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Spatiotemporal Variations of Fire Frequency in Central Boreal Forest

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ABSTRACT

Determination of the direct causal factors controlling wildfires is key to understanding wildfire–vegetation–climate dynamics in a changing climate and for developing sustainable management strategies for biodiversity conservation and maintenance of long-term forest productivity. 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 temporal and spatial factors. We reconstructed the fire history of an 11,600-km^2 area located in the northwestern boreal forest of Ontario, using archival data of large fires occurring since 1921 and dendrochronological dating for fires prior to 1921. The fire cycle decreased from 295 years for the period of 1820–1920 to approximately 100 years for the period of 1921–2008. Spatially, fire frequency increased with latitude, attributable to higher human activities that have increased fragmentation and fire suppression in the southern portion of the study area. Fire frequency also increased with distance to waterbodies, and was higher on Podzols that were strongly correlated with moderate drainage and coniferous vegetation. The temporal increase of fire frequency in the central region, unlike western and eastern boreal forests where fire frequency has decreased, may be a result of increased warm and dry conditions associated with climate change in central North America, suggesting that the response of wildfire to global climate change may be regionally individualistic. The significant spatial factors we found in this study are in agreement with other wildfire studies, indicating the commonality of the influences by physiographic features and human activities on regional fire regimes across the boreal forest. Overall, wildfire in the central boreal shield is more frequent than that in the wetter eastern boreal region and less frequent than that in the drier western boreal region, confirming a climatic top-down control on the fire activities of the entire North American boreal forest.

Key words: fire frequency; boreal forest; time-since-fire; survival analysis; temporal pattern; latitude; soil order; distance to fire breaks.

INTRODUCTION

In the North American boreal forests, wildfire is the most dominant natural disturbance (Johnson 1992; Stocks 1993) and plays a vital role in maintaining biodiversity and essential forest ecosystem functions (Johnson and others 1995). With fire regimes expected to change with global climate change in these forests (Chapin and others 2000; Bond-Lamberty

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and others 2007), determination of the direct causal factors controlling wildfire at multiple spatial and temporal scales may shed light on wildfire–vegetation–climate dynamics in a changing climate (Parisien and Moritz 2009; Chen and others 2009; Ilisson and Chen 2009; Johnstone and others 2010). Furthermore, understanding of regional fire regimes is essential to the development of sustainable management strategies for biodiversity conservation and maintenance of long-term forest productivity (Bergeron and others 2002; Simard and others 2007).

Across the North American boreal forest, reconstruction of historical fire regimes shows a higher fire frequency in western (for example, Johnson and others 1998; Weir and others 2000) than eastern boreal forest (for example, Lefort and others 2004; Bergeron and others 2006), attributable to the interplay of temperature and precipitation in the respective regions (Krawchuk and others 2009; Parisien and Moritz 2009). Temporally, changes in the incidence of humid air masses and precipitation following the end of the Little Ice Age (approximately 1850 AD) have been correlated with a decrease in fire frequency in eastern (Bergeron and Dansereau 1993; Bergeron and others 2004b) and western boreal forests (Larsen 1996; Weir and others 2000). Although infrequent, increases in boreal fire frequency in the past century have been reported (Chapin and others 2000). Within a region, spatial variations of fire regime appear to be regulated by physiographic features such as slope aspect and drainage that alter fuel moisture levels thereby moderating whether local ignitions become large fire events (Payette and others 1989; Larsen 1997; Lefort and others 2004; Cyr and others 2007); temporal variations in fire regime seem to be regulated by direct effects of climatic change associated shifts in temperature and precipitation, indirect effects of climatic change on vegetation dynamics and land-use changes associated with human activities such as timber harvesting and fire suppression (Clark 1988; Bergeron 1991; Clark and Royall 1996; Weir and others 2000).

The central boreal shield represents the transition between the eastern and western boreal forests and has one of the highest plant species diversities in the North American boreal forest (Qian and others 1998). The high plant species diversity is attributable to possibly an intermediate frequency of wildfire, a predominant disturbance type in the boreal forest, as predicted by the intermediate disturbance hypothesis (Connell 1978). Despite the ecological and economical importance of the central boreal forest, its fire regime is largely unknown although it is often assumed to be intermediate to those reported in western and eastern boreal forests (Brassard and Chen 2008; Hart and Chen 2008). 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 temporal and spatial factors. We hypothesized that the fire frequency of the studied forest would be in-between those documented in the eastern and western boreal regions because of the climate and vegetation associated with its intermediate geographical position in the boreal zone. Within the region, we hypothesized a temporal increase in regional fire activities for two reasons: (i) dissimilar to other boreal regions, increases in the incidence of dry spell extremes have been reported in the western boreal shield of Ontario (Beverly and Martell 2005) and nearby northwestern Minnesota (Clark 1990), the occurrence of which are strongly correlated with large fire (Girardin and others 2006a), and (ii) human activities in the region have increased throughout the 20th century, resulting in more human-caused fire ignitions. Furthermore, we hypothesized that topographic features through influencing the presence of flammable vegetation may exert large-scale controls on regional wildfire activities (Cyr and others 2007; Parisien and Moritz 2009).

METHODS

Study Area

The study was conducted north of Thunder Bay, Ontario in the mixedwood boreal forest. The study area (49–50°N; 88.5–90.5°W) lies in the northwestern commercial forests of Ontario, within the Lake Nipigon ecoregion, and covers approximately 11,600 km^2 (Figure 1). The area is within the Moist Mid-Boreal (MBx) ecoclimatic region (Ecoregions Working Group 1989), characterized by a strong continental climate with long winters and short summers. The mean summer temperature within the Lake Nipigon ecoregion is typically 14°C (ranging from 11 to 16°C), with the mean winter temperature averaging −13°C. Mean annual precipitation ranges from 700 to 800 mm. The study area is subject to relatively cool summer and warm winter temperatures, due to the moderating lake effect of both Lake Superior and Lake Nipigon (Ecoregions Working Group 1989).

