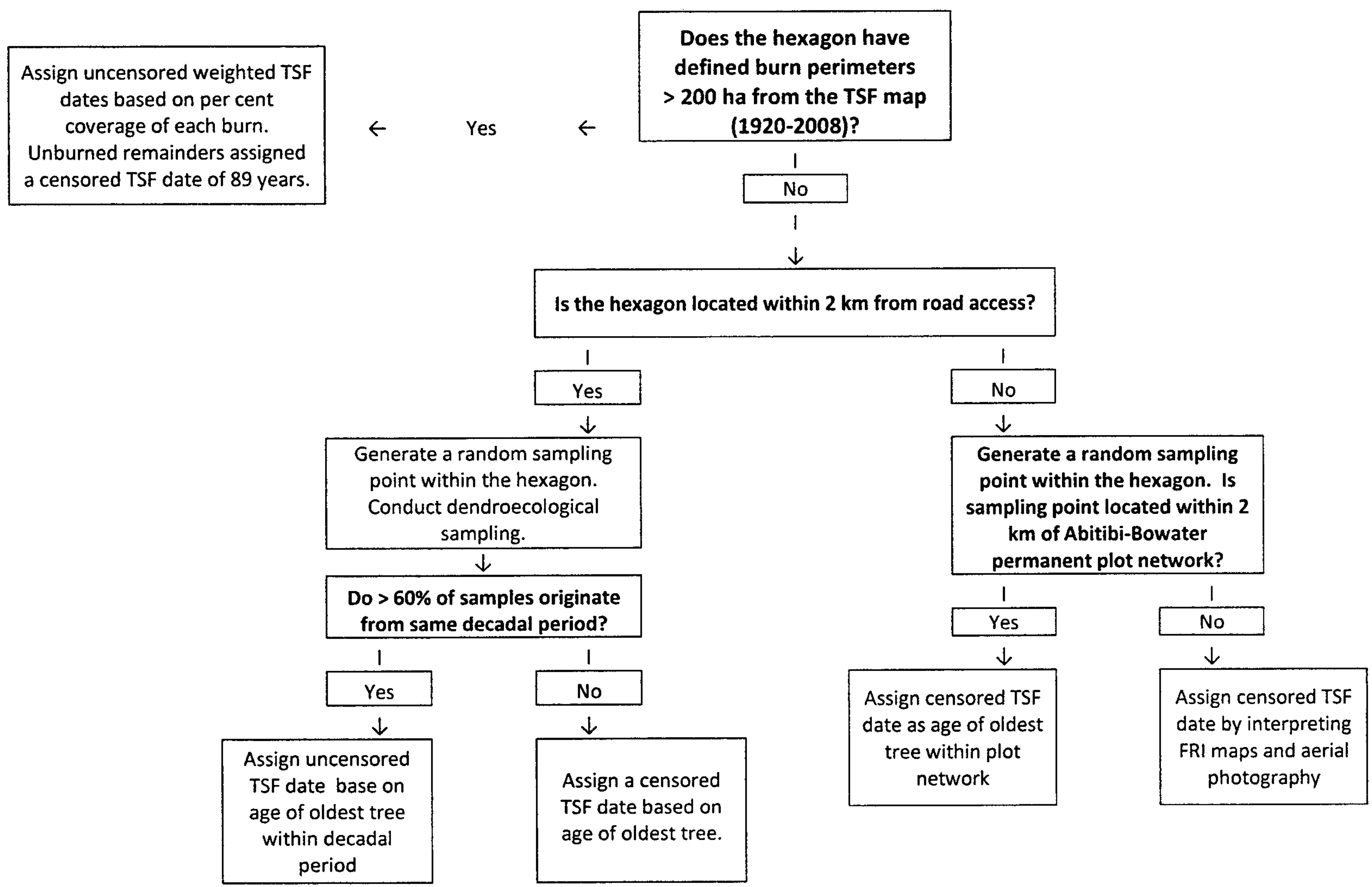
APPENDIX 2 Spearman's correlation matrix of variables considered in analysis

