Spatial and temporal variability of stand-replacing fire frequency in Quetico Provincial Park, Ontario

by

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AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Fire is the primary natural disturbance vital to the ecological integrity of Quetico Provincial Park, Ontario, Canada. A new provincial park planning process (i.e., Class Environmental Assessment) has required the review of Quetico's Fire Management Plan. To support this review, large and severe (stand-replacing) Quetico fires were studied using 1966 Ontario Ministry of Natural Resources (OMNR) forest resource inventory (FRI) mapping. A Geographic Information Systems (GIS) database of the FRI was created and updated with the OMNR digital fire atlas. This database was used as a time-since-fire and fire interval dataset to estimate fire frequency. It also served to archive the 1966 FRI for the largest protected area in the transition between the Boreal and Great Lakes-St. Lawrence forest regions. Non-parametric (Kaplan-Meier) survival analysis was used to estimate survival functions and mean fire intervals (i.e., the expected time between two consecutive stand-replacing fires for any location within the Park). Previous studies that have used Kaplan-Meier survival analysis methods have based fire frequency estimates solely on time-since-fire data. However, time-since-fire data cannot be equated with fire interval data when using non-parametric methods. At least one fire interval is required to obtain reliable results. The mean fire interval for the entire 475,782 ha Park between the years 1668 and 2007 was 230 years. Performing the analysis on various geographic and temporal partitions revealed fire frequency spatial and temporal variability. A constant (independent of time-since-fire) probability of burning was not observed for Quetico which is contrary to accepted conjecture for northwestern Ontario boreal/mixed-wood forests. A current fire cycle was also estimated for the Park (342 years) using the digital fire atlas. The results suggested that use of historical static fire frequency estimates as fire management prescriptions may not be justified given considerable fire frequency temporal variability. The observed fire frequency spatial variability suggests that studies should be undertaken at coarser scales than is the norm to characterise the regions fire regime in support of landscape level fire management planning.

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Dedication

This research is dedicated to Quetico Provincial Park and anyone who has loved or will come to love this “diamond in the rough”.

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List of Acronyms

BFOLDS – Boreal Forest Landscape Dynamics Simulator

BWCAW – Boundary Waters Canoe Area Wilderness

FMP – Fire Management Plan

FRI – forest resource inventory

FU – Forest Unit

GCP – ground control points

GIS – geographic information system

GLSL – Great Lakes-St. Lawrence (forest region)

GPS – global positioning system

MLE – Maximum Likelihood Estimator

MU – Management Unit

NRVIS – Natural Resource Values Information System

OBM – Ontario Base Mapping

OMNR – Ontario Ministry of Natural Resources

RMS error – root mean square error

WG – Working Group

Key Definitions

Burning rate – A computation used to provide an area-based estimate of fire frequency (fire cycle) indicating how fast the total area could burn once (Heinselman, 1973).

Digital fire atlas – Ontario forest fire history maps originally compiled by Donnelly and Harrington (1978). Includes all fires south of 52^{o} North, known as "The Area of Undertaking" (all land and water within forest management unit boundary lines) (OMNR, 2008b) and greater than or equal to 200 ha. Updated and converted to a GIS database by Perera et al. (1998) and continued today by the Ontario Ministry of Natural Resources.

Ecological integrity - "A condition in which biotic and abiotic components of ecosystems and the composition and abundance of native species and biological communities are characteristic of their natural regions and rates of change and ecosystem processes are unimpeded" (Canada National Parks Act, 2000; Ontario Legislative Assembly, 2006; Woodley, 1995).

Fire cycle - The number of years expected to burn an area equal in size to the landscape of interest (Reed, 2006). The reciprocal of the fire cycle is the percentage of annual area burned (Li, 2000).

Fire frequency - The number of fires that occur within a given time period; most commonly expressed in terms of mean fire interval or fire cycle (Van Sleeuwen, 2006).

Forest stand – "a community of trees possessing sufficient uniformity in composition, constitution, age, arrangement or ecological condition to be distinguishable from adjacent communities, so forming a silvicultural or management entity" (OMNR, 2007).

Global survival function – The survival curve of the entire landscape of interest and time period under investigation.

Maximum Likelihood Estimator (MLE) – A statistical method used to estimate survival curves, fire frequency (fire cycle) and the hazard of burning. First used by Reed (1994) in fire frequency studies.

Mean fire interval - The expected number of years between two successive stand-replacing fires at any location in the landscape of interest (Reed, 2006)

Negative exponential fire model – A parametric model used in fire frequency studies whereby the hazard of burning is assumed to be independent of stand age (usually reckoned as time-since-fire) and therefore constant (Van Wagner, 1978).

Non-parametric fire model – Survival analysis methods where the survival function is derived from empirical fire frequency data (e.g., Kaplan Meier, 1957).

Parametric fire model - Survival analysis methods where fire frequency data are fitted to a theoretical model (negative exponential or Weibull are most commonly used).

Hazard (probability) of burning – The hazard of burning is a statistical concept which refers to a hazard rate (Johnson and Gutsell, 1994). It is proportional to the instantaneous probability of fire at any given point within a sampling unit (i.e., forest stand) (D. Matthews, personal communication, 2008). Johnson and Gutsell (1994) defined hazard of burning as "the *per capita* age-specific mortality from fire. It is the probability of fire occurring in an interval, assuming survival up to the beginning of the interval" (p. 244). Their definition assumed that fire frequency data are fitted to either the Weibull or negative exponential theoretical fire models. To avoid confusion, the hazard of burning will be referred to as the probability of burning for this research. This choice of terminology is particularly appropriate as the emphasis in this study was on the use of non-parametric survival analysis methods which typically estimate probabilities.

Right-censoring – A statistical technique whereby stand age/time-since-fire data (incomplete observations) are treated as minimum estimates (a lower bound) on fire interval data (complete observations).

Root Mean Square Error – A computation used when georectifying GIS data that measures the geometric distortion between the control layer (the layer with known coordinates) and the Ground Control Point values (the coordinates used to rectify the GIS data) (Jensen, 2005).

Stand-replacing fire – A severe fire that results in the death of the majority (at least 90%) of the dominant canopy trees within a forest stand (Brown, 1995; Van Sleeuwen, 2006).

Survival analysis – A statistical technique used to predict (estimate) the time to an event. In this research forest stand death due to fire is the event of interest.

Survival function – The probability that a randomly chosen forest stand is not replaced by fire as a function of stand age (time-since-fire).

Timber berth – forest licensed by the provincial government (Crown) to be harvested for commercial purposes.

Time-since-fire class – A grouping of time-since-fire dates. Equivalent to grouping forest stand ages into classes of 10 or 20 years.

Time-since-fire distribution – The cumulative percent of the forest landscape of interest surviving in each time-since-fire class.

Weibull fire model - A parametric model used in fire frequency studies whereby the hazard of burning is assumed to be dependent on stand age (time-since-fire) (Johnson, 1979).

Chapter 1
Introduction

1.1 Excerpt from Matt Scoular's Diary: Quetico Provincial Park, August 1995

Day 3 – Lonely Lake to Russell Lake

"The paddling was tough today. The winds were in our faces no matter which way we turned! As I came to the end of the portage, all I could think was man, I'm glad we're laying low tomorrow! Russell Lake was a great place to spend an entire day: great fishing and hiking trails along Chatterton Falls. However, we would have to diligently keep the campsite clean as Ministry warnings of nuisance bear activity were posted on entry points and portages. The bears were likely scavenging more because the berry crop hasn't been good due to the extremely hot and dry summer. Watched an incredible sunset tonight; the smoke and particulate matter from a fire in the southeastern part of the Park was the cause of this beautiful spectacle. We thought it probable that careless campers were responsible for destroying the beautiful wilderness we cherished."

Day 4 – Russell Lake

"Woke up to sniffing and grunting outside the tent! A young black bear, likely 2 or 3 years old, had stolen our peanut butter and summer sausage! Our campsite was immaculate!?! Cheeky bear! Although a restful day, we were not able to explore or fish as we had planned because the strong winds made it too dangerous to paddle."

Day 7 – Jesse to Nym Lake

"...less than 50 m visibility on Batchewaung Lake today...put wet bandanas over our faces as a precautionary measure to protect us from the smoke..."

1.2 Need for the Research

In August 1995, Fire #141, ignited from a single lightning strike and burned approximately 25,000 ha (roughly 5% of the Park) of the southeast portion of Quetico Provincial Park. It was the largest fire in the Parks history since the six fires of July 1936, which together burned 60,000 ha. Fire #141 was not initially deemed a priority, as the extremely dry and hot summer was keeping fire crews busy with other fires throughout the region's commercial timber lands. However, priority status was given to the fire when strong winds shifted to the southeast, and the fire raced towards adjacent timber lands.

Just as Fire #141 varied in size from day-to-day, the frequency of fire also varies spatially and over longer timeframes (years, decades and centuries). Large stand-replacing fires within Quetico (i.e., during the late 1800s, early 1900s and 1995) and their spatial and temporal variability are responsible for the renewal of Quetico's forests providing the tree species composition, age structure and configuration of the forest mosaic upon which it's wildlife depend (Heinselman, 1996).

New legislation (Ontario Legislative Assembly, 2006), fire policy (OMNR, 2004c) and a park planning process (OMNR, 2004d) regard ecological integrity as the management priority in Ontario's provincial parks. Quetico has had a Fire Management Plan (FMP) since 1997 (OMNR, 1997), and it's managers have already made progressive steps to integrate the concept of ecological integrity into Quetico's fire-dependent ecosystem. A review of Quetico's FMP was completed in the summer of 2008 to comply with the OMNR planning process which mandates a review of all FMPs every 10 years (OMNR, 1997).

The characterisation of fire frequency and its spatial and temporal variability is essential for understanding fire regimes and thus implementing effective fire management. Fire frequency studies have typically been undertaken using statistical and simulation approaches. However, a combination of approaches utilizing multiple datasets yields the most accurate results. Fire frequency studies completed for Quetico (Woods and Day, 1977a) and the Boundary Water Canoe Area Wilderness (BWCAW) (Heinselman, 1973) have laid the groundwork for the Park's FMP. However, Woods and Day's study covered less than a quarter of the Park in any detail, and Heinselman's study was south of the Park. Furthermore, much has changed since the 1970s, including:

- A decrease in the Park's fire suppression activities (OMNR, 1997; Solomon, 2007);
- Re-introduction of fire through prescribed fires and burns (Solomon, 2007);
- More apparent climate warming and its effects on fire regimes (Flannigan and Van Wagner, 1991; Flannigan et al., 1998; Thompson et al. 1998); and
- Advances in fire frequency study methods (survival analysis) (Reed 1994; Reed, 1998; Reed 2006).

Consequently, Quetico's fire management policies are based upon an incomplete scientific ecological understanding of the region's fire regime. This highlights the need for an updated and expanded fire frequency study. Updating the Park's fire data was also deemed a priority by Park managers (L. Solomon, personal communication, 2006) due to imminent Park Master Plan (1977, 1996) revisions, and related efforts to review and revise the Park's FMP.

This research used a historical approach by creating a Geographic Information System (GIS) database of the 1966 Ontario Ministry of Natural Resources (OMNR) Forest Resource Inventory (FRI) of Quetico. The FRI provided an initial time-since-fire map and was updated to 2007 with current digital fire atlas data. Fire #141 of 1995 provided the majority of the fire interval data needed to estimate fire frequency (mean fire interval) variability in space and through time using survival analysis.

1.3 Research Goal, Objectives and Hypotheses

The goal of this research was to characterise the fire regime of Quetico Provincial Park. More specifically, stand-replacing fire frequencies were estimated for different areas of the Park and different time periods to support the review of the Park's fire management policy. To achieve this goal, survival analysis was undertaken for the statistical estimation of fire frequency. The results were compared with previous fire frequency studies in the Park and the adjacent BWCAW.

Given the broad goal of the research, the eight research objectives were to:

1. **Conduct a literature review of fire management in Ontario, it's provincial parks and specifically, Quetico Provincial Park**; to understand the evolution of fire management in Ontario, how it led to provincial fire strategies and polices as well as the current context of fire management within Quetico.
2. **Conduct a literature review of fire frequency study concepts, data collection and statistical approaches and summarize the results of fire frequency studies relevant to the Park and the region**; so as to understand the fire frequency concepts used in this research, how fire frequency data has been collected and the different statistical approaches employed in fire frequency studies. The literature review was also undertaken to assemble a current understanding of fire frequency and its spatial and temporal variability within Quetico and the region.
3. **Georectify and digitize the 1966 OMNR FRI**; to create a GIS database of the FRI to provide an initial time-since-fire map for the entire Park for 1668 (i.e., oldest stand in the 1966 FRI) to 1966 (i.e., the year of FRI mapping). This map was updated with the digital fire atlas for large fires (> 200 ha) post-1966 to create a current time-since-fire and fire interval dataset.

4. **Perform a historical forest analysis using the 1966 FRI**; to plot and map the forest age distribution; to plot the relative area occupied by each OMNR Working Group (WG) as a function of time-since-fire; and to plot the time-since-fire distribution to assess the influence of stand-replacing fires on the forest composition and age structure of the Park.
5. **Estimate survival functions, mean fire intervals and the probability of burning for the Park using contemporary methods**; to statistically estimate the survival functions, mean fire intervals and probability of burning using non-parametric Kaplan-Meier survival analysis.
6. **Characterise fire frequency spatial and temporal variability within Quetico;** to determine the variability of Quetico's fire frequency by partitioning the time-since-fire data set both spatially and temporally.
7. **Make recommendations for future fire frequency research**; to further develop hypotheses for characterising Quetico's fire regime based on the acceptance or rejection of the tested hypotheses.
8. **Evaluate the significance of the research findings**; to discuss the implications for Quetico's fire management and fire management in general as well as implications concerning the scales of data gathering for resource management and planning.