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

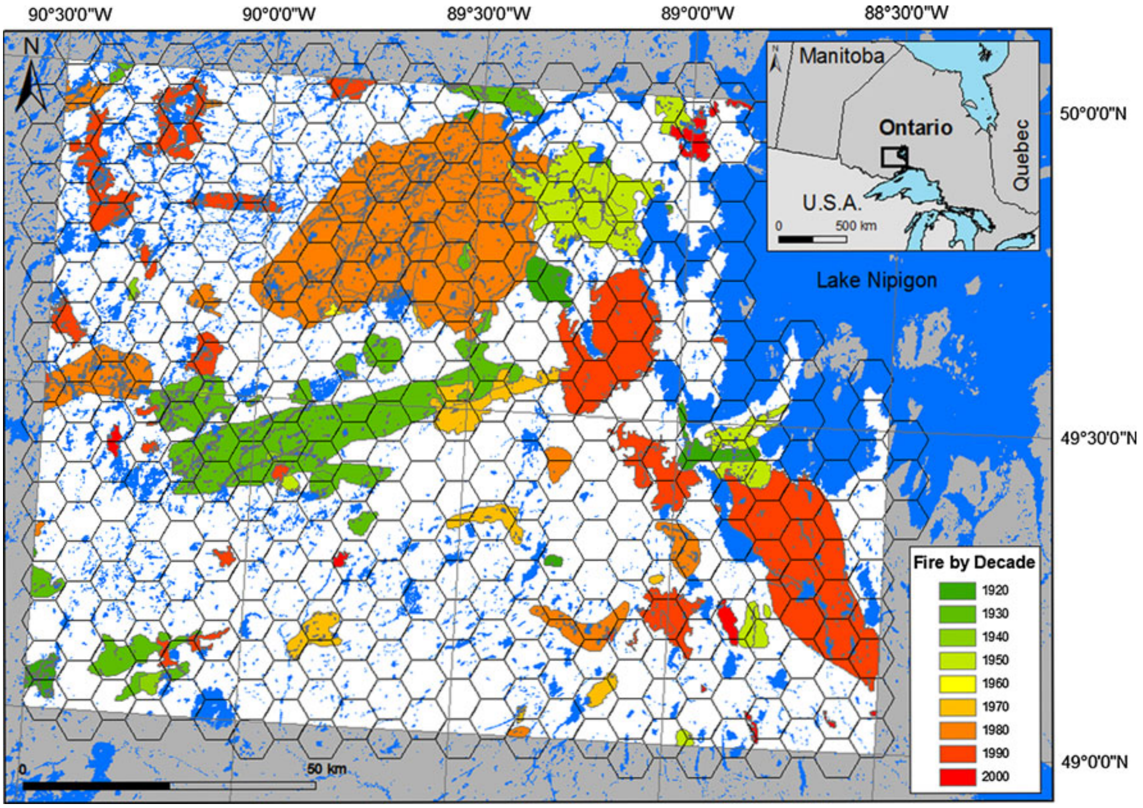

Figure 1. Location of the study area within the central boreal region of northwestern Ontario, Canada.

relief in other parts of the region. Surficial deposits are 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 majority of soils in the area belong to either dystric brunisol or humo-ferric podzol soil orders. Humo-ferric podzol dominates the central and southeastern portion of the study area with the primary concentration of dystric brunisol occurring in the southwest and northern portions, particularly in the extensive sand and gravel deposits of the plateau surrounding the Lake Nipigon basin (Figure 1). Podzolic soils in the area develop under coniferous forest stands in moderately well-drained sites on coarse-textured, stony, glacial tills and outwash deposits on acidic parent material (Soil Classification Working Group 1998; Baldwin and others 2001; Soil Landscapes of Canada Working Group 2007). Brunisols are more closely associated with acidic bedrock and loamy to sandy acidic glacial till, with outwash and lacustrine materials (Baldwin and others 2001) on sites of excessively to very well-drained drainage classes (Soil Landscapes of Canada Working Group 2007).

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 black spruce (*Picea mariana*), white spruce (*P. glauca*), and balsam fir (*Abies balsamea*) replacing the pioneer species over time (Chen and Popadiouk 2002; Hart and Chen 2006; Brassard and others 2008). Forest composition is primarily mixedwood with a varying percentage of conifer species on uplands; lowlands are dominated by black spruce (Chen and Popadiouk 2002).

The region has an extensive history of human activities including forest harvesting and management, forest fire suppression, and recreational activities. Timber harvesting began early in the 20th century (1910–1920), primarily in the southern half of the study area encroaching northward over time. In recent decades forestry activity has shifted towards the northwestern portion of the study area and the areas further north of the study area.

Fire History

Fire history of the study area was reconstructed by using existing fire records and dendroecological sampling. For fires occurring between 1921 and 1995, Ontario's Forest Fire History, a record of the location and spatial extent of large fires (≥200 ha), was used. All fires between 1996 and 2008 were mapped from provincial records, aerial photography, and satellite imagery. For fires occurring prior to 1921, we used dendroecological sampling techniques (Bergeron 1991) to assess the amount of time elapsed since the most recent fires.

To take full advantage of the available fire records and focus our efforts on ground sampling where fire history was unknown, for example, areas not burned during 1921–2008, a preliminary time since fire (TSF) map of the study area was constructed in ArcGIS 9.2 (ESRI, Redlands, CA), based on the archival data, dendrochronological analysis, permanent sample plots, and forest resource inventory. Samples for dendrochronological analysis were then taken in the study area according to a systematic random plan. A systematic hexagonal grid system was used to create a consistent gapless network throughout the study area, resulting in 407 hexagon sampling units, each measuring an area of 40 km^2 (Figure 1).

Of the 407 hexagons, 220 (54%) were assigned TSF dates from the preliminary TSF map generated from recent fire records. Because not all hexagons were completely burned by a single fire, TSF determination for partially burned hexagons was weighted based on the proportional areas burned by fires and the terrestrial land area within each hexagon. Among the 187 hexagons with no TSF determination from the preliminary TSF map, 102 hexagons that were located within 2 km from road access were visited and sampled for dendroecological analysis. The 85 hexagons located greater than 2 km from road access were deemed inaccessible. Of the 85 inaccessible hexagons, we were able to estimate TSFs based on stand age for 16 hexagons by using the Abitibi-Bowater Continuous Forest Inventory plots that were sufficiently close (<2 km) (sensu Le Goff and others 2007). For the remaining 69 inaccessible hexagons, we derived TSF estimates from Abitibi-Bowater Forest Resource Inventory (FRI) maps by assuming the stands originated from stand replacing fires.