| | Variable | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | Ecodistrict | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 2 | Latitude | 0.12 | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 3 | Longitude | **-0.83** | -0.19 | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 4 | Soil Order | 0.22 | 0.28 | -0.28 | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 5 | Beach and Aeolian Deposit | 0.20 | -0.32 | -0.32 | 0.24 | * | * | * | * | * | * | * | * | * | * | * | * |
| 6 | Organic Deposit | -0.36 | 0.32 | 0.29 | 0.16 | -0.09 | * | * | * | * | * | * | * | * | * | * | * |
| 7 | Outwash Deposit | 0.05 | -0.18 | -0.03 | -0.22 | -0.10 | -0.15 | * | * | * | * | * | * | * | * | * | * |
| 8 | Ground Moraine | 0.12 | 0.06 | -0.03 | -0.07 | -0.35 | -0.25 | -0.29 | * | * | * | * | * | * | * | * | * |
| 9 | Mean Firebreak Distance | -0.29 | -0.29 | 0.36 | -0.25 | 0.08 | 0.21 | 0.15 | -0.30 | * | * | * | * | * | * | * | * |
| 10 | Aspect (x) | -0.10 | -0.10 | 0.16 | -0.14 | -0.07 | -0.06 | -0.03 | 0.10 | 0.03 | * | * | * | * | * | * | * |
| 11 | Aspect (y) | -0.04 | 0.02 | 0.05 | -0.01 | -0.07 | -0.07 | -0.06 | 0.13 | 0.07 | -0.02 | * | * | * | * | * | * |
| 12 | Elevation | **0.73** | -0.26 | **-0.77** | 0.10 | 0.35 | -0.43 | 0.13 | 0.03 | -0.24 | -0.05 | -0.07 | * | * | * | * | * |
| 13 | Coniferous Dominated | -0.04 | -0.06 | 0.09 | -0.02 | 0.01 | 0.02 | 0.02 | -0.04 | 0.05 | -0.01 | -0.03 | 0.00 | * | * | * | * |
| 14 | Mixedwood Dominated | 0.04 | 0.06 | -0.09 | 0.02 | -0.01 | -0.02 | -0.02 | 0.04 | -0.05 | 0.01 | 0.03 | 0.00 | -1.00 | * | * | * |
| 15 | Drainage Class: Extreme | 0.00 | **0.71** | -0.08 | **0.57** | -0.14 | 0.27 | -0.23 | 0.04 | -0.31 | -0.16 | 0.01 | -0.23 | -0.03 | 0.03 | * | * |
| 16 | Drainage Class: Very | 0.22 | -0.11 | -0.21 | 0.16 | 0.05 | -0.04 | 0.05 | -0.04 | -0.16 | 0.03 | 0.02 | 0.30 | 0.02 | -0.02 | -0.21 | * |
| 17 | Drainage Class: Moderate | -0.14 | **-0.56** | 0.21 | **-0.62** | 0.09 | -0.22 | 0.17 | -0.02 | 0.39 | 0.13 | -0.03 | 0.00 | 0.02 | -0.02 | **-0.76** | -0.48 |

APPENDIX 3 Time-Since-Fire (TSF) map for the study area showing fire by decade between 1920-2008



APPENDIX 4 Flowchart: assignment of TSF dates within a hexagon



APPENDIX 5 Sampling sheet used at dendroecological sampling points

| | |
|---|---|
| **Hexagon #** | |
| Coordinates | |

| | |
|---|---|
| Stumps (Y/N) | |
| Charcoal (Y/N) | |
| Sample (Y/N) | |

| | |
|---|---|
| S-type | |
| Mesic/xeric/hydric | |
| V-type | |
| Elevation | |
| NOTES: | |

| Core # | Species | Year |
|---|---|---|
| **1** | | |
| **2** | | |
| **3** | | |
| **4** | | |
| **5** | | |
| **6** | | |
| **7** | | |
| **8** | | |
| **9** | | |
| **10** | | |
| 11 | | |
| 12 | | |
| 13 | | |
| 14 | | |
| 15 | | |

| Disk # | Species | Year |
|---|---|---|
| **1** | | |
| **2** | | |
| **3** | | |
| **4** | | |
| **5** | | |
| **6** | | |
| **7** | | |
| **8** | | |
| **9** | | |
| **10** | | |
| 11 | | |
| 12 | | |
| 13 | | |
| 14 | | |
| 15 | | |