Four hypotheses were tested:

1. In 1966, the relative area occupied by the fire-dependant species Black Spruce and Jack Pine was not a function of time-since-last stand-replacing fire
2. Stand-replacing fire frequency does not vary between different areas within the Park
3. Stand-replacing fire frequency does not vary between a recent time period (1970-2007) and the entire time period of study (1668-2007)
4. The probability of burning is independent of time-since-last stand-replacing fire

1.4 Structure of the Thesis

The thesis is organized in the following chapters:

Chapter 2 – A literature review of relevant fire management and fire frequency studies.

Chapter 3 – Describes the context of Quetico Provincial including: regional context, geology and soils, terrain and hydrology, climate, forest communities, natural disturbance, the influence of humans on fire frequency and logging history.

Chapter 4 – Describes the data used in this research, the creation of the 1966 FRI GIS database and the time-since-fire/fire interval dataset and the survival analysis.

Chapter 5 – Includes the results of the current fire cycle estimate, historical forest analysis and survival analysis. The survival function and mean fire interval are provided for the entire Park over the entire time period (1668-2007) and for spatial (e.g., four quadrant) and temporal (i.e., 1970-2007) partitions. Also presented is the probability of burning and its temporal variability.

Chapter 6 – Discusses the statistical methods, the current burning rate, the historical forest analysis and the survival analysis. The implications of the findings related to the management of fire in the Park and in general are discussed in this chapter.

Chapter 7 – Summarizes recommendations for future fire frequency research and fire management within Quetico and in general.

Chapter 2
Quetico Provincial Park

Quetico Provincial Park was chosen as the study area given the available funding. This funding enabled the creation of the time-since-fire and fire interval dataset required to estimate the Park's fire frequency. When undertaking fire frequency studies it is important to describe the study area's current and historical land use and its biophysical characteristics. To explain, variation in fire frequency has been correlated to surficial geology, terrain, hydrology (i.e., the spatial pattern of lakes and wetlands), climate, forest tree species composition, natural disturbances (such as wind throw and insect outbreak) and logging. Therefore, this chapter describes Quetico's:

- Regional Context
- Geology and Soils
- Terrain and Hydrology
- Climate
- Forest Communities
- Natural Disturbance
- Human Influence on Fire Frequency
- Logging History

2.1 Regional Context

Quetico Provincial Park is located south of Atikokan and approximately 160 km west of Thunder Bay, Ontario, Canada. Its approximate coordinates are 48°N 90°W. Although originally established as a Forest Reserve in 1909 and then a Provincial Park in 1913, at 476,000 ha, it is now the third largest Wilderness class provincial park in Ontario. It is contiguous with the BWCAW and Voyageurs National Park protected areas in Minnesota, U.S.A to the south. The Lac La Croix First Nation (Neguagon Lake No. 25D Reserve) abuts the southwest boundary of the Park, extending along the north shore of Lac La Croix. Crown land that lies to the east (Dog-River Mattawin and Lakehead Forest Management Units (MUs)) and the north (Crossroute and Sapawe Forest MUs) is actively managed for timber (Figure 2.1).

Figure 2.1 Regional context (Canoe Country, 2008).

2.2 Geology and Soils

Quetico is situated within the southwestern portion of the Canadian Precambrian Shield where some of the oldest granite and metamorphic rock on earth are found; originating some 2.7 to 2.5 billion years ago (Solomon, 2007). Quetico's landforms result from glacial events that scattered the landscape with eskers and moraines (Heinselman, 1996; Solomon, 2007). The provincially significant Steep Rock Moraine, an end moraine, exceeds 30 m in places and lies within Quetico Provincial Park and the BWCAW (Heinselman, 1996; Solomon, 2007). The dominant surficial deposits in the Park consist of a thin, discontinuous mantle of sandy till ground moraine (Solomon, 2007). Consequently, the most common soil substrate in Quetico is a glacial till (a mix of sand, silt and stones) which is low in nutrients, and contributes to low plant diversity throughout the majority of the Park (Solomon, 2007). Soil depth within the region ranges from a few cm on ridge tops, to approximately 3 m at the bases of slopes and lowland depressions (Heinselman, 1996).

2.3 Terrain and Hydrology

Quetico's terrain varies from gently rolling in the northern portion of the Park, to moderately rugged, with long, deep, northeast-southwest oriented lakes in the rest of the Park. This unique landscape is a result of successive glaciations moving from the northeast. The Park's elevation varies between 376 and 540 m asl. Lakes comprise 98,803 ha (or 21%) of the Park's total area and drain westward to Rainy Lake and eventually north to Hudson Bay. The largest lakes completely within the Park boundaries are, Pickerel in the northeast, at 5,754 ha, Kawnipi in the central east, at 4,488 ha and

Quetico in the northwest, at 4,265 ha. In the southwest, Basswood Lake at 4,840 ha, lies within Quetico and the BWCAW (Solomon, 2007).

2.4 Climate

Quetico's climate is influenced by the convergence of continental polar air masses and dry air masses from the prairies. This convergence results in warmer and drier conditions relative to the rest of northwestern Ontario (Kronberg et al., 1996). Quetico's fire season falls between April 1st and October 31st each year (Solomon, 2007). During the fire season, winds predominate from the south to northwest and range from 6.8 to 8.5 km/hr (Environment Canada, 2008). The mean total annual precipitation for the region is 74 cm (Solomon, 2007). The Park's average annual temperature is 2^0C, with an average January temperature of -17.1^0C, and an average July temperature of 17.9^0C (Ontario Parks, 2007). Annual mean temperatures have warmed in northwestern Ontario more than any other area of the province (Racey, 2004) and this trend is expected to continue (Colombo et al., 1998; Lemieuk, 2007; Racey, 2004). According to Racey (2004), while total rainfall has increased slightly, participation is concentrated in fewer, larger rain events. Additionally, snowfall amount and the duration snow is on the ground have declined. Racey (2004) asserted that this warming, and consequent changes in participation, has increased fire season length and resulted in higher intensity early spring fires. Racey (2004) warns that this warming may increase the annual area burned in the future. However, recent simulation modelling by Lemieux (2007) suggested that Quetico will see a considerable increase in temperature and precipitation amounts and events by the year 2080. There is an agreement of a trend towards hotter summer days, milder winter days, and reduced snow cover in northwestern Ontario. However, the trend for precipitation remains ambiguous.

2.5 Forest Communities

Quetico lies in a transition zone between the Boreal forests to the north, the Mixed (Great Lakes-St. Lawrence(GLSL)) forests to the south and southeast, and the Great Plains forests to the west and southwest (Solomon, 2007). However, Quetico falls within a single ecoregion as defined by three authorities:

1. OMNR's 4W Pigeon River Ecoregion,
2. Hills' (1959) boreal shield ecoregion; and
3. Rowe's (1972) "Quetico" forest section within the GLSL forest region.

A photogrammetric survey of the Park (Woods and Day, 1976) revealed that boreal forest tree species are dominant comprising approximately 90% of the Park's forest communities. Boreal tree species are "pioneer" species and are generally identified as being fire originated. Hence, the majority of Quetico's forest communities originated from large fires during the late 1880s and early 1900s (Heinselman, 1973; Kronberg et al., 1998; Rowe, 1972; Woods and Day, 1976, 1977a).

Through their photogrammetric survey, Woods and Day (1976) found Jack Pine (*Pinus banksiana* Lamb.) to be the most common tree, dominating over 30% of the Park's forests. Jack Pine was frequently associated with Trembling Aspen (*Populus tremuloides* Michx.) and Black Spruce (*Picea mariana* (Mill.) BSP). Black Spruce was the second most abundant tree, dominating over 26% of the forested area. Approximately 20% of forests were dominated by Trembling Aspen, although they occurred in 70% of the forests as the dominant, secondary, or tertiary species. White Birch (*Betula papyrifera* Marsh.) dominated approximately 10% of the forests, and was commonly associated with Trembling Aspen and Black Spruce. Red Pine (*Pinus resinosa* Ait.) was dominant in less than 5% of the forests, and was commonly associated with White Pine (*Pinus strobes* L.), dominated only 3% of the forests, but occurred as a secondary species in approximately 13% of the forests. Balsam Fir (*Abies balsamea* (L.) Mill.) was found in approximately 16% of the forests, but dominated in only 3% (Woods & Day, 1976).

Around nearly every lake in the Park, wetlands such as bogs, fens, rich swamps, and marshes are found, especially along shorelines. However, they make up a small proportion of the Park's total land area (Scoular and Suffling, 2008; Solomon, 2007; Woods and Day, 1976). Black Spruce dominates the treed peat land bogs, whereas White Cedar (*Thuja occidentalis* L.), Tamarack (*Larix laricina* (Du Roi) K. Koch) and Black ash (*Fraxinus nigra* Marsh.) dominate the nutrient rich swamps (Heinselman, 1996).

The 1966 OMNR FRI used in this research provides an inventory of forest stand tree species composition for the entire Park and completion of a current FRI is expected in December of 2009 (L. Solomon, personal communication, 2008). Together, these two datasets will be valuable for studying vegetation and fire dynamics over time.

2.6 Natural Disturbance

Fire, wind throw, insect outbreak and disease are the major types of natural disturbances within Quetico and the boreal forest generally. Fire is the primary natural disturbance in Quetico. The Park's fire regime is characterised by frequent, small, low-severity fires and relatively infrequent large stand-replacing fires, with the latter accounting for the majority of area burned. Since 1920, approximately 120,000 ha (32%) of the Park has burned with most of the area attributable to four large wildfires, three in 1936 at approximately 60, 000 ha, and one in 1995 at approximately 25,000 ha (Solomon, 2007; OMNR, 2008). Prior to 1920, the major fire years were 1803, 1804, 1891(M. Gluck, personal communication, 2008), 1895 (and/or 1894), 1910 and 1917 (OMNR, no date).

Wind throw events are typically responsible for smaller gap disturbances within the Park. For example, the "Pines" blowdown in 2003 on Pickerel Lake killed 650 ha of Red Pine. There has been one known large occurrence, the 1999 Independence Day blow down, on the southern boundary of the Park that spanned the international border at 161,000 ha (10,000 ha of which were in Quetico) (Solomon, 2007). On October 12, 2000 a prescribed burn was conducted to reduce an increased fire hazard from deadfall and create a fuel break between the Quetico and the BWCAW blowdown areas.

Insect outbreak, Spruce budworm (*Choristoneura fumiferana* Clem.) and Jack Pine budworm (*Choristoneura pinus* Freeman) and disease, White Pine blister rust (*Cronartium ribicola* J.C. Fisch.), are two other natural disturbances that influence the succession of Quetico's forest communities. However, these are typically more localized and less severe than fire, as they are specific to individual tree species (e.g., Spruce budworm starts in Balsam Fir but can spread to White Spruce). Flannigan et al. (2005) explained that areas devastated by Spruce budworm have an increased fire fuel load and therefore can burn more readily than unaffected areas. A Jack Pine budworm outbreak occurred in 2006 and continues today (Solomon, 2007). There has not been active management of either insect outbreaks or disease within the Park.

2.7 Human Influence on Fire Frequency

People first appeared in the region around 10,000 years ago following ice sheet retreat. They hunted moose, caribou (now extirpated) and bear as well as gathering other resources (Ontario Archaeology, 2008; Peruniak, 1990). North American oral history and fire frequency meta-analysis studies indicate the use of fire for hunting, gathering, and warfare by native peoples (Martinson & Omi, 2003; Omi, 2005). However, it is unclear whether they significantly influenced the historic fire frequency of the region (Fritz, Suffling & Younger, 1993; Suffling and Speller, 1998).

In the late 1700s to early 1900s, European traders and settlers in the region brought increased accidental fire occurrence, especially near transportation corridors (waterways and railways), mines (e.g., Jack Fish Lake gold mine), mining exploration camps and timber lands (Fritz, Suffling & Younger, 1993; Peruniak, 1990). The Northwest Company (in the late 1700s) and the Hudson Bay

Company (in the 1800s) had fur trade routes throughout the Park's waterways (Peruniak, 1990) as it was the main route from the West to the Great Lakes. In 1901, the East-West Canadian Northern Railway was completed just north of Quetico and sparks from the locomotives increased the occurrence of fires, especially in 'slash' left over from logging operations (Killan 1993; Lambert and Pross, 1967; Peruniak, 2000). For example, Peruniak (2000) explains that in 1919 the Park's fire crew fought 22 fires in 21 days (29 May to 18 June), which burned 15,000 acres (6,070 ha) in three timber berths (areas licensed to be cut) in or near Quetico.

The first fire suppression legislation was enacted in 1917 *(Forest Fires Prevention Act)* in Ontario. However, fire control was not effective until the late 1960s following advances in transportation and fire fighting technology (Ward and Tithecott, 1993; Kasischke and Stocks, 2000). Such advances led to a provincial policy of total suppression and influenced the occurrence of fire within Quetico. Today, Quetico's Wilderness class designation does not permit development (e.g., logging, mining, etc.) or mechanized travel. Therefore, the only significant source of human-caused ignition is from that of careless backcountry campers or Park staff conducting prescribed burns.