Field Sampling

For the 102 (25% of the data) hexagons that required field sampling, a single random point was generated within the terrestrial land area of each hexagon. 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 organic interface for charcoal. In total fire-scarred trees were observed at two sites, whereas charcoal was observed at the base of the humus layer in all sites sampled.

Within each of the field sampled hexagons, a 200 m transect with a random starting point and random direction was laid out according to the point-centered quarter method (Mueller-Dombois and Ellenberg 1974). Increment cores were extracted from the 10 largest canopy trees within 30 m along the transect. Preference was given to the largest canopy trees of pioneer species that regenerate immediately post-fire in the order of jack pine, paper birch, and trembling aspen, and black spruce. One increment core or disk was taken per tree depending on the diameter and species of tree sampled (a disk was taken if the 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 to minimize underestimation of tree age (Phipps 1986).

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 and others 2008). The TSF for the hexagon was estimated by using the age of 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 10 year period if the sample composition was greater than 80% jack pine, and 20 year period for all other species (sensu Lesieur and others 2002; Bergeron and others 2004b). These differential decadal periods were selected to account for missing rings and post-disturbance regeneration delay that may be present in shade-tolerant or mixed species stands that could underestimate the true age (Vasiliauskas and Chen 2002; Girardin and others 2006b). When samples did not meet the requirement of at least 60%, the TSF for the hexagon was determined by using the age of the oldest tree sampled as a minimum estimate. We assumed the samples captured the time-since-last-fire and as such were assigned full weight.

Sampling points were inspected for evidence of harvesting by searching for cut stumps; if a cut stump was present at the sampling point a new random point was generated and visited for sampling. When no accurate TSF determination was possible and the area was not visited, we assigned a minimum TSF of 89 years to complete the coverage of the study area (sensu Drever and others 2006). A total of 32 hexagons (8%) were assigned in this manner.

Statistical Analysis

Several methods were used to estimate fire frequency and fire cycle (FC). Fire frequency (burn rate) is defined as the annual percent area burned within a study area. The fire cycle, that is, average fire return interval, is the inverse of fire frequency, and is the time required to burn an area equal in size to the studied area. For the period when fire records were available (1921–2008), FC was calculated following the methodology of Heinselman (1973) by dividing the total terrestrial land area by the average annual area burned during the time period covered (1921–2008). This 'burn rate' method of calculating fire cycle did not allow us to estimate pre-1921 fire frequency.

Fire cycles were also calculated for three different time periods using the Cox regression model (see description below) using fire records, dendrochronological, permanent sample plot, and FRI data. The first time period, from 1820 to 1920, covers the time from the end of the Little Ice Age (1850 AD) to the beginning of forestry activities in the study area. The second, from 1921 to 2008, covers recent history in which we suspect fire frequency has shifted. The third, from 1820 to 2008, covers the length of the reconstruction. Calculations were made using the *coxph* and *basehaz* functions in R (R Development Core Team 2009) to compute the Cox regression model and extract the estimated cumulative hazard function, which is the accumulation of hazard over time, using the Nelson–Aalen estimator (Tsiatis 1981). The hazard function or 'burn rate' refers to the instantaneous probability of fire and is statistically equivalent to fire frequency. Calculations of the accumulative hazard allow us to estimate the fire cycle by averaging the hazard over a designated time period (Johnson and Gutsell 1994) and calculating the inverse.

Survival analysis is a statistical tool used to take censored data into consideration (Klein and Molin 2005). A censored observation is a minimum estimate of the true time-since-fire used when the last-fire event is difficult to date with a satisfying level of confidence. This is relevant as there were many censored TSF estimates due to the following reasons: (1) inaccessibility necessitating TSF estimates from permanent sample plots and FRI maps, (2) minimum 89 years of TSF assigned to the unburned and unvisited portions of a hexagon, and (3) cohort sampling within a hexagon having lower than 60% representation from the same decadal period. The final data set included 375 hexagons consisting of 674 weighted TSF observations ($n = 674$), with 40% of the data censored.

The influence of environmental variables on fire frequency was examined for the entire temporal range using the Cox proportional hazards regression model (Cox 1972), commonly known as Cox regression, which is semi-parametric, giving it a high level of robustness given that the baseline hazard of burning function does not have to be specified beforehand, as in the case of parametric regressions (Klein and Molin 2005). Although the resulting estimates are not as efficient as maximum-likelihood estimates for a correctly specified parametric regression model (for example, negative exponential or weibull), Cox regression compensates for this by not having to make assumptions about whether or not hazard of burning increases with stand age prior to data analysis as the baseline hazard of burning is derived from the empirical TSF distribution. The data were fit using the *coxph* procedure in R version 2.9.2 (R Development Core Team 2009) using the 'survival' package (Therneau 2009).

Although fire regimes at the large regional and continent scales are determined by large regional differences in climate and coupled vegetation and sources of ignitions, fire activities in a landscape where climate is relatively uniform tend to be driven by topography, weather, and fuel conditions (Parisien and Moritz 2009). Predictors included soil order and surficial deposit (coded as dummy variables), latitude and longitude (in decimal degrees), drainage, elevation, aspect, and distance to fire breaks (Table 1). Soil order, surficial deposit, elevation, aspect, and drainage class were adapted from *Soil Landscapes of Canada*, a series of GIS coverages that show the major characteristics of soil and land of Canada (Soil Landscapes of Canada Working Group 2007). Firebreak distance was calculated as the mean distance to a waterbody defined as any river or lake from the randomly generated points for field sampling within each hexagon, measured in kilometers, along the four cardinal directions and their intermediates on a 1:250,000 topographic map. To incorporate aspect, a continuous variable, which varies within a circular scale, into the linear context of the Cox regression model, we converted the aspect class (north, north–east, east, and so on) into x and y coordinates following the methodology described in Legendre and Legendre (1998) and applied in a similar context by Cyr and others (2007). 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 east–west axis, whereas the

Table 1. Characteristics of the Environmental Variables Considered in the Survival Analysis

Vegetation	Stand type[1]	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[2]	Ground moraine	Broad scale	Nominal
		End moraine	Broad scale	
		Outwash deposit	Broad scale	
		Beach and aeolian deposit	Broad scale	
	Soil order[2]	Dystric brunisol	Broad scale	Nominal
		Humo-ferric podzol	Broad scale	
	Drainage[2]	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[3]		Intermediate	Continuous (m)
	Elevation		Local scale	Continuous (m)

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[1] *Stands were categorized according to greater than 75% basal area composition belonging to either coniferous or deciduous vegetation. Stands in which neither coniferous nor deciduous vegetation account for 75% or more total basal area were categorized as mixedwood. Collinearity with other variables was tested, however, stand type was not selected as an explanatory variable for cox regression analysis.*

[2] *Evaluated from data present in Soil Landscapes of Canada Working Group (2007)*

[3] *The mean firebreak distance considered the eight nearest waterbodies measured along all cardinal directions and their intermediates from the random sampling point in ArcGIS 9.2.*

vertical axis is 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 $\sin(45°) = \cos(45°) \approx 0.7071$.

To assess possible collinearity among the variables, Spearman's rank correlation coefficients were calculated allowing us to discard several variables as their probable effects on fire frequency could be explained by another more ecologically appropriate variable. Collinearity between dominant stand type (coniferous, deciduous or mixedwood) and the other variables were evaluated, but stand type was not submitted to survival analysis as it is not a permanent landscape feature. Most of the correlations among variables were less than 0.30 or greater than −0.30 (Appendix 1 in Supplemental Materials), with the exception of longitude and elevation ($r = 0.73$); soil order and drainage classes extremely well and moderately well-drained ($r = -0.56, -0.71$); and drainage class extremely well-drained and latitude ($r = -0.76$). Longitude and elevation are closely related in the study area 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 whereas sites of humo-ferric podzol were associated with the moderately well-drained drainage class. Finally, in the study area excessive drainage generally corresponded with higher latitudes along the northern border whereas moderately drained soils were mostly distributed in the south and central regions. Based on our correlation matrix, we thus eliminated drainage class from our survival model in favor of latitude and soil order. Latitude remained because it approximates a temporal gradient of human activities occurring at different times in the study area, that is, progressive movement of human activities such as harvesting from south to north since 1910 (Paul Poschmann, pers. comm.). Soil order remained as it accounts for drainage class, parent material, vegetation type, and climate. Additionally, unlike drainage class, soil order was not significantly correlated with latitude.

To test the influence of individual variables on fire frequency, we first modeled each factor individually, we then performed multiple Cox regressions (full model) which included all variables.

RESULTS

Total Area Burned (1921–2008)

The total area burned (including areas burned multiple times) between 1921 and 2008 covers 645,213 ha, or approximately 55% of the study area. According to the TSF map, during the 1921–2008 time period (Figure 1), fires burned an average of 7332 ± 35,503 ha (mean ± 1 SD) or about 0.6% of the landscape per year. The fire cycle for the 1921–2008 time period as calculated using the burning rate method was 158 years (Table 2). There was significant interannual variability in area burned (1921–2008) by large fires, ranging from 0 ha in some years to a high of 143,000 ha in 1980; cumulatively, the 1960s were the least active decade with approximately 400 ha burned whereas the most active decade were the 1980s with approximately 181,000 ha burned.

Based on non-censored fire records (Figures 1, 2), there were two periods of high fire activities: one during the period of 1921–1930 and the other during the period of 1981–1990. Based on the age distribution of the landscape that included both censored and non-censored data, approximately 35% of the stands were established during the decades of 1921–1930 and approximately 20% were burned during 1981–1990.

Table 2. Fire Cycle Calculations (in Years) for Three Time Periods

Method	Time period investigated		
	1820–2008	**1820–1920**	**1921–2008**
Burning rate	N/A	N/A	158
Inverse hazard	150 (126–188)[1]	295 (229–499)	96 (84–111)

[1] *95% CI in parentheses.*

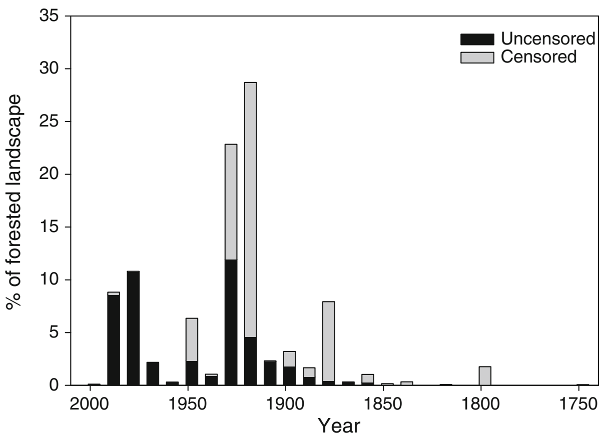

Figure 2. Forest age distribution for the study area 1756–2008 presented as the proportion of forested area burned by decade.

Temporal Variation of Fire Frequency

Survival analysis revealed a fire cycle of 150 years for the entire period (1820–2008) (Table 2; Figure 2). The more recent time period (1921–2008) had a fire cycle of 96 years, whereas the 1820–1920 time period had a fire cycle of 295 years. The two fire cycles differed significantly because their conference intervals did not overlap (Table 2).

Spatial Variation of Fire Frequency

When individual variables were tested in the Cox regression models, dominant soil order and firebreak distance were significantly related to fire frequency (Table 3). After incorporating all variables in the full model, latitude in addition to dominant soil order and firebreak distance were also significantly related to fire frequency.

The fire frequency (FF) ratios, also called hazard ratios in survival analysis literature, resulting from the Cox regressions, were used to quantify the influence of each variable on fire frequency. For indicator (dummy) variables, 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 variables). For quantitative variables the FF ratios give the estimated percent change in the burn rate for each one-unit increase in the variable. The ratio of 1.78 in the full model associated with soil order indicates that fire frequency is 78% greater on humo-ferric podzol soils than on dystric brunisol soils (Table 3; Figure 3B).

Latitude had a FF ratio of 2.2 in the full model (Table 3; Figure 3C), indicating an increasing gradient in fire frequency from south to north, as for each 1° increase in latitude the burn rate increased by an estimated 120%. Distance to firebreaks had a FF ratio of 1.38 (Table 3; Figure 3D), indicating that for each 1 km increase in distance from waterbodies, fire frequency increased by 38%.