2.8 Logging History

In 1896, timber berths were surveyed throughout the district of Fort Frances, including the future Quetico Park. Twenty-eight forestry licenses, comprising roughly three-quarters of the Park's total area, were surveyed within the Park. Peruniak (1990) estimated that between 1909 and 1946, approximately 1.2 million m^3 of White and Red Pines (approximately 4,735 ha) were cut from the Quetico, Bearpelt, and Maligne River watersheds in the northwest section of the Park. Between 1961 and 1971, 510,000 m^3 of Jack Pine, Black Spruce, Balsam Fir and Trembling Aspen were logged in the northeast corner of the Park (Solomon, 2007). By this time, most of the large stands of White and Red Pine had already been cut and the emphasis was placed on Jack Pine and Black and White Spruce dimension lumber (OMNR, 1993). Within the resultant 4,735 ha of cutover in the northeast, approximately 3,117,500 tree seedlings were planted and the area is now dominated by Jack Pine stands (Solomon, 2007). Hunter Island, located in the southern half of the Park, remained for the most part untouched by logging, due to inaccessibility. However, between 1940 and 1943 some logging occurred in several areas within Hunter Island. This logging yielded more than 106,188 m^3 of Red Pine, White Pine, Jack Pine, and Spruce saw logs from this area (Solomon, 2007). According to Peruniak (1990), logging was banned within the Park in 1971. OMNR harvest ledgers (Bowling, 2008) indicate that forest stands outside of the licensed areas were also cut from 1968 to 1971 (Figure 2.2). Since the harvest ledger corresponded with the 1966 FRI forest stand spatial reference identifiers, the stands depicted in Figure 2.2 were likely logged sometime between 1968[1] and 1971. However, there are discrepancies between the harvest years provided in the timber berth dataset and the actual harvest years in the harvest ledger.

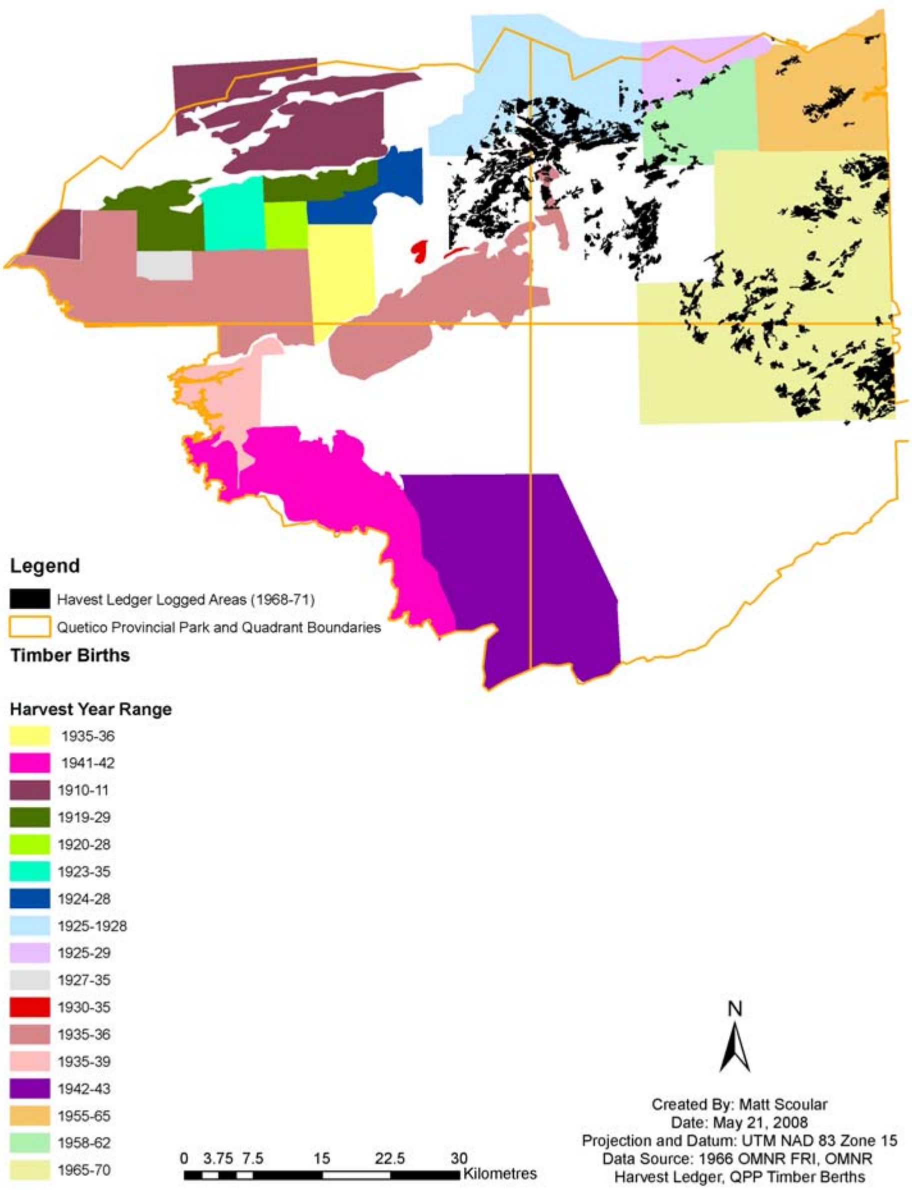

Figure 2.2 Quetico timber berths vs. harvested stands.

Note: the presence of a timber berth does not indicate that all of the area was logged. Areas outside the licensed areas (timber berths) were also logged.

[1] The FRI was completed in 1968 for use in forestry management planning.

Chapter 3
Literature Review

The topics covered in this chapter include Ontario, Provincial Park and Quetico fire policy and fire management. Fire frequency concepts, data collection and approaches to fire frequency estimation are also reviewed. A review of fire frequency studies completed for Quetico and the BWCAW that have led to the understanding of the region's fire regimes is also presented. A review of fire frequency study literature was necessary to understand the respective advantages and limitations of the various methods employed. The methods best suited to this research given the available funding and data, parallel OMNR research, timeline, budget and expertise of the researchers (i.e., GIS, statistical analysis and landscape ecology) was also confirmed by this review.

3.1 Managing Fire: To Burn or Not to Burn?

3.1.1 The Evolution of Fire Management Policy in Ontario

In 1878, the first fire prevention legislation in Ontario was enacted (OMNR, 2008b). However, it was not until 1885 when fire rangers, primarily nominated by logging companies, patrolled the province's forests attempting to protect them from fires (OMNR, 2008b). In the early 1900s, the expansion of the Canadian Northern Railway just north of Quetico, brought with it recreational retreats, mining and exploration camps and commercial logging (Lambert and Pross, 1967; Peruniak, 2000). During this time, forests licensed to be harvested were the main priority of fire suppression efforts. Many fires within these logged areas were caused by railway locomotive sparks (OMNR, 2008b; Peruniak, 2000). By the early 1900s, fire rangers were stationed on Crown land throughout the province, including Quetico, to protect the province's timber resources.

Large fires in the period between1916 and 1922 prompted changes to Ontario fire management policy. Fire management began to include human safety and infrastructure protection as priorities. In 1917, the Ontario government passed the *Forest Fires Prevention Act* in reaction to the destructive fires of June 16th, 1916. Fires that ignited from lightning and locomotive sparks combined into a firestorm that killed 244 people and devastated the Northeastern Ontario towns of Matheson and Cochrane (OMNR, 2008b; Lambert and Pross, 1967). After the 1922 Haileybury fire that destroyed 6,000 homes, the Ontario government established the Ontario Provincial Air Service (1924) to detect fires earlier (OMNR, 2008b). These large fires also prompted amendments to the Provincial Parks Act of 1927 for the purposes of fire protection. The new act gave the Minister of Lands and Forests the authority to revoke timber licenses or ban logging in parks (Killan, 1993). After the Second World War, advances in aviation (e.g., water bombers) and fire fighting (e.g., pumps and bulldozers) technology made remote areas more accessible and significantly improved fire suppression efforts (Pyne, 2007). Such advances eventually led to a provincial policy of fire suppression up until the 1970s.

The "management phase" of the 1970s (Woodley, 1995), was inspired by the increased knowledge of the ecological role fire played in perpetuating many ecosystems throughout the province. This phase was led by researchers studying in parks and protected areas and the staff that managed them. For example, Heinselman's (1973) landmark fire history study quantified area burned within the BWCAW using written records, fire scars and stand origins. He was able to prove that fire was responsible for the ecological integrity of the region as it shaped the composition, structure and spatial pattern of its vegetation patches (Heinselman, 1996). He stressed the importance of re-introducing fire to the region through the use of monitored lightning fires and prescribed burns. Heinselman's study triggered resource management agencies to evaluate the impact of fire suppression throughout North America.

Woods and Day (1977a) followed in Heinselman's footsteps with a fire ecology study in Quetico which used similar methods and covered approximately 21% of the Park. A year later, an official provincial parks policy and the *Ontario Provincial Parks Planning and Management Policies Manual* (revised 1992) were approved. Quetico's Fire Management Plan (FMP) was created to fulfill the fire management objectives set out in this policy manual which stated: "the occurrence of natural fire in certain wilderness environments is recognized as a process integral to evolving natural succession. A Fire Management Plan will be prepared for each Wilderness Park" (p. 49). Quetico's FMP was implemented in 1997.

Although provincial park fire policy came before national park fire policy, in 1979 Parks Canada was the first to implement the use of prescribed burns to reduce fuel load (Pyne, 2007). In 1986, amendments to the policy placed emphasis on "active management" through the use of prescribed burns to preserve landscapes and "duplicate nature as closely as possible" (Pyne, 2007). Although focused on assessing forest fire danger and behaviour for the forestry sector, research by the Canadian Forest Service (Van Wagner, 1978; Van Wagner and Methven, 1980) also equated fire management with vegetation management. It has possibly been the most influential Canadian fire ecology research and subsequently led to the development of the *Canadian Forest Fire Danger Rating System* and the *Canadian Forest Fire Behaviour Prediction System* (Van Wagner, 1974; Van Wagner, 1987; Stocks et al., 1987; Hirsch, 1996).

The next major influence on Canadian fire policy occurred after the 1988 fires in Yellowstone National Park. These fires led to a fire policy controversy on whether prescribed burns would help to prevent large fires. During the driest summer of Yellowstone Park's recorded history, fires burned approximately 485,623 ha, of which 320,916 ha were inside the park (National Park Service, 2007). The fire covered an area larger than Quetico Park. In 1988, U.S. federal policy stated that human-caused fires should be suppressed and natural fires (those started by lightning) should be allowed to run their course. Initially, the ecological destruction and economic loss caused by the fires brought managers and their "natural-fire policy" (or what the critics labeled as "let-it-burn"), under considerable socio-political pressure (Bonnicksen, 1989; Nodvin and Waldrop, 1990). However, research indicated that approximately 30-50% of the area within the perimeter of the fire was unburned creating a mosaic of burned and unburned vegetation and provided ideal conditions for vegetation regeneration and wildlife habitat (Christensen et al., 1989; Heinselman, 1973; Romme and Despain, 1989). Some fire ecologists believed that years of fire suppression in Yellowstone had created an accumulation of dangerous amounts of fuel which led to the large catastrophic fires of 1988 (e.g., Bonnicksen, 1989). They hypothesized that reducing this fuel through prescribed burns could have avoided the fires. Other ecologists believed that large stand-replacing fires were unavoidable, a part of the Yellowstone ecosystem and were controlled by regional climate variation at the landscape level (e.g., Christensen et al., 1989; Romme and Despain, 1989). This debate continues today.

Research and lessons learned from the Yellowstone fires are now reflected in both U.S. and Canadian fire management policy. Carefully monitored to protect socioeconomic values, both purposely set prescribed burns and natural fires allowed to run their course, and are synonymous with fire management.

The "let-it-burn" fire policy was eventually replaced by a new policy grounded in the concept of ecosystem management. Ecosystem management is the integration of ecological knowledge and socio-economic values to protect ecosystem integrity over the long term (Holling, 1978). Ecosystem management also considers that the interaction between ecological and social systems is complex and therefore management should be adaptive (Holling, 1978; Walters, 1986). Salafsky et al. (2001) defined adaptive management as follows: "Adaptive management incorporates research into conservation action [here, fire management]. Specifically, it is the integration of design, management, and monitoring to systematically test assumptions in order to adapt and learn" (p.12).

3.1.2 Current Provincial Fire Management

In 1999, *Ontario's Living Legacy Land Use Strategy* (OMNR, 1999a) and the associated *Ontario Forest Accord* (OMNR, 1999b) called for the development of a comprehensive fire management strategy for the province. A *Forest Fire Management Strategy for Ontario* (OMNR, 2004a) was approved in 2004 and provided strategic direction for the management of fire on 107 million ha of Crown and private land. At the same time the province approved a *Forest Fire Management Policy* (OMNR, 2004b). The objectives of both the Strategy and the Policy are: 1) to prevent personal injury, value loss, and social disruption resulting from a forest fire; and 2) to promote the understanding of the ecological role of fire and utilize its beneficial effects in resource management (OMNR, 2004a p.6; OMNR, 2004b, p.63).

The Strategy divided the province into seven Fire Management Zones based on common resource and fire management objectives, land use, fire load, and forest ecology (OMNR, 2004a). For example, the Strategy recognized that "parks and protected areas that contain fire dependent ecosystems will not continue to represent the natural heritage they were designed to protect unless exposed to fire in the coming decades" (OMNR, 2004a, p. 17). Consequently, a distinct management zone called the "Parks Zone" was created. The Parks Zone consisted of large parks, such as Quetico Provincial Park, designated as Natural Environment and/or Wilderness. This zone included approximately 60% of the 9.5 million ha within parks and conservation reserves in Ontario (Davis et al., 2003). Provincial parks that do not fall within this zone are managed according to the Strategies of the surrounding Fire Management Zone unless alternate direction is provided by a Park Management (Master) Plan or FMP.

3.1.3 Current Provincial Park Fire Management

At the same time the province's Fire Management Strategy and policy were introduced, a Fire Management Policy for Provincial Parks and Conservation Reserves (OMNR, 2004c) and a Class Environmental Assessment for Provincial Parks and Conservation Reserves (Class EA) (OMNR, 2004d) were approved. The successful FMPs for Quetico, Rondeau and Pinery Provincial Parks were instrumental in the creation of this policy. For the first time in the province a Class EA required a formal decadal review of the direction of projects outlined in FMPs.