DISCUSSION

This study represents the first attempt to quantify fire regimes of the central boreal mixedwood region. Our results indicate the fire cycle has changed temporally and varies spatially in the large landscape we studied. Changes in climate have influenced patterns of drought severity throughout the 20th century (Skinner and others 2002; Girardin and others 2006b), which have been correlated with decreases in fire frequency evident in both eastern (Bergeron and Dansereau 1993; Bergeron and others 2004b) and western boreal forests

Table 3. Summary of the Variables in the Cox Proportional Hazards Models

Variable	Individual model		Full model	
	Prob > χ^2	Fire frequency ratio	Prob > χ^2	Fire frequency ratio
Latitude	0.092	1.523	0.041	2.203
Longitude	0.605	0.929	0.105	0.619
Soil order: humo-ferric podzol	0.009	1.533	0.004	1.78
Surficial deposit: beach and aeolian deposit	0.352	0.696	0.96	1.025
Surficial deposit: organic deposit	0.955	0.986	0.511	0.82
Surficial deposit: outwash deposit	0.364	1.228	0.828	1.053
Surficial deposit: ground moraine	0.841	0.966	0.999	0.999
Mean firebreak distance	0.016	1.299	0.011	1.382
Aspect (west–east axis)	0.811	0.973	0.794	0.971
Aspect (north–south axis)	0.525	0.931	0.377	0.903
Elevation	0.614	0.999	0.19	0.997

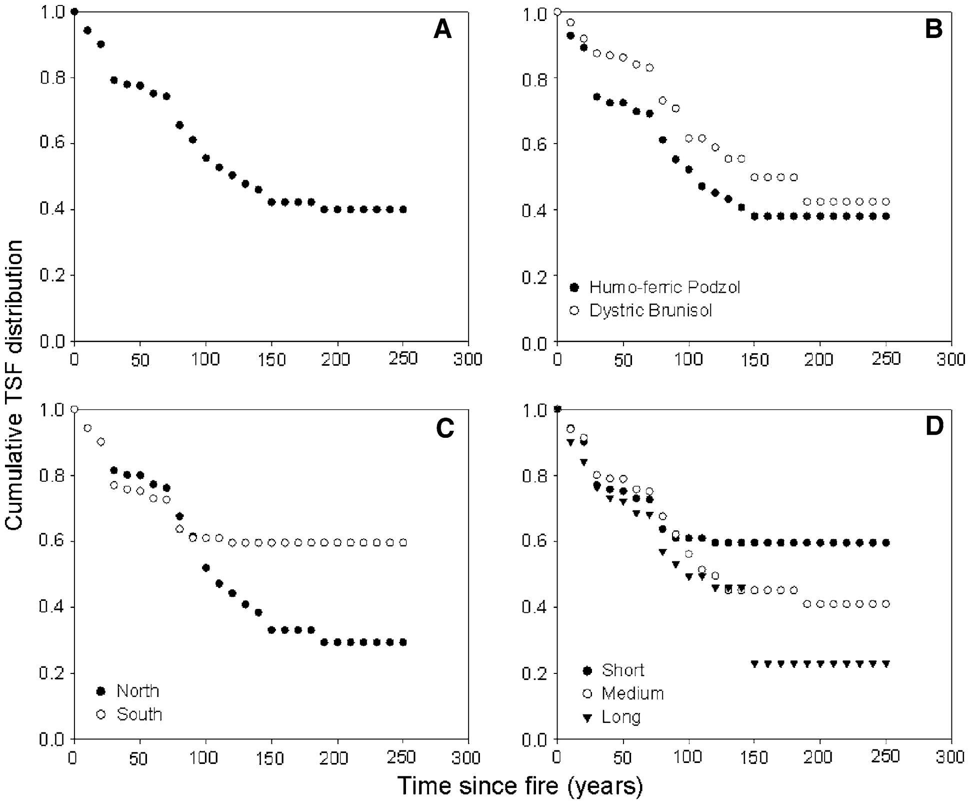

Figure 3. Cumulative TSF distribution using logarithmic scale comparing: **A** global distribution, **B** dominant soil order (podzol vs. brunisols), **C** latitude (north >49°30′N and south ≤ 49°30′N), and **D** mean distance to firebreak (MDF) (*short* ≤ 1350 m, *intermediate* 1351–3800 m, *long* ≥ 3801 m).

(Larsen 1996; Weir and others 2000). However, the decreasing fire cycles we observed support an opposite trend towards increasing fire frequency confirming that wildfire in the central boreal shield has been much more frequent than in the wetter eastern boreal region (Bergeron and others 2004a) and less frequent than in the drier western boreal region (Larsen 1997) of North America throughout the 20th century. Our findings together with those from Larsen (1997) and Bergeron and others (2004a) provide support for a climatic top-down control on fire activities of the entire North American boreal forest (Krawchuk and others 2009; Parisien and Moritz 2009).

Temporal Trends in Fire Frequency

There is a discrepancy in fire cycle calculations between the burn rate and survival analysis for the most recent period 1921–2008. Similar discrepancies between the two methods were also found in other studies (for example, Lauzon and others 2007). There are two possible explanations for such discrepancies. Because we used censored minimum TSF (that is, 89 years), a commonly used approach in fire frequency studies (for example, Weir and others 2000; Heyerdahl and others 2001), for inaccessible hexagons and the remaining portion of the hexagons that were partially burnt during 1921–2008 in survival analysis, the distribution of censored intervals might have biased the estimation of the fire cycle towards a lower value for that period. Alternatively, fire occurrences in the early 1920s may not have been accurately documented as the region was considered very remote and largely inaccessible, leading to an overestimation of the fire cycle using the burn rate method. Therefore, we believe the fire cycle for the period of 1921–2008 is between 96 years estimated from survival analysis and 158 years from the burn rate method, and is likely closer to 100 years than 150 years because fires were not often completely documented in the early part of the last century (Paul Poschmann's, personal communication).

Our results show that a significant change in fire activity occurred around 1920 when the fire cycle decreased from 295 years (1820–1920) to 96 years (1921–2008) (Table 2). It should be noted that the decades of the 1980s and 1990s were marked by extremely high fire activity in the study area. Similarly, major fire activities have been reported in these decades elsewhere across the boreal forest (Skinner and others 1999, 2002; Gillett and others 2004; Le Goff and others 2007). An analysis for the periods of 1921–1970 and 1971–2008 showed a fire cycle of 140 years for the 1921–1970 period and 69 years for 1971–2008, indicating that the high activity in these two decades accounts for a major component of the shorter fire cycles for the period of 1921–2008 we observed.

The temporal pattern we observed is consistent with the findings in northwestern Minnesota, an area possessing a continental climate similar to that of our study area, where fire frequency increased by 25% in the 20th century as a result of warm, dry conditions following the Little Ice Age and the absence of fire suppression in the early part of the century (Clark 1988). However, these results are in contrast to a trend of fire cycle lengthening reported in the eastern (Lesieur and others 2002; Lefort and others 2003) and western (Weir and Johnson 1998; Weir and others 2000) boreal forest. The different temporal patterns in fire regimes suggest distinctive regional responses to climate change, attributable to the occurrence rates of extreme drought years (Parisien and Moritz 2009; Girardin and others 2009). It is difficult to discern the relative contributions of climate change and human activities to the increased fire frequency in the study region as the start of harvesting operations (~1920) coincide with extreme fire weather occurring across the boreal region (Girardin and others 2005, 2006b). The overall increase in fire frequency has occurred despite an increase in landscape fragmentation, area under fire management, and more efficient fire suppression techniques, suggesting that the change in fire activity appears driven primarily by climate coupled with human activities contributing to spatially mixed fire frequencies.

Spatial Variation

Our results show a latitudinal gradient of fire frequency, lower fire frequency in the southern portion and higher in the north of the region. Apart from climatic differences associated with latitude and potential regional responses to climate change, historical human activities may account for this spatial variation. The pattern observed is similar to that found in the central boreal forest of Saskatchewan where landscape fragmentation caused by timber removal significantly decreased fire frequency (Weir and Johnson 1998; Weir and others 2000). Other studies also suggest that diminishing fire activities are related to landscape fragmentation from timber harvesting (Lefort and others 2003; Marlon and others 2008). In addition, effectiveness of fire suppression resulted in reduced fire activities (Mouillot and Field 2005; Marlon and others 2008). The road network, a consequence of the historical pattern of harvesting within our study landscape, is far denser in the southern area, and has been shown not only to permit more effective detection and faster access for firefighting purposes (Lefort and others 2004; Bergeron and others 2006), but also augments the number of firebreaks in the landscape inhibiting fire ignition and spread (Foster 1983; Turner and Romme 1994).

Soil order and firebreak distance were also significant determinants of fire frequency. Soil orders were strongly correlated with the drainage and texture of the growing substrate, thereby influencing species composition by availability of soil moisture and nutrients. The development of humo-ferric podzol is positively associated with coniferous forest litter (Jauhiainen 1973; Flanagan and Van-cleve 1983; Lundstrom and others 2000). Considering the high correlation between humo-ferric podzol and moderate drainage, the conifer species most likely to have dominance on these soils is black spruce, whose abundance has been reported to be proportional to fires burning large areas (Hely and others 2000, 2001; Bergeron and others 2004b). Our findings and those reported by others illustrate the mutually reinforcing and interdependent relationships between soils, vegetative cover, and fire (Schulte and others 2005; Drever and others 2006).

As the mean distance from a firebreak increased, so did fire frequency. These results have been found in previous boreal forest fire studies (Heinselman 1973; Larsen 1997; Cyr and others 2005). Reduced fire frequency associated with closeness to waterbodies can be a result of fragmentation by and increased air moisture from waterbodies. Additionally, covariation between soil type and waterbreak (firebreak) distance influences the dominant vegetative cover, which in turn influences fire frequency (Larsen 1997).

Aside from soil order and firebreak distance, no significant differences in fire frequency owing to physiographic features of elevation, aspect, or surficial deposit were detected. Relationships between these local factors and fire frequency may not have been detected due, in part, to the hexagon size (40 km^2) used in our sampling. Our large hexagon size allows an examination of temporal and spatial dynamics of a large landscape (11,600 km^2); this coarse scale of sampling, however, may be inadequate to reveal the effects of the local spatial factors on probability of ignition or to smaller fires (Parisien and Moritz 2009).

CONCLUSIONS

For the landscape we studied, fire activities have increased with a fire cycle of 295 years for the period prior to 1921 and between 96 and 156 years for the period of 1921–2008. The increased fire activities appear to be a result of increased drought associated with climate change in the central boreal forest region. The different temporal patterns of fire frequency between the central boreal region and the eastern and western boreal regions suggest that the responses of the regional fire regimes to global climate change can vary strongly among regions.

The significantly lower fire frequency in the southern part of the region indicates that fire frequency has been reduced by human activities such as increased landscape fragmentation and fire suppression. Dominant soil order was found to create spatially heterogeneous fire frequencies, indicating that fuel type associated with vegetation cover and soils plays a major role in affecting fire frequency at the regional scale. Additionally, waterbodies appear to inhibit fire ignition and spread as sites with close water had a lower fire frequency. Overall, wildfire in the central boreal shield is more frequent than that in the wetter eastern boreal region and less frequent than that in the drier western boreal region, confirming a climatic top-down control on the fire activities of the entire North American boreal forest.

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REFERENCES

Baldwin DJB, Desloges JR, Band LE. 2001. Physical geography of Ontario. In: Perera A, Euler D, Thompson I, Eds. Ecology of a managed terrestrial landscape: patterns and processes of forest landscapes in Ontario. Vancouver: UBC Press.

Bergeron Y. 1991. The influence of island and mainland lakeshore landscapes on boreal forest-fire regimes. Ecology 72(6):1980–92.

Bergeron Y, Dansereau PR. 1993. Predicting the composition of Canadian southern boreal forest in different fire cycles. J Veg Sci 4(6):827–32.

Bergeron Y, Leduc A, Harvey BD, Gauthier S. 2002. Natural fire regime: a guide for sustainable management of the Canadian boreal forest. Silva Fenn 36(1):81–95.