According to the OMNR (2004c), a document entitled *Fire Management Planning Guidelines for Provincial Parks and Conservation Reserves* is currently under development. Once released, this document will direct the process of fire management planning within provincial parks which includes:

1. Considering and documenting the role of fire and preliminary fire management objectives through the preparation of a Statement of Fire Intent;
2. Incorporating fire management direction within relevant protected area planning documents such as interim management statements, park management plans, resource stewardship implementation plans and FMPs; and
3. Preparing FMPs where appropriate.

If fire management direction is not developed and approved for individual provincial parks in a FMP or the previously mentioned planning documents, then fire management is carried out in accordance with the *Forest Fire Management Strategy for Ontario* (OMNR, 2004a).

A document prepared by Ontario Parks titled *Natural Fire Regimes in Ontario* (Van Sleeuwen, 2006) provided a conceptual understanding of the natural fire regimes of the province at the landscape scale. The document provides an overview of all fire frequency studies completed within Ontario and adjacent ecosystems such as the BWCAW. This overview gives estimates of "natural" (usually pre-European settlement) fire frequencies (i.e., mean fire intervals and fire cycles). Park mangers use these fire frequency estimates to set appropriate fire management goals such as annual area burned renewal targets.

Most recently, the *Ontario Parks and Conservation Reserves Act* (2006) includes the recognition that "the maintenance of ecological integrity shall be the first priority" in the planning and management of Ontario's provincial parks. The *Act* defines ecological integrity as: "a condition in which biotic and abiotic components of ecosystems and the composition and abundance of native species and biological communities are characteristic of their natural regions and rates of change and ecosystem processes are unimpeded". Fire is the primary process responsible for the ecological integrity of Ontario's forested areas (Van Sleeuwen, 2006). Therefore, the current understanding of how fire drives ecological integrity should be reflected in the planning and management of Ontario's provincial parks.

3.1.4 Quetico Provincial Park Fire Management

Although Quetico falls within the "Park Zone" of the province's Fire Management Strategy (OMNR, 2004a), it has had a Fire Management Plan (FMP) in place since 1997. As stated earlier, Quetico's fire polices were instrumental in the creation of the provincial Strategy. Fire management within Quetico is directed by its Fire Management Plan (FMP) (OMNR, 1997) and Fire Operations Plan (OMNR, 1999) as they provide specific fire management prescriptions for the Park. New Class EA requirements and fire policy required a review of FMPs every ten years and an amendment to the 1997 FMP. The amendment was required to continue the use of prescribed fire and burns within the Park during the 2008 fire season. The amended 1997 FMP provides interim fire management direction until the approval of the new FMP and the Quetico Provincial Park Management Plan. These documents are expected to be completed in the second quarter of 2009.

The Park recently completed the first step of the fire management planning process by preparing a Statement of Fire Intent (OMNR, 2008a). The Statement of Fire Intent summarizes the findings of Woods and Day's (1977a) fire ecology study and Frech, Caputo & McCulloch's (1999) fire frequency study to provide an overview of the existing knowledge of the role of fire in Quetico and background information relevant to the development of the new FMP. Although Woods and Day (1977a) and Frech, Caputo, & McCulloch (1999) were the only fire frequency studies referenced in the Statement of Fire Intent, Heinselman's (1973) "natural" (pre-European settlement) 100 year fire cycle estimate for the time period 1727-1910 is used to guide the reintroduction of fire in the Park (L. Solomon, personal communication, 2008). According to Heinselman's (1973) fire cycle estimate, the Park has an annual area burned ecological renewal target of 4,500 ha (L. Solomon, personal communication, 2008) (see section 3.3.1 for a more detailed discussion on these estimates).

Within the context of Quetico's FMP, the reintroduction of fire could occur in one of two ways; through prescribed fire or prescribed burn. Their definitions are found in Table 3.1.

Table 3.1 Definition of Prescribed Fire vs. Prescribed Burn.

Prescription	Quetico FMP Definition
Prescribed Fire	Forest fires deliberately utilized in a predetermined area in accordance with a pre-specific and approved burning prescription to achieve preset objectives.
Prescribed Burn	The deliberate, planned and knowledgeable application of fire by authorized personnel and in accordance with MNR policy and guidelines to a specific land area to accomplish pre-determined forest management or other land use objectives.

Source: OMNR, 1997

Since 1997, a total of 3,419 ha have burned within the Park of which prescribed burns or prescribed fires make up 2,711 ha of the 3,419 ha (Solomon, 2007).

Quetico's FMP has divided the Park into two distinct zones based on different fire management goals; the measured zone and the prescribed natural fire zone (Figure 3.1). Quetico's 1997 FMP stated that "the measured zone has been established adjacent to and inside the park boundary as a buffer to protect adjacent lands from unwanted effects of fire" (p. 10). For example, the following needs to be considered with respect to Quetico's fire management:

- public safety (e.g., park visitors, adjacent First Nation community to the west and town of Atikokan to the northwest);
- capital assets and infrastructure within and adjacent to the Park (e.g., hydro corridors, park offices throughout park and Trans-Canada highway to the north); and
- employment (e.g., forestry lands to the north and east and a sawmill which employs 20-30% of the Atikokan population to the north).

Fires within the measured zone are managed according to the provincial Strategy (OMNR, 1997; OMNR, 2004). Specifically, fire suppression within this zone is dependent on what values are threatened, the weather forecast (extreme fire conditions) and the availability of firefighting resources (MNR, 1997). However, Quetico's FMP allows for prescribed burns within the measured zone to reduce fuel buildup (e.g., after the Independence Day blow down) or to achieve ecological restoration goals. The prescribed natural fire zone comprises approximately 63% of the Park, and supersedes the measured zone of the provincial strategy. For example, fires will be allowed to burn if specific criteria are met and socio-economic values are not at risk. Land surrounding the Park is zoned as Intensive. Fires within this zone receive immediate suppression. For example, forestry lands and settlements occur within this zone and must be protected from fire damage (OMNR, 1997).

Figure 3.1 **Quetico's fire management zones (OMNR, 1996).**

3.2 Fire Frequency Studies

3.2.1 Making Sense of Fire Frequency Concepts

Fire regime refers to the nature of fires occurring over an extended period of time. Fire frequency, size, magnitude (intensity and severity), predictability, season, and pattern have all been used to characterise fire regimes (Agee, 1993; Heinselman, 1981; Pickett and White; 1985; Suffling and Perera, 2004). Fire frequency and severity are usually considered in fire frequency studies. Severity is typically defined according to the degree of mortality in canopy trees (Heinselman, 1981; Morgan et al., 2001).
Table 3.2 provides a description of the three general fire regime severity classes and their respective effects.

Table 3.2 Effects of fire regime severity classes.

Fire Regime Severity Class	Effects
Non-lethal surface	Kills few (<10%) canopy (dominant) trees Does not usually influence age structure of dominant trees
Stand-replacing*	Kills the majority (90%) of dominant trees Results in even-aged post-fire cohorts
Mixed severity	A mix of stand-replacing, non-lethal and partially stand- replacing fires Usually results in uneven post-fire cohorts

Sources: Brown, 1995; Heinselman, 1996; Van Sleeuwen, 2006

**all fires in this research are assumed to be stand-replacing*

Many fire frequency concepts exist (see Johnson and Van Wagner, 1985 and Johnson and Gutsell, 1994; Merrill and Alexander, 1987; Romme, 1980). Inconsistent and incorrect applications of these concepts throughout the literature have resulted in considerable ambiguity and confusion. Fire frequency has been most commonly expressed in terms of mean fire interval (e.g., Johnson 1979; Swetnam, 1993; Woods and Day, 1977a) and fire cycle (e.g., Heinselman, 1973; Van Wagner, 1978; Johnson and Van Wagner, 1985; Reed, 1998). Table 3.3 provides definitions of the fire frequency concepts used in this research and their respective relationships.

Table 3.3 Definitions and relationships between different fire frequency concepts.

Concept	Definition	Relationship
Fire frequency	The number of fires that occur within a given time period.	Often expressed in terms of *mean fire interval* or *fire cycle*.
Mean fire interval	The expected number of years between two successive stand-replacing fires at any given location.	Equivalent to the *fire cycle* only when fire frequency data fit (or are fitted) to the negative exponential fire model.
Fire cycle/fire rotation/burning rate	The number of years expected to burn an area equal in size to the study area.	Equivalent to the *mean fire interval* only when fire frequency data fit (or are fitted) to the negative exponential theoretical fire model.
Hazard (probability) of Burning	The hazard (probability) of any given location within the study area being burned in a stand-replacing fire.	*Mean fire interval* and *fire cycle* are the reciprocal of the hazard of burning only when fire frequency data fit (or are fitted) to the negative exponential theoretical fire model.

Sources: Johnson and Gutsell, 1994; Reed 2006; Van Sleeuwen, 2006

Johnson and Van Wagner (1985) and more recently Johnson and Gutsell (1994) equated the concept of fire cycle with the concept of mean fire interval. Johnson and Gutsell (1994) provided an explanation of the relationship between these two concepts given certain assumptions: “Since by definition frequency is the inverse of the return period, the fire cycle (average fire interval) is the inverse of annual percent burn (fire frequency). Also, each concept is expressed on a per unit basis (element) or as a proportion of the whole population (universe)” (p. 254). However, this relationship is true only when fire frequency data are fitted to the negative exponential theoretical fire model where fixed proportions of the landscape burn each year (Reed, 2006). In such cases, the average forest age has also been articulated as an estimate of fire frequency (Agee 1993; Bergeron et al.; 2001). This theoretical fire model and the Weibull fire model will be explained further in section 3.2.3.

Even though the fire cycle concept has been used in recent fire frequency studies, Reed (2006) suggested that it should no longer be used to characterise fire frequency. He explained that “the fire cycle concept represents a random variable and therefore even if the ‘expected’ value of this random time is used, it does not coincide with the local mean fire interval (or the reciprocal of the local hazard of burning)” (p.1886). Furthermore, fire cycle estimates normally exceed the mean fire interval and only when fire frequency data are fitted to a negative exponential theoretical model will the mean fire interval be equal to the fire cycle (Reed, 2006). The mean fire interval is also a more useful and intuitive measure for managing a given piece of land (e.g. deciding whether to put out a small fire that may spread). Consequently, the goal of this research was to obtain mean fire interval estimates as opposed to fire cycle estimates. This also coincided well with parallel research being undertaken by the OMNR Forest Policy Section. The OMNR are using the 1966 FRI GIS database digitized in this thesis along with other data to obtain estimates of natural[2] mean fire interval spatial, temporal and stochastic variability using Boreal Forest Landscape Dynamics Simulator (BFOLDS) (Perera et al., 2002), a spatially explicit stochastic simulation model.

Quetico Provincial Park's fire regime is primarily driven by large stand-replacing fires, which account for the majority of area burned and thus are responsible for the composition, structure and function of its ecosystems (Heinselman, 1996). Considering this, and the available data (1966 FRI and digital fire atlas), the focus of this research was to determine the mean fire interval of stand-replacing fires. The following section provides an overview of the collection of fire frequency data and the assumptions, limitations and advantages of such data.

3.2.2 Fire Frequency Data Collection

Fire intervals have typically been reconstructed by dendroecological analysis of fire scar records. Fire scars are created at the base of tree trunks by non-lethal surface fires. However, fire-scarred trees may represent residual trees or patches of trees within large stand-replacing fires, or are located near the edge of these fires. The annual growth-rings on tree cores or wedges are counted back to fire-scars (pointer years) and then cross-dated using a master chronology giving the exact year and even season of the fire that caused the scar (for a detailed description of these methods see Stokes and Smiley, 1966 or Schweingruber, 1996).

In areas such as the boreal forest where stand-replacing fires are common and therefore fire scar evidence is rare, time-since-fire estimates have been obtained through tree cores (Heinselman, 1973; Woods and Day, 1977a; Weir et al., 2000). Charcoal and pollen records detected in lake (Clark, 1990, Swain, 1978; Swain, 1980), soil (Carcaillet, 1998), peat sediments (Kuhry, 1994), and glaciers (Taylor et al., 1996) have been used to study the historical frequency of fire. Such data provides indirect evidence of either stand or non-stand replacing fire occurrence. However, because wind can blow pollen and charcoal considerable distances, the exact location of these fires is indeterminate

[2] The Park keeps a record of whether fires were started by lightning or humans. Therefore, the OMNR was able to run the BFOLDS simulation based on natural (lighting) fires only.

(Heinselman, 1996). Historical records such as historical land surveys (Suffling and Wilson, 1994; Suffling, 1982), individual fire records (OMNR, no date), annual fire maps (Li, 2000; Bridge, 2001) and paleomagnetism records (Kletetschka and Banerjee, 1995) are also useful for verifying certain fire dates and validating study methods. Egan and Howell (2001) provide a comprehensive overview of these methods. Individual fire records may include fire characteristics such as date, approximate coordinates, size, cause, age at burn, pre-burn vegetation, etc. for each fire; whereas annual fire maps (digital fire atlases) show the spatial locations of burned areas.

The majority of fire frequency studies have attempted to estimate fire cycle by constructing maps of stand origin/time-since-fire (Heinselman, 1973; Johnson and Larsen, 1991; Larsen, 1997; Weir, 2000). Time-since-fire maps result in a comprehensive understanding of stand-replacement history; however, they cannot characterise non-lethal or mixed severity fire regimes. Time-since-fire maps record the occurrence of the most recent fire in each unit of the landscape. Consequently, they require considerable resources and time to produce. The forest age attributes of a forest resource inventory (FRI) (Li, 2000; Suffling, 1982) and digital fire atlases (Bridge, 2001; Frech, Caputo, & McCulloch, 1999) can be used to approximate a time-since-fire map when complete inventory is unattainable. The majority of time-since-fire mapping methods use air photos to identify different aged patches of vegetation. Field data are then collected to determine the fire boundaries and validate fire years based on ages of tree cores, and fire-scars respectively. Currently, GIS is used to carry out the majority of data storage and pre-statistical analysis processing tasks in fire frequency studies (Cyr et al., 2007; Grenier et al., 2005; Lauzon et al., 2007).

Time-since-fire maps rely on the fundamental assumption that all stands establish immediately following fire. In reality however, other disturbances such as severe insect attacks, blowdown by strong winds, or logging can also cause stand-replacement. However, fire is the primary natural disturbance responsible for the establishment of continental boreal forest communities where logging has not occurred. Another limitation of time-since-fire maps is that the fire year is a minimum estimate if not validated by either historical records or fire scars. If none of these records are available, an assumption is made that time of death is equal to time of post-fire community establishment otherwise know as recruitment. This assumption is acceptable because boreal tree communities typically recruit within 1 to 4 years following fire (Heinselman, 1996; Greene et al., 2004). Gutsell and Johnson (2007) defined “recruitment” as a tree that germinates from seed or sprouts from basal buds of live vegetative structures (e.g., lignotubers, rhizomes, roots, or burls) and survives for at least a year. Boreal forest fire return intervals also typically occur within the lifespan of post-fire cohorts and therefore understory cohorts rarely replace post-fire cohorts in the canopy (Master 1990; Bergeron and Archambault, 1993, Weir et al., 2000). Studies in the Canadian boreal forest have found that conifer species with serotinous cones such as Black Spruce and Jack Pine, recruit sexually within 4 years after fire, with recruitment peaking in either the first and second post-fire summers (Greene et al., 2004) or the second and third post-fire summers (Charron and Greene, 2002). There appears to have been only two boreal forest studies that have attempted to make causal connections between fire and asexual recruitment processes (Greene et al., 2004; Schimmel and Granstrom, 1996). Greene et al. (2004) found that asexual recruitment of a Trembling Aspen community in the boreal forest of Northern Quebec occurred within 2 years post-fire. Further study is needed to apply recruitment patterns from local community studies to landscape level studies (Gutsell and Johnson, 2007).

Many studies have grouped the age time-since-fire map data into 5, 10 or 20 year age classes to deal with inaccuracies in dating trees (e.g., the time it takes tree seedlings to grow to the height at which they were cored as an adult) and fire events (e.g., Grenier, 2005; Heinselman, 1973; Weir, 2000). For example, when aging a tree from breast height (1.3 m above ground level) or any other height (e.g., as close to root collar as possible), a bias is introduced. Several studies have demonstrated differences in the time it takes a seedling to grow to breast height in various regions using age/height regression analyses (Gutsell and Johnson, 2002; Parisien, Sirois and Parent, 2005; Vasiliauskas and Chen, 2002). However, high landscape level variability requires the sampling of each stand to correct age inaccuracies for studies involving multiple stands (Gutsell and Johnson, 2002; Parisien, Sirois and Parent, 2005). For a detailed review of the techniques used to establish accurate fire dates using dendroecological methods and their limitations see Madany et al., (1982) and Zackrisson (1977).

Fire regimes have been predicted using rule-based classification (Kasischke et al., 2002; Morgan et al., 1996), discriminant function statistical analysis (Johnson and Larsen, 1991; Reed, 1998) and simulation modelling (He and Mladenoff, 1999; Perera et al., 2002) strategies using stochastic, empirical and biophysical approaches. Gardner et al. (1999) provided an overview of the three strategies and Keane et al. (2004) provided a review of the three approaches. Both authors assessed their respective advantages and limitations and therefore these will not be addressed in detail.

This research employed a statistical analysis strategy given the available data and the opportunity to collaborate with a statistician with expertise in survival analysis. An empirical Kaplan Meier approach was taken as it has been deemed the most accurate for estimating current fire frequencies (Keane et al., 2004) and could provide estimates of fire frequency spatial and temporal variability.

However, the following limitations must be realized for the statistical methods employed during this research (adopted from Keane et al., 2004):

1. the analysis is aspatial;
2. cause and effect cannot be inferred (e.g., fire suppression has decreased fire frequency) to deduce landscape-scale fire dynamics (Keane et al., 2004);
3. one must be careful when generalizing the findings for other areas or other time periods (outside the extents of this research) as differences in climate, topography, soils, vegetation and lightning occurrence, which drive fire frequency spatial and temporal variability, cannot be accounted for (Baker, 1989; Keane et al., 2004; Knight, 1987) (see Literature Review section 3.3 for a more detailed discussion); and
4. fire frequency estimates are only applicable to the spatial scale at which they were made. For example, multiple-scale factors (e.g., wind interactions, topography aspect and elevation, etc.) are difficult, and possibly inappropriate, to incorporate at the scale typically used for estimation. Therefore, such factors are difficult to account for using a single statistical approach (Keane et al., 2004).

The long time period and spatial extent covered by this study's dataset will allow for an accurate estimate of Quetico's current fire frequency and will determine if the Park's fire frequency varies both spatially and temporally. The FRI GIS database can be used to run the BFOLDS simulation model using fire weather data for the time period 1963-2003[3] (Perera et al., 2002). The fire frequency estimates from this research can be compared to the results of the BFOLDS simulation. The OMNR uses BFOLDS to characterise and predict landscape level forest fire regime and forest cover dynamics for forestry policy formulation in Ontario.

The following section provides an overview of the statistical analysis methods employed in this research.

[3] BFOLDS uses a Fire Weather Index (FWI) stream from this time period (M. Gluck, personal communication, 2008; Perera et al., 2002)

3.2.3 Burning Rate and Forest Age Distribution Approaches

Initial fire frequency studies consisted of the collection of informal evidence of fire occurrence, based usually on fire scarred-trees. However, these methods did not allow for the estimation of fire frequency (mean fire interval or fire cycle) spatial and temporal variability over large areas. Heinselman (1973) pioneered a more comprehensive approach in his BWCAW study by mapping stand origin (time-since-fire) using recent fire boundaries from aerial photography, the ages of trees known to grow well following stand-replacing fire, and fire-scarred trees. Heinselman (1973) expressed the area covered in each time-since-fire class as a cumulative proportion of the study area to estimate the general survivorship from fire in the study area (refer to section 3.2.4 for current methods). The burning rate (Br) is an area-based estimate of fire frequency (fire cycle) indicating how fast the total area could be burned once. The fire cycle is the reciprocal of the proportion of annual area burned. Heinselman (1973) used the following computation to estimate the fire cycle of the BWCAW:

Equation 1 Burning rate

Br = (T/B)*A

Where A is the size of the study landscape (ha), B is the total area burned (ha) during T years and T is the number of years covered by the study. In this research, the fire cycle for the time period 1921-2007 was calculated for the Park using a digital fire atlas.

Heinselman's (1973) methods were expanded by Van Wagner (1978) whereby he equated the present standing age distribution of the forest with the time-since-fire distribution. The reverse cumulative standing age distribution has also been used (Bergeron, 1991; Johnson and Larsen, 1991; Larsen, 1997) but is incorrect (Huggard and Arsenault, 1999). Van Wagner (1978) fitted the time-since-fire distribution to a theoretical model, the negative exponential. The exponential probability density function is expressed as:

Equation 2 Exponential probability density function

$$f(x;\lambda) = \begin{cases} \lambda e^{-\lambda x} & , \ x \geq 0, \\ 0 & , \ x < 0. \end{cases}$$

The probability density function of the exponential distribution is represented here as λe − λx. This model for the random variable *x* has a single scale parameter, λ which is the hazard rate and is assumed to be constant (independent of time-since-fire) (Van Wagner, 1978). It is the most commonly used fire frequency model for studies in the boreal forest. The negative exponential theoretical model survival curve is presented in Figure 3-2.

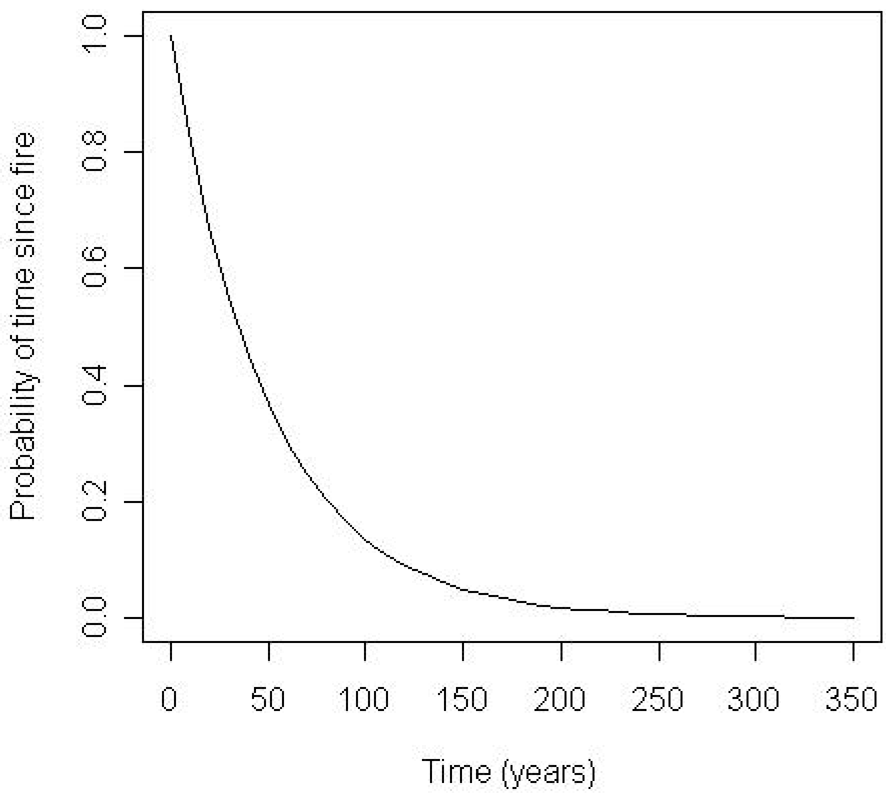

Figure 3.2 Negative exponential theoretical model survival curve.

At around the same time as Van Wagner's study, Johnson (1979) applied the Weibull theoretical model to fire frequency studies. The negative exponential model is a special case of the Weibull model. The probability density function of the Weibull model is expressed as:

Equation 3 Weibull probability density function

$$f(x;k,\lambda) = \frac{k}{\lambda}\left(\frac{x}{\lambda}\right)^{k-1} e^{-(x/\lambda)^k}$$

The probability density function of the Weibull distribution is represented here as: $(k/\lambda)(x/\lambda)^{(k-1)}e^{-(x/\lambda)^k}$. The Weibull model for the random variable *x* is characterised by a scale (λ) and a shape (*k*) parameter whereby the shape parameter can be adjusted to regulate the hazard rate (hazard of burning) with time-since-fire. To explain, if the hazard rate decreases with time-since-fire $k < 1$. If the hazard rate increases with time-since-fire, then $k > 1$. If the hazard rate is constant (independent of time-since-fire), then $k = 1$ and is simplified to the negative exponential model (Johnson, 1979; Johnson and Gutsell, 1994; Johnson and Van Wagner; 1985). The Weibull theoretical model survival curve is presented in Figure 3-3.

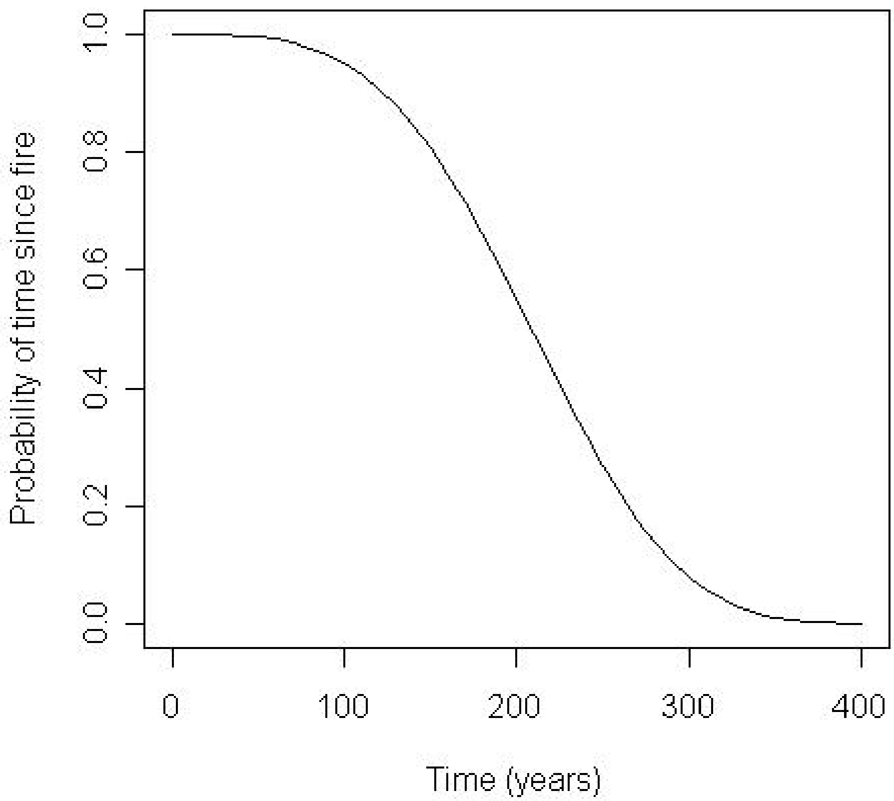

Figure 3.3 **Weibull theoretical model survival curve.**

Johnson and Gutsell (1994) defined hazard of burning as “the per capita age-specific mortality from fire. It is the probability of fire occurring in an interval, assuming survival up to the beginning of the interval” (p. 244). It is important to differentiate the hazard of burning from fire hazard. Fire hazard is an index used to qualify the potential for fire based on fuel load from vegetation structure and phenology (Van Wagner, 1977). The hazard of burning is a statistical concept which refers to a hazard rate (Johnson and Gutsell, 1994). It is proportional to the instantaneous probability of fire at any given point within a sampling unit (i.e., forest stand) (D. Matthews, personal communication, 2008). To avoid confusion, the hazard of burning will be referred to as the probability of burning in this research. This choice of terminology is particularly appropriate as the emphasis in this study was on the use of non-parametric survival analysis methods which typically estimate probabilities.

The use of the negative exponential and the Weibull theoretical fire models represented the first time that explicit statistical distributions were used to test hypotheses on the spatial and temporal patterns of fire frequency. A review of the theory and use of these two models was provided in Johnson and Van Wagner (1985). Li (2000) explains that both theoretical models apply to "homogeneous, stochastic processes and therefore need to approximate the following two stability criteria: 1) all forest stands have the same fire regime (i.e., stand-replacing) and 2) each of the stands has, on average, a constant fire regime during the time period of study" (p.131). This methodology was reviewed by Johnson and Gutsell (1994) and further developed by Reed (1994) and Reed (1998) to deal with ecologically problematic assumptions. In summary, several authors have identified problematic assumptions mainly age-independent flammability (Johnson et al., 1990; Johnson and Larsen, 1991; Romme and Despain 1989) as well as spatially and temporally homogenous fire frequencies (Baker, 1989; Suffing, 1990; Boychuck et al., 1997). For example, age independent burning rates (Johnson, 1990; Johnson and Larsen, 1991; Van Wagner, 1978) and age dependent burning rates (Schimmel and Granstrom, 1997; Cumming, 2001; Hellberg et al., 2004; Tanskanen et al., 2005) have both been observed in boreal ecosystems. Non-steady state ecosystem conditions have also been observed which contradict the assumption of spatial and temporal homogeneity (Baker, 1989; Boychuck et al, 1997; Suffling, 1990). Boychuck et al. (1997) suggested that a stable forest age distribution should not be expected even at large scales due to differences in contagion of fire and the proportion of the landscape burned from year to year.

The assumptions, advantages and limitations of Heinselman's (1973) burning rate approach, the negative exponential (Van Wagner, 1978) and Weibull (Johnson, 1979) theoretical fire models and Reed's (1998) Maximum Likelihood Estimator (MLE) fire model are summarized in Appendix A. To overcome these limitations, the statistical technique of survival analysis has been adopted in current fire frequency studies (Reed, 1994, Reed, 1998; Grenier et al., 2005). Such analysis methods are discussed in the following section.

3.2.4 Survival Analysis Approach

3.2.4.1 Non-parametric methods

The majority of recent fire frequency studies have used survival analysis to estimate the fire cycle. Survival analysis is a statistical technique that provides a way to estimate time to event data, where forest stand death due to fire is considered the event of interest. There are survival analysis methods based on either parametric (Van Wagner, 1978; Johnson 1979; Reed, 1994; Reed, 1998) or non-parametric (e.g., Kaplan Meier, 1958) (Cyr et al., 2005; Lauzon et al., 2007) estimation of the survival function. Non-parametric methods do not fit fire frequency data to a theoretical model or make assumptions with respect to the probability of burning. Rather they are directly derived from the empirical data.

Time-since-fire map data are used in fire frequency studies to construct survivorship/mortality distributions to estimate fire frequency (Reed, 1994; Reed, 1998). Johnson and Gutsell (1994) explained the interpretation of these distributions in fire frequency studies: "time-since-fire (survivorship) distributions represent the proportion of the entire landscape surviving longer than the time period under study…" (p.244). Most studies have plotted the forested area (in ha) in each forest stand age-class to estimate the survivorship from fire giving an "area-based" fire frequency (fire cycle) estimate (Reed, 1998; Johnson and Gutsell, 1994; Bergeron et al., 2004; Van Wagner et al., 2006). However, for this research, the survivorship function is used to plot the probability of a landscape unit (FRI forest stand) being replaced by fire for each stand age (time-since-fire) class. The survivorship function is used to estimate the "point-based" mean fire interval using a historical time-since-fire map (1966 FRI) updated with a current digital fire atlas. The probability of burning function can also be plotted to infer the probability of a forest stand being killed by a stand-replacing fire in each time-since-fire class.

Based on the literature reviewed[4], most fire frequency studies (except two recent studies by Cyr et al., 2007 and Lauzon et al., 2007) appear to have fitted fire frequency data to one of the two theoretical fire models instead of first plotting the empirical distribution and testing the goodness of fit to each of these models. If the empirical distribution was tested, it was not reported in the literature. Generally the fewer parameters one uses (and therefore assumptions one makes) when using mathematical models based on probabilistic explanation (e.g., survival analysis), the more reliable the results will be (D. Matthews, personal communication, 2008). Each parameter can add a level of uncertainty. Non-parametric methods should be considered first to avoid curve-fitting the data when one does not know if the empirical distribution follows the curve of a theoretical model (D. Matthews, personal communication, 2008). Once the empirical distribution is plotted, comparative simulation can be used to test the fit of the data to theoretical models.

Spatial biases and temporal parameters which may have affected fire frequency can be explored by non-parametric methods by partitioning the data in time and space as long as at least one complete observation (fire interval) ends within the partition of interest.

[4] Based on a systematic search of Web of Science, Scopus, Google Scholar and various U.S. and Canadian government databases.

3.2.4.2 Censoring fire frequency data

Forest stands where a fire interval (the time between two successive stand-replacing fires) has not been observed represent incomplete observations on the time interval of interest. Thus, forest stands in the time-since-fire map that have not burned since the year of the mapping are considered right-censored, or incomplete. Incomplete data represent a lower bound on the time to fire event estimate because all that is known is that these stands have survived unburned to their current age. Complete observations are data where the time between two successive stand-replacing fires (fire interval) is known.

Van Wagner's (1978) parametric method using data fitted to the negative exponential theoretical model does not require the censoring of incomplete observations to estimate the fire cycle. It therefore treats time-since-fire (right-censored) and fire-interval (complete) data as equal. Incomplete observations can be treated as complete observations because the hazard of burning is assumed to be independent of time-since-fire (constant) leading to the special "no-memory"[5] property of the negative exponential (see Polakow and Dunne, 1999 for a detailed explanation). Johnson's (1979) parametric method (data fitted to the Weibull theoretical model) also allows for the estimation of a fire cycle. According to Polakow and Dunne (1999), however, although time-since-fire and fire interval can be equated when fire frequency data are fitted to the exponential model, when using the Weibull model, incomplete data must be right-censored. The method of censoring data does not appear to have been applied correctly in previous studies using Kaplan-Meier methods as incomplete observations have not been right-censored (Cyr et al., 2005; Lauzon et al., 2007)[6]

Fire frequency studies based on parametric and non-parametric methods have provided various rationales for censoring fire frequency data. Old-growth and uneven-age stands were treated as censored in survival analysis (e.g., Bergeron et al. 2001, Bergeron et al. 2004). It is possible that old growth and uneven-aged stands were censored because they are not the result of stand-replacing fires, but the result of partial stand-replacing and/or surface fires, and gap dynamics such as insect outbreaks and/or wind throw. However, an explanation as to why these stands were censored was not given in the literature. In the same studies, censoring also occurred where the time-since-fire could only be estimated by the age of the oldest canopy tree, instead of the age of the five or more trees originating during a particular age-class (usually 10 or 20 year). The rationale given for censoring stands aged using the oldest individual tree in a stand was that time-since-fire estimates were not as accurate as stands where the age was determined using five or more trees. Although these studies for the most part

[5] Stands do not remember the time they were last burned and therefore each stand has the same probability of burning regardless of age.
[6] They did not explicitly state that they right-censored time-since-fire observations.

described the reasoning for censoring data, they did not explicitly state whether or not the data were "right-censored" in the survival analysis. This is an example of how previous studies (Cyr et al., 2005 and Lauzon et al., 2007) have not been explicit in methodology, or have possibly incorrectly treated incomplete observations as complete observations in non-parametric survival analyses.

The weakness of non-parametric survival analysis is that if the distribution does not fit a theoretical model, then the censored data do not have the same weight in the analysis as the non-censored data and can lead to an underestimation of the length of the fire cycle (Lauzon et al., 2007). However, the parametric tests discussed earlier, require fitting the data to negative exponential or Weibull models that may not prove a good fit to the empirical data. Both non-parametric Kaplan-Meier and recent parametric (Reed, 1994; Reed, 1997; Reed, 1998) methods provide standard errors to obtain a 95% confidence interval on the fire frequency estimate.

3.2.4.3 Partitioning fire frequency data

Most empirical fire frequency distributions are multimodal (Heinselman, 1973; Van Wagner, 1978; Johnson, 1992). That is, fire frequency has varied in time and space. For example, natural climate variation at the end of the Little Ice Age ~ 1850 (Bergeron 1991; Grenier et al., 2005; Lauzon et al., 2007; Lesieur et al., 2002), distance to fire break (e.g., lake, wetland, rock barren or recent burn) (Bergeron, 1991; Larsen, 1997), topography (Turner and Romme 1994; Heyerdahl et al., 2001) and composition and spatial configuration of vegetation (Bergeron, 1991; Cumming 2001; Romme and Despain, 1989) have been attributed to fire frequency variability.

Johnson and Larsen (1991) provided graphical methods for partitioning mixed distributions over time. Partitioning is performed to determine the processes that caused the change in fire frequency over time. This process can be either ecological or human in nature. Reed (1998) explains that to avoid selection bias, change points must be made independent of the data. Therefore, change points should be chosen before plotting the distribution. For example, the end of the Little Ice Age (~1850) in northeastern North America has been hypothesized as a time where fire frequency has changed (Bergeron and Archambault, 1993; Bergeron et al., 2001; Stocks, 1993).

Many fire frequency studies assume spatial homogeneity if the study area is homogeneous at landscape level (e.g., within a single ecoregion such as Quetico Provincial Park). Spatial partitioning can be performed if the study area is heterogeneous at the landscape level. Fire frequency data can also be partitioned based on potential spatial biases such as distance to fire break, topography and surficial deposit. This is done in an attempt to achieve homogeneous distributions and subsequently infer which ecological factors influence fire frequency variability. However, several studies have been unsuccessful at spatially partitioning time-since-fire maps in this manner (Johnson and Larsen 1991, Masters 1990, Johnson et al., 1990). That is they have not been able to produce homogeneous distributions through data partitioning. Bergeron (1991), Bergeron (1993), and Fryer and Johnson (1988) also explained that large stand-replacing fires (i.e., those that control fire frequency) tend to burn independent of topography especially given extreme fire weather conditions.

3.2.4.4 Thesis approach

Non-parametric Kaplan-Meier survival analysis will be used to estimate fire frequency in this research. As noted previously, Reed (1994 and 1998) recommended the use of a parametric model to account for: 1) older stands that have been subject to a different hazard of burning than younger stands because they have lived through more recent time periods; and 2) the contagion of fire as stands close to one another are more likely to have burned during the same fire than stands that are far away. However, it was not necessary to address these limitations during this research because 1) non-parametric probability of burning estimates are empirically derived, and 2) a point-based mean fire interval (without area weighting) was estimated rather than an area-based fire cycle (see Results section 5.4.1 for an explanation of area weighted vs. non-area weighted techniques).

Few fire frequency studies have been completed for long time periods and broad spatial scales. The variability and heterogeneity of fire regimes are rarely quantified and multiple datasets are needed to aggregate across both spatial and temporal scales (Niklasson and Granstrom, 2000; Heyerdahl et al., 2001; Morgan et al., 2001; Weir et al., 2000). The fire frequency data generated during this research was managed in a GIS database and was therefore easily partitioned to test the spatial and temporal variability of fire frequency. Non-parametric Kaplan Meier survival analysis methods used in this research allowed for the statistical comparison of fire frequency between time periods to account for stochastic nature of fires. Significance tests based on survival curves and mean fire interval confidence interval estimates were used.

In order to compare the existing knowledge of Quetico's fire regimes with the results of this research, the existing fire frequency studies for Quetico and the adjacent BWCAW are reviewed in the following section. First, an overview of boreal fire regimes is given for context.

3.3 Boreal Fire Regimes

Early opinions were that fire regimes of the North American boreal forest were primarily characterised by stand-replacing fires which burned at short intervals (Heinselman 1973; Johnson, 1991; Van Wagner, 1978). However, later studies have found that fire frequency varies considerably from one region to another (Bergeron, 1991; Bergeron et al., 2001; Engelmark et al., 1994; Larsen 1997; Suffling et al., 1988; Weir et al., 1999; Van Wagner et al., 2006).

Changes in fire frequency have been attributed to regional climate variation (Stocks, 1993; Wotton and Flannigan, 1993; Flannigan et al., 2000; Bergeron et al., 2001). For example, an increase in the distribution of precipitation and humid air masses following the end of the Little Ice Age (approximately 1850 AD) has been correlated with a decrease in fire frequency in eastern (Bergeron and Archambault, 1993; Bergeron et al., 2001) and western (Larsen 1996; Weir et al., 2000) North American boreal forests. Other regional scale factors have also been suggested as being important in controlling boreal fire regimes. For example, the proportion of deciduous trees in the landscape (Hély et al., 2001; Cumming, 2001) and water bodies or wetlands that act as natural fire breaks (Hienselman, 1973; Larsen 1997; Bergeron 1991; Romme and Knight 1981) have been recognized as inhibiting the spread of fires and therefore the fire cycle. Fire frequency variability has also been attributed to variation in surficial deposits (Harper et al., 2002; Bergeron et al., 2004). In addition to these regional scale factors, physiographic factors that interact at multiple spatial scales, such as topography, aspect and elevation have also been known to influence fuel moisture and thus affect the probability of fire (Heyerdahl et al., 2001; Whelan, 1995; Rowe and Scotter, 1973; Ryan, 2002).

Humans have influenced the frequency of fire. For example, early settlers used fire to clear forested land for agriculture and several studies have found an increase in the annual area burned during this time period (Lefort et al., 2003; Niklasson and Granström 2000; Weir et al. 2000). Many authors have hypothesized that fire suppression has influenced the fire frequency (annual area burned) of Canadian boreal forests (Cumming, 2005; Martell 1994, Martell 1996, Stocks 1991; Ward and Tithecott 1993; Weber & Stocks 1998, Li 2000, Ward & Mawdsley 2000). Others have suggested that fire suppression has had minimal influence on the frequency of fire (Miyanishi & Johnson, 2001, Miyanishi et al., 2002; Bridge et al., 2005) especially prior to the use of water bomber tankers in the 1970s. Several fire frequency studies in and around Ontario reported no change in the fire cycle since the 1920s (Bergeron, 1991; Bridge et al., 2005; Heinselman, 1973; Suffling et al., 1982; Woods & Day 1977a).

Many studies have predicted that climate change will have a considerable influence on boreal fire regimes in the future. The predicted combination of warmer temperatures and frequency of drought is expected to increase the length of the fire season and cause more frequent and severe fires in Ontario (Flannigan et al., 2005; Gillett et al., 2004; Wotton and Flannigan, 1993; Wotton et al., 2003) and specifically northwestern Ontario (Lemieuk, 2007; Racey, 2004; Thompson, Flannigan, Wotton & Suffling, 1998). Through the statistical reconstruction of tree-ring width data, Girardin and Mudelsee (2008) compared 21st century forecasts of fire occurrence (number of large forest fires per year) with historical estimates over the past 240 years, for northwestern Ontario and eastern boreal Manitoba. They asserted that an increase in precipitation (consistent with Lemieuk's (2007) findings) will be insufficient to compensate for increasing temperature in this region. By 2061-2100 fire occurrence could increase by more than 34% compared with the past 240 years and reach the upper limit of its historical range.

Duchesne and Hinrichs (1996) provided a comprehensive summary of fire frequency studies that have attempted to characterise the "natural" mean fire interval and/or fire cycle for Canadian northern ecosystems before and after European settlement. Van Sleeuwen (2006) did the same for Ontario and adjacent regions. Ward and Tithecott (1993) estimated that Ontario's fire cycle ranged from 20 to 135 years, with an average of 65 years in the pre-suppression period (pre-1920).

3.3.1 Fire Frequency Studies within Quetico and the BWCAW

To ascertain the current understanding of fire frequency within the region, fire frequency studies completed within Quetico (Bridge, 2001; Frech, Caputo, & McCulloch, 1999; Woods and Day, 1977a) and the adjacent BWCAW (Fall and Lertzman, 1999; Heinselman, 1973; Heinselman, 1981; Swain, 1973; Van Wagner, 1978) were reviewed. Table 3.4 provides a comprehensive summary of this review and includes the: 1) type of fire frequency estimate (mean fire interval or fire cycle); 2) time period of the estimate; 3) size of the study area; 4) fire regime studied (stand-replacing or mixed); and 4) methods employed.

Table 3.4 Quetico and BWCAW Fire Frequency Estimates.

Author(s) and Date	Fire Regime Characterised	Mean Fire Interval Estimate (Time Period)	Fire Cycle/Fire Rotation Estimate (Time Period)	Study Area Location and Size (Hectares)	Methods (fire frequency data collection and statistical)
Woods and Day (1977b and c)	Mixed	870 years (1850-1969) 66 years* (1870-1919) 202 years (1920-1969)	200 years (1920 - 1977) 78 years* (1850 - 1920)	Quetico Provincial Park 79,118 ha (21% of the Park)	Age-class structure inferred from aerial photography, 1948 and 1966 FRI and tree cores (stand origin) and mapping of old, unrecorded fires through the use of fire scars.
Heinselman (1973) ***Fall and Lertzman (1999)***	Mixed	See Table 3.5	100 years* (1727-1910) 1900 years (1917 – 1973) 58 years* (1766 - 1917)	BWCAW 526,091 ha	Age-class structure inferred from aerial photography, 1948 resource inventories, tree cores (stand origin) and mapping of old, unrecorded fires through the use of fire scars. Performed Maximum Likelihood Estimator (MLE) survival analysis on Heinselman's (1973) data

Van Wagner (1978)	Stand-replacing	N/a	50 years*	BWCAW	Using a probability approach fitted Heinselman's (1973) data to negative exponential theoretical model
Swain (1973) (1980)	Mixed	65 years (1580-1970)	N/a	Around a single lake (Hug Lake) within the BWCAW	Based on charcoal and pollen analysis
Bridge (2001)	Stand-replacing	N/a	Area based fire cycle 326 years (1921-1995) 331 years (1921-1995) 890 years (1972-1995)	Quetico 4W (Hills (1959) site region)	Calculated the burning rate using a partial time-since-fire map (digital fire atlas) MLE survival analysis (using digital fire atlas) MLE survival analysis (using digital fire atlas)
Frech, Caputo, & McCulloch (1999)*	N/a	N/a	379 years (1976-1998)	Quetico Provincial Park	N/a

*Notes: * "natural" fire frequency estimate; **cited in Quetico's Statement of Fire Intent, although unable to obtain report*

The region's "natural" (pre-European settlement) fire cycle has been estimated to be 78 years (Woods and Day, 1977a), 58 years (Fall and Lertzman, 1999) and 100 years (Heinselman, 1973). Woods and Day (1977a) estimated the natural mean fire interval for Quetico to be 66 years. However, this estimate included mixed fire severities (i.e., non-lethal, partially stand-replacing and stand-replacing fires) and their study area covered only 21% of the Park. Their data have not been reanalyzed using current statistical methods and therefore confidence intervals are not available for their estimates[7]. Heinselman (1973) estimated the mean interval between major fires within the BWCAW to be 48 years during the time period 1542-1972. Van Wagner (1978) estimated the fire cycle to be 50 years after fitting Hienselman's (1973) data to the negative exponential theoretical model. Swain (1973) estimated a mean fire interval of 65 years for the time period 1580-1970. However, his estimate covered only a single area surrounding Hug Lake within the BWCAW.

Heinselman's mean interval between major fires estimate (48 years), Van Wagner's hypothesized model of an expected fire cycle for boreal ecosystems (50 years) and Fall and Lertzman's natural fire cycle estimate (58 years) are in agreement. However, the similarity in Fall and Lertzman's and Van Wagner's estimates is likely explained by their using the same dataset (Hienselman's 1973 data) and both methods required fitting fire frequency data to the negative exponential theoretical model and therefore results should be interpreted with caution. Heinselman made his estimate based on the fire history he was able to construct using air photos, resource inventories, stand origin dates and fire scars.

Bridge (2001) estimated a 331 year (1921-1995) and 890 year (1972-1995) fire cycle for Queitco using the digital fire atlas and MLE survival analysis. Bridge (2001) obtained similar results for the time period 1921-1995 at 326 years using the digital fire atlas and a burning rate approach.

Woods and Day (1977b) and Heinselman (1973) also provided mean fire interval estimates for the major vegetation communities within Quetico and the BWCAW and these are summarized in Table 3.5.

[7] Day, R.J. was contacted; however, the data used in the 1977 study could not be acquired. Day, R.J. was unable to verify if or where the original data exists (R.J. Day, per comm., 2007).

Table 3.5 “Natural” fire frequency estimates for the regions fire-dependent forest communities.

Forest Community	Heinselman’s (1973) mean fire interval estimates (fire regime)	Heinselman’s (1981) fire cycle estimates (fire regime)	Woods and Day’s (1977b) mean fire interval estimates (fire regime)
Jack Pine	28 years (stand-replacing) 50 years (stand-replacing)**	50 years (stand-replacing)	80-120 years (Mixed)
Black Spruce	28 years (stand-replacing) 50 years (stand-replacing)**	50 years (stand-replacing)	90-120 years (Mixed)
Red Pine and White Pine	36 years (non-lethal to partial stand-replacing) 160 years (stand-replacing)	N/a	175 – 250 years (Mixed)
Poplar	70 -110 years (stand-replacing)	80 years for Aspen/Birch/Fir (Mixed)	70-80 years (Mixed)
Black Spruce bog ***Black Spruce rich swamp***	*100-150 years (Mixed) * 200 years (Mixed)	150 years (Mixed)	N/a

Sources: Heinselman, 1981; Heinselman, 1996; Ontario Parks, 2006; Woods and Day, 1977b

*Notes: *Heinselman (1996) explained that without a basis for estimating fire intervals within Black Spruce bogs and rich swamps, his estimates are best guesses;**1981 estimate*

Although Heinselman's and Woods and Day's estimates provided a general characterisation of the natural mean fire interval for the major forest communities within the region, fire frequency varies geographically and therefore such estimates should not be used as management prescriptions (e.g., timing and location of prescribed burns). For example, a Black Spruce forest community in the southern portion of Quetico can have an entirely different stand-replacing fire frequency than a stand of the same composition in the northern portion of the Park. The large variation in fire frequency by forest community observed by Heinselman (1973) and Woods and Day (1977b) mainly results from the difference in species physiographic characteristics such as fire tolerance/resistance and adaptation as well as the sites they occupy. For example, Black Spruce can escape fire for centuries on bog and rich swamp wetland sites. However, Jack Pine/Black Spruce communities that occupy dry upland sites have the shortest fire return intervals. Jack Pine and Black Spruce also have an adaptation (i.e., closed cones) that allows them to regenerate immediately following stand-replacing fires (Heinselman, 1996).

Heinselman was influenced by Van Wagner's estimate in his 1981 reanalysis of his 1973 data. Van Wagner's methods characterised stand-replacing fire frequency and Hienselman chose to base his 1981 estimates for forest communities that are dependent on these stand-replacing fires (Jack Pine and Black Spruce) to match Van Wagner's 50 year fire cycle estimate. Hienselman also equated his mean fire interval estimates with his fire cycle estimates. However, the mean fire interval only equals the fire cycle in one special case: when all fires are of the same size, and the study area is an integer multiple of this size (for a detailed explanation see Reed 2006). In reality, the fire cycle will always exceed the mean fire interval. Van Wagner's methods assumed that all fires are the same size; however, but Heinselman's did not and therefore the latter's mean fire interval estimate should not have equaled his fire cycle estimate.

As mentioned previously, Hienselman's (1973) natural fire cycle estimate (100 years) is currently used by Quetico's managers as a annual area burned target rather than Woods and Day's (1977b) estimates. That is, the annual area burned target including natural fires, prescribed burns and prescribed natural fires given a 100 year fire cycle is roughly 4,500 ha. This is likely because Woods and Day relied heavily on forest community break-up estimates for their fire frequency estimates, which is a completely different ecological process than fire (i.e., successional and self-organizational change, as well as other kinds of disturbance like wind throw, insect outbreak and disease).

Given the insight gained from preceding literature review, the following chapter explains the methods used during this research.

Chapter 4
Methods

This chapter describes the methods used to characterise Quetico's fire regime. In summary: the 1966 FRI was georectified and digitized as GIS data. Second, the historical time-since-fire distribution and species composition as a function of time-since-fire was inferred. Third, the FRI was updated with the digital fire atlas to produce a current time-since-fire and fire interval dataset. This dataset was used to estimate survival functions, stand-replacing mean fire intervals, the probability of burning and fire frequency spatial and temporal variability using non-parametric (Kaplan-Meier) survival analysis methods. Non-parametric methods were chosen as they are based on the empirical data and therefore avoid curve fitting the data to theoretical parametric fire models (i.e., negative exponential and Weibull). Parametric fire models make potentially problematic ecological assumptions (i.e., probability of burning is independent of time-since-fire) and therefore should only be used after the empirical data has been analyzed. The methods employed during this research could be applied to any area covered by the OMNR FRI and digital fire atlas (or similar data for other provinces or countries) including other protected areas and land managed for timber.

4.1 Spatial Data

4.1.1 Quetico 1966 Forest Resource Inventory

In 1966-1968 the Ontario Department of Lands and Forests produced a FRI of Quetico Provincial Park consisting of 31 maps at 1:15,840 (1 cm = 0.634 km) (McCulloch, 2007) each covering approximately 250 km^2. The FRI was spatially accurate to approximately one chain or 20 m, which is roughly twice the width of the forest stand boundary line on the maps (McCulloch, 2007). In 1966, aerial photography was acquired and the majority of the ground verification field work was also completed in 1966. Ground verification was finalized in 1967 (McCulloch, 2007). The FRI protocol was described in detail in Anon (1973), Anon (1977c) and Dixon and Jenns (1965).

The Park was subdivided into 3 Management Units (MU) for the 1966 FRI. A portion of one MU was licensed to Jim Mathieu Lumber Limited. The other two MUs were part of the Quetico Crown MU (Jean Working Circle and Hunter Island Working Circle). Usually, the company would have inventoried the MU because it was larger than 50 mi^2 (130 km^2) (OMNR, no date). However, the provincial government showed foresight and completed the inventory for the entire Park due to its ecological significance (OMNR, no date). The FRI classified the Park into four general categories:

1) Water (e.g., lakes and streams).
2) Non-forested (e.g., hydro corridor right-of way).
3) Non-productive forest (e.g., treed and open muskeg, bedrock outcroppings).
4) Productive forest (land capable of growing merchantable timber).

The productive forest was further categorized as "protection forest" and "production forest". Protection forest was to be primarily managed for purposes other than commercial forestry (i.e., conservation) and production forest was to be primarily managed for forestry purposes. The FRI usually further subdivided only the production forest, but because Quetico was managed for both forestry and conservation purposes, the production and protection forest were subdivided into:

1) Relatively homogenous forest stands characterised by their species composition, age and other variables.
2) Barren and scattered (land which is less than 25% stocked or has no trees at all due to natural or anthropogenic disturbance).

For each of the 7,493 forest stands, tree species composition (% of total Basal Area to the nearest 10%), age and height of predominant tree (canopy) species, site class, and stocking data are available in tabular form and also appear as map symbology (see Appendix B 1966 FRI Data Dictionary for additional attributes captured during this research). In an Ontario FRI, site class is a relationship between the height and age of a stand where the higher class stands have a higher height/age ratio and thus produce merchantable timber faster (OMNR, 1996; OMNR, 2007c). Stocking refers to the basal area of the stems in a stand versus a presumed optimum (OMNR, 1996; OMNR, 2007)[8].

When comprehensive time-since-fire maps are outside the scope and/or budget of a study, FRI forest stand ages permit the construction of an estimated time-since-fire distribution. In the past, forest age distributions inferred from FRI data were questioned (e.g., Alexander 1980; Johnson et al., 1990; Johnson and Gutsell, 1994) when stand ages were estimated primarily from air photo interpretation. However, the Quetico FRI is the product of both photo interpretation and field sampling. This study's digitizing project identified approximately 1,429 (19%) of the forest stands were "cruised" (field inventoried). Although there were no specific studies on the accuracy of FRI stand age estimates, photo-interpreted stand ages were likely accurate to within 15 to 20 years (W. Day, personal communication, 2007). When photo-interpreting stand ages, age estimates are based on tree height and

[8] Site class and stocking are derived using Plonski's Normal Yield Tables (Plonski, 1981). Formal definitions of these terms are available in Anon (1977).

species composition relationships using Plonski's Normal Yield Tables (OMNR, 1996; OMNR; 2007; Plonski, 1981). Therefore, the 1966 age estimates for Quetico were highly dependent on the interpreter's experience and their knowledge of the area (W. Day, personal communication, 2007).

Despite potential inaccuracies and limitations, the 1966 FRI represents the only comprehensive dataset for Quetico suited to the analysis of stand-replacing fire frequency. The same situation applies throughout many of Ontario's commercial timberlands. The completion of a detailed and accurate time-since-fire map of Quetico based on extensive field work (and the Quetico Foundation data collected in recent years) was beyond the scope and budget of this research. An updated FRI will be completed in 2009 and will prove valuable in subsequent studies of fire and vegetation dynamics in Quetico. Like its predecessor, the new FRI will provide time-since-fire estimates based on high resolution (20 and 40 cm) digital air photo interpretation and field inventory (OMNR, 2008d). Reconnaissance field inventory and research on the ecology of the Park will be required in the near future to properly implement the Park's fire and vegetation management plans, thus creating the opportunity to conduct field-checks of the 1966 and forthcoming (2009) FRI datasets. The time-since-fire map produced in this research will be an important resource for the sampling design of more detailed studies including: 1) collecting tree-core and fire scar data to supplement the Quetico Foundation Student Summer Program inventory for validating stand age and fire years respectively; and 2) collecting empirical data on surface fire regimes as well as old growth and gap disturbance dynamics.

4.1.2 Quetico Digital Fire Atlas

Ontario forest fire history 1:500,000 maps were originally compiled in 1978 by Donnelly and Harrington. Although formal fire reporting began in Ontario in 1917, it was not until 1921 that maps of the fires were included. The mapping includes all fires south of 52^{o} North, known as "The Area of Undertaking" (all land and water within forest management unit boundary lines) (OMNR, 2008c), greater than or equal to 200 ha during 1921-1976. Donnelly and Harrington (1978) estimated that 5% of the total area burned is not included in this mapping because of missing fire reports and/or fire boundary mapping, and non-reporting (especially in northern Ontario and during the Second World War years). Additionally, fires less than 200 ha were said to comprise 5% of the total area burned. Given these limitations, Donnelly and Harrington's (1978) fire history mapping accounted for approximately 90% of the total area burned for this area of the province during the period of record. In 1995, the Forest Landscape Ecology Program of the OMNR updated the Donnelly and Harrington atlas as a GIS database (Perera et al. 1998) using data from aerial photography and post-fire Global

Positioning System (GPS) flyovers (1976-1995) and classified (supervised) satellite imagery for the period 1973-1991 (digital, transparency and hardcopy) to improve the spatial accuracy of the fire mapping. The final resolution of the mapping was 1 ha pixel size. The OMNR continues to update this dataset via remote sensing and GIS technology. The digital fire atlas used in this research covers 1921-2007. A limitation of the digital fire atlas was that it maps the perimeter of only large burns and does not include small fires (<200 ha) and areas within the fire perimeter that were not burned. For example, according to Song (2002), in 1995 approximately one-third of the area within Fire #141 in Quetico was unburned.

4.1.3 Ontario Base Mapping Lakes

The OMNR Natural Resource Values Information System (NRVIS) lakes GIS (vector) 1:20,000 database meets Ontario Base Mapping (OBM) standards for Northern Ontario. The lakes GIS layer was used as the control layer to provide known coordinates in the georectification of the 1966 FRI.

4.2 1966 OMNR Forest Resource Inventory: Historical Maps to GIS Database

In September 2007, funding obtained from the Forest Policy Section of the OMNR/Ontario Parks was used to georecctify and digitize the 1966 FRI of Quetico.

4.2.1 Conversion to Raster Imagery and Georectification

The original 1966 FRI was mapped on Mylar. Thirty-one .pdf format 1966 FRI map sheets of Quetico were provided by Park staff. The nature of this mapping medium (i.e., stretchy) and the scanning process introduced distortion leading to unknown inaccuracies. Adobe Photoshop was used to convert the .pdfs to high-resolution raster imagery (.jpeg format at 1,200 DPI) compatible with ArcGIS 9.2 software. Jpegs were created instead of .tiffs to avoid file size/data storage issues. The imagery was then converted to ESRI GRIDs using ArcMap for the georectification process. The GRIDs were projected to Universal Transverse Mercator North American Datum 83 (UTM NAD 83) Zone 15.

The UTM NAD 83 NRVIS lakes layer was used as the control layer for the image-to-map georectification process. First, Ground Control Points (GCPs) were identified on the FRI GRID and matched to the corresponding points on the control layer. The number of GCPs needed to achieve an acceptable Root Mean Square (RMS) error and positional accuracy varied depending on the extent of the features on the map sheet. For example, when map sheets were completely covered with features (e.g., forest stands) approximately 50-75 GCPs were required, whereas as few as 5-10 GCPs were required for maps sheets with sparse features. The GCPs also needed to be distributed evenly throughout the map sheet. RMS error refers to a computation that measures the geometric distortion between the control layer and the GCP values (Jensen, 2005). Roll, pitch, and yaw errors often found in imagery obtained from aircraft platforms introduce unsystematic distortion and are corrected by using high order polynomial transformations (Buiten and Van Putten, 1997; Jensen, 2005). The 1966 FRI was based on air photo interpretation, and a first order affine polynomial transformation failed in all cases to produce an acceptable RMS error. Subsequently 2nd and 3rd order polynomial transformations provided the most acceptable RMS error, while preserving the closeness (positional accuracy) of the FRI GRID coordinate values to the control layer coordinate values. According to Jensen (2005), “higher order polynomials often produce a more accurate fit for areas immediately surrounding GCPs” (p.240). However, Gibson and Power (2000) suggested that “other geometric errors may be introduced at large distances from GCPs” (p. 240). Therefore, both RMS error and positional accuracy were considered with equal importance in the georectification of the FRI.

The positional accuracy assessment ensured that the FRI was no more than 30-50 m away from the control layer in any location. When the FRI was more than 30 m away, the inaccuracy was never more than 50 m and was usually localized. A random sample of 30 coordinate points for each of the 31 FRI maps confirmed that on average the inaccuracy was within 30 m. The final GIS database may appear to be out more than 30 m in some locations as wetlands (open and treed muskeg), alder patches and bedrock outcroppings were not digitized. Because the analysis was performed at the landscape level, 30-50 m accuracy was considered acceptable.

No OMNR standards exist for georectification accuracy; therefore *U.S. National Map Accuracy Standards* were followed for desirable RMS error. The U.S. standards state that for map scales of:

- 1:10,000 RMS error must be < 7.8 units (before transformation); and
- 1:20,000 RMS error must be < 4.0 units (before transformation).

A RMS error of below 5 units (before transformation) and below 1 unit (after transformation) was consistently achieved for the 1:15, 840 FRI.

4.2.2 Digitizing and Quality Control

The ArcScan extension was tested for 1 FRI map as it automates the digitizing process. However, manual digitizing was chosen because cleaning unwanted vectors (e.g., map symbology) after the automated process took more time than manually digitizing. This is due to the level of detail on the FRI. The FRI forest stands and associated attributes (see Appendix B 1966 FRI Data Dictionary) were digitized using ArcMap (ArcGIS 9.3). Due to time and budget constraints forest stands (productive forest) were the only features digitized. Digitizing wetlands (treed and open muskeg), alder patches, lakes and streams was outside the scope of this research. In some instances bedrock outcroppings could have been the result of severe fires that removed all vegetation and soil. Bedrock outcroppings were not digitized. Therefore in some cases the area burned could have been underestimated. The minimum mapping unit (smallest forest stand digitized) was 0.0299 ha (299 m^2). The attributes were captured from map symbology on the FRI map sheets.

A Personal Geodatabase was created to store the initial geometry (line work) of the forest stands (feature class polylines) and attributes (feature class points). For every forest stand digitized, a point was placed within its boundaries. The attributes of the forest stands were captured in the attribute tables of these points. Once the line work was complete for a map sheet, the “feature to polygon” tool was used to create forest stand polygons. The attributes of the point features were transferred to the final polygon during this process. The “feature to polygon tool” was also used as a quality control step because forest stands that were not complete (i.e., the polygon had “dangles” or “gaps”), would not successfully be converted to polygons.

In the final GIS database, the 31 FRI maps were merged into one polygon layer. When two adjacent map sheets were completed (digitized and checked for quality) they were merged together using the Arc Map editor toolbar. Where the geometry of a stand spanned two or more map sheets, the end nodes of the line work from each map sheet would not usually match up perfectly because the map sheets were georectified individually. However, because RMS error and positional accuracy were considered during the earlier process of georectification, the maximum discrepancy of the end nodes was approximately 50 m. When the end nodes did not match, the end node was moved 25 m south on one end node and 25 m north on the other end node before the line work was merged. This was applied along all four map sheet edges as necessary.

In some cases, a single stand could lie on two to four separate maps. To preserve the original map and forest stand identifier a relational table was created (Table 4.1). Once all maps were complete, this relational table was added as attributes to the final GIS database via an attribute table relation and then exported to shapefile. To create the relational table, each of the forest stands was given a new number, in the “StandNumF” field. The FRI stand numbers were then recorded in the relational table, along with the appropriate MapID number. In the example provided in Table 4.1, StandNum 3 on map 484904 has been merged with StandNum 67 from map 485904. This stand was given a new number (in this case, 1002).

Table 4.1 Example of the relational table used to preserve FRI map and forest stand numbers.

StandNumF	MapID1	StandNum1	MapID2	StandNum2
1002	484904	3	485904	67

Various techniques were used to ensure the quality of the final GIS database. Each field in the attribute table was sorted in ascending and descending order to assist in detecting recording errors. The geometry and attributes of every forest stand were checked by the individual who digitized them and by a colleague during the quality control process. To ensure that each forest stand was captured, unique colours were assigned to every forest stand polygon on a map sheet and the transparency was adjusted to 45%, so that the FRI was visible beneath. Programming (Python) was also used to identify errors in the species composition (SPCOMP) field string due to coding errors.

4.3 Historical Forest Analysis

4.3.1 Historical Forest Age Distribution

The 1966 FRI GIS database of Quetico allowed for the analysis of the historical forest age distribution for the entire Park. The age distribution was grouped into bi-decadal (twenty year) age classes to account for the lack of precision in estimating stand ages. The forest age distribution provides an estimate of the time-since-fire distribution from which fire frequency can be statistically estimated as the forest stand ages are equated to time-since-fire. Consequently, the time-since-fire distribution does not have the resolution required to detect small and/or low-severity fires that do not initiate succession. The creation of the time-since-fire and fire interval dataset, using the FRI and digital fire atlas is discussed in section 4.4.3.

4.3.2 Historical Forest Composition as a Function of Time-since-fire

Using Gauthier's (2001) methods, vegetation dynamics as a function of time-since-fire were assessed by exploratory analysis of the forest composition (by OMNR working group (WG)) and time-since-fire layers of the 1966 FRI GIS database. Specifically, each of Quetico's WGs in 1966 was explored in terms of their relative area occupied in each decade of origin (time-since-fire). It was hypothesized that the relative area occupied by fire-dependant species, such as Black Spruce and Jack Pine, was a function of time-since stand-replacing fire.

4.4 Fire Frequency Estimation

4.4.1 Estimating a Historical Time-since-fire Distribution

Once the 1966 forest age distribution map (an estimated time-since-fire map) was complete, the cumulative percent of area unburned (i.e., the proportion of the Park surviving) for each stand (time-since-fire) age-class could be plotted to produce a historical time-since-fire distribution for the Park. Once plotted, changes in fire frequency over time could be assessed visually, graphically or statistically (Johnson and Van Wagner, 1985; Johnson and Gutsell, 1994; Reed, 1998). Fire cycle (the time expected to burn an area equal in size to the Park) estimates could also be estimated from the time-since-fire distribution. However, to do so the distribution must be fitted to a negative exponential distribution. Current methods (Reed, 1998) are also required to obtain precise fire frequency estimates (i.e., 95% Confidence Intervals