Bergeron Y, Flannigan M, Gauthier S, Leduc A, Lefort P. 2004a. Past, current and future fire frequency in the Canadian boreal forest: implications for sustainable forest management. Ambio 33:356–60.

Bergeron Y, Gauthier S, Flannigan M, Kafka V. 2004b. Fire regimes at the transition between mixedwood and coniferous boreal forest in Northwestern Quebec. Ecology 85(7):1916–32.

Bergeron Y, Cyr D, Drever CR, Flannigan M, Gauthier S, Kneeshaw D, Lauzon E, Leduc A, Le Goff O, Lesieur D, 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 36(11):2737–44.

Beverly JL, Martell DL. 2005. Characterizing extreme fire and weather events in the Boreal Shield ecozone of Ontario. Agric For Meteorol 133(1–4):5–16.

Bond-Lamberty B, Peckham SD, Ahl DE, Gower ST. 2007. Fire as the dominant driver of central Canadian boreal forest carbon balance. Nature 450(7166):89.

Brassard BW, Chen HYH. 2008. Effects of forest type and disturbance on diversity of coarse woody debris in boreal forest. Ecosystems 11(7):1078–90.

Brassard BW, Chen HYH, Wang JR, Duinker PN. 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 38(1):52–62.

Chapin FS, McGuire AD, Randerson J, Pielke R, Baldocchi D, Hobbie SE, Roulet N, Eugster W, Kasischke E, Rastetter EB, Zimov SA, Running SW. 2000. Arctic and boreal ecosystems of western North America as components of the climate system. Glob Chang Biol 6:211–23.

Chen HYH, Popadiouk RV. 2002. Dynamics of North American boreal mixedwoods. Environ Rev 10:137–66.

Chen HYH, Vasiliauskas S, Kayahara GJ, Ilisson T. 2009. Wildfire promotes broadleaves and species mixture in boreal forest. For Ecol Manage 257:343–50.

Clark JS. 1988. Effect of climate change on fire regimes in northwestern Minnesota. Nature 334(6179):233–5.

Clark JS. 1990. Fire and climate change during the last 750 yr in northwestern Minnesota. Ecol Monogr 60(2):135–59.

Clark JS, Royall PD. 1996. Local and regional sediment charcoal evidence for fire regimes in presettlement north-eastern North America. J Ecol 84(3):365–82.

Connell JH. 1978. Diversity in tropical rain forests and coral reefs—high diversity of trees and corals is maintained only in a non-equilibrium state. Science 199(4335):1302–10.

Cox DR. 1972. Regression models and life tables (with discussion). J R Stat Soc Series B 34:187–220.

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

Cyr D, Gauthier S, Bergeron Y. 2007. Scale-dependent determinants of heterogeneity in fire frequency in a coniferous boreal forest of eastern Canada. Landsc Ecol 22(9):1325–39.

Drever CR, Messier C, Bergeron Y, Doyon F. 2006. Fire and canopy species composition in the Great Lakes-St. Lawrence forest of Temiscamingue, Quebec. For Ecol Manage 231(1–3): 27–37.

Ecoregions Working Group. 1989. Ecoclimatic regions of Canada, first approximation. Ecological Land Classification Series, No. 23. Ottawa, ON: Sustainable Development Branch, Canadian Wildlife Service, Conservation and Protection, Environment Canada.

Flanagan PW, Vancleve K. 1983. Nutrient cycling in relation to decomposition and organic-matter quality in Taiga ecosystems. Can J For Res 13(5):795–817.

Foster DR. 1983. The history and pattern of fire in the boreal forest of southeastern Labrador. Can J Bot 61(9):2459–71.

Gillett NP, Weaver AJ, Zwiers FW, Flannigan MD. 2004. Detecting the effect of climate change on Canadian forest fires. Geophys Res Lett 31(18):L18211. doi:10.1029/2004GL020876.

Girardin MP, Tardif JC, Flannigan MD, Bergeron Y. 2005. Synoptic-scale atmospheric circulation and boreal Canada summer drought variability of the past three centuries. J Clim 19:1922–47.

Girardin MP, Tardiff JC, Flannigan MD, Bergeron Y. 2006a. Forest fire-conducive drought variability in the southern Canadian boreal forest and associated climatology inferred from tree rings. Can Water Resour J 31(4):275–96.

Girardin MP, Bergeron Y, Tardif JC, Gauthier S, Flannigan MD, Mudelsee M. 2006b. A 229-year dendroclimatic-inferred record of forest fire activity for the Boreal Shield of Canada. Int J Wildland Fire 15(3):375–88.

Girardin MP, Ali AA, Carcaillet C, Mudelsee M, Drobyshev I, Hely C, Bergeron Y. 2009. Heterogeneous response of circumboreal wildfire risk to climate change since the early 1900s. Glob Chang Biol 15(11):2751–69.

Hart SA, Chen HYH. 2006. Understory vegetation dynamics of North American boreal forests. Crit Rev Plant Sci 25(4): 381–97.

Hart SA, Chen HYH. 2008. Fire, logging, and overstory affect understory abundance, diversity, and composition in boreal forest. Ecol Monogr 78(1):123–40.

Heinselman ML. 1973. Fire in the virgin forest of the Boundary Waters Canoe Area, Minnesota. Quat Res 3:329–82.

Hely C, Bergeron Y, Flannigan MD. 2000. Effects of stand composition on fire hazard in mixed-wood Canadian boreal forest. J Veg Sci 11(6):813–24.

Hely C, Flannigan M, Bergeron Y, Mcrae D. 2001. Role of vegetation and weather on fire behavior in the Canadian mixedwood boreal forest using two fire behavior prediction systems. Can J For Res 31(3):430–41.

Heyerdahl EK, Brubaker LB, Agee JK. 2001. Spatial controls of historical fire regimes: a multiscale example from the interior west, USA. Ecology 82(3):660–78.

Ilisson T, Chen HYH. 2009. Response of six boreal tree species to stand replacing fire and clearcutting. Ecosystems 12(5):820–9.

Jauhiainen E. 1973. Effect if climate on podzolisation in southwest and eastern Finland. Commentat Phys Math 43:214–42.

Johnson EA. 1992. Fire and vegetation dynamics—studies from the North American boreal forest. Cambridge, UK: Cambridge University Press.

Johnson EA, Gutsell SL. 1994. Fire frequency models, methods and interpretations. Adv Ecol Res 25:239–87.

Johnson EA, Miyanishi K, Weir JMH. 1995. Old-growth, disturbance, and ecosystem management. Can J Bot 73(6): 918–26.

Johnson EA, Miyanishi K, Weir JMH. 1998. Wildfires in the western Canadian boreal forest: landscape patterns and ecosystem management. J Veg Sci 9(4):603–10.

Johnstone JF, Hollingsworth TN, Chapin FS, Mack MC. 2010. Changes in fire regime break the legacy lock on successional trajectories in Alaskan boreal forest. Glob Chang Biol 16(4):1281–95.

Klein JP, Molin JD. 2005. Survival analysis: techniques for censored and truncated data. 3rd edn. New York: Springer.

Krawchuk MA, Moritz MA, Parisien MA, Van Dorn J, Hayhoe K. 2009. Global pyrogeography: the current and future distribution of wildfire. Plos One 4(4):e5102. doi:10.1371/journal.pone.0005102.

Larsen CPS. 1996. Fire, climate dynamics in the boreal forest of northern Alberta, Canada, from AD 1850 to 1989. Holocene 6(4):449–56.

Larsen CPS. 1997. Spatial and temporal variations in boreal forest fire frequency in northern Alberta. J Biogeogr 24(5):663–73.

Lauzon E, Kneeshaw D, Bergeron Y. 2007. Reconstruction of fire history (1680–2003) in Gaspesian mixedwood boreal forests of eastern Canada. For Ecol Manage 244(1–3):41–9.

Le Goff H, Flannigan MD, Bergeron Y, Girardin MP. 2007. Historical fire regime shifts related to climate teleconnections in the Waswanipi area, central Quebec, Canada. Int J Wildland Fire 16(5):607–18.

Lefort P, Gauthier S, Bergeron Y. 2003. The influence of fire weather and land use on the fire activity of the Lake Abitibi area, eastern Canada. For Sci 49:509–21.

Lefort P, Leduc A, Gauthier S, Bergeron Y. 2004. Recent fire regime (1945–1998) in the boreal forest of western Quebec. Ecoscience 11(4):433–45.

Legendre P, Legendre L. 1998. Numerical ecology. 2nd edn. Amsterdam: Elsevier Science.

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

Lundstrom US, van Breemen N, Bain D. 2000. The podzolization process. A review. Geoderma 94(2–4):91–107.

Marlon JR, Bartlein PJ, Carcaillet C, Gavin DG, Harrison SP, Higuera PE, Joos F, Power MJ, Prentice IC. 2008. Climate and human influences on global biomass burning over the past two millennia. Nat Geosci 1(10):697–702.

Mouillot F, Field CB. 2005. Fire history and the global carbon budget: a 1 degrees x 1 degrees fire history reconstruction for the 20th century. Glob Chang Biol 11(3):398–420.

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

Parisien MA, Moritz MA. 2009. Environmental controls on the distribution of wildfire at multiple spatial scales. Ecol Monogr 79(1):127–54.

Payette S, Morneau C, Sirois L, Desponts M. 1989. Recent fire history of the northern Quebec biomes. Ecology 70(3):656–73.

Phipps RL. 1986. Collecting, preparing, crossdating and measuring tree increment cores, 85-4148. Lakewood, Colorado: U.S. Geological Survey Water-Resources Investigations. p 5.

Qian H, Klinka K, Kayahara GJ. 1998. Longitudinal patterns of plant diversity in the North American boreal forest. Plant Ecol 138(2):161–78.

R Development Core Team. 2009. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Version 2.9.2. Vienna, Austria. http://www.r-project.org.

Schulte LA, Mladenoff DJ, Burrows SN, Sickley TA, Nordheim EV. 2005. Spatial controls of pre-Euro-American wind and fire disturbance in Northern Wisconsin (USA) forest landscapes. Ecosystems 8(1):73–94.

Simard M, Lecomte N, Bergeron Y, Bernier PY, Pare D. 2007. Forest productivity decline caused by successional paludification of boreal soils. Ecol Appl 17(6):1619–37.

Skinner WR, Stocks BJ, Martell DL, Bonsal B, Shabbar A. 1999. The association between circulation anomalies in the mid-troposphere and the area burned by wildland fire in Canada. Theor Appl Climatol 63:89–105.

Skinner WR, Flannigan MD, Stocks BJ, Martell DL, Wotton BW, Todd JB, Mason JA, Logan KA, Bosch EM. 2002. A 500-hPa synoptic wildland fire climatology for large Canadian forest fires 1959–1996. Theor Appl Climatol 71:157–69.

Soil Classification Working Group. 1998. The Canadian system of soil classification, 3rd edn. Agriculture and Agri-Food Canada Publication 1646 (Revised). 187 pp. http://sis.agr.gc.ca/cansis/references/1998sc_a.html.

Soil Landscapes of Canada Working Group. 2007. Soil Landscapes of Canada v 3.1.1. Agriculture and Agri-Food Canada. http://sis.agr.gc.ca/cansis/nsdb/slc/v3.1.1/intro.html.

Stocks BJ. 1993. Global warming and forest-fires in Canada. For Chron 69(3):290–3.

Therneau T. 2009. R package ‘survival, version 2.35-8. http://cran.r-project.org/web/packages/survival/survival.pdf.

Tsiatis A. 1981. A large sample study of the estimate for the integrated hazard function in Cox’s regression model for survival data. Ann Stat 9:93–108.

Turner MG, Romme WH. 1994. Landscape dynamics in crown fire ecosystems. Landsc Ecol 9(1):59–77.

Vasiliauskas S, Chen HYH. 2002. How long do trees take to reach breast height after fire in northeastern Ontario? Can J For Res 32(10):1889–92.

Weir JMH, Johnson EA. 1998. Effects of escaped settlement fires and logging on forest composition in the mixedwood boreal forest. Can J For Res 28:459–67.

Weir JMH, Johnson EA, Miyanishi K. 2000. Fire frequency and the spatial age mosaic of the mixed-wood boreal forest in western Canada. Ecol Appl 10(4):1162–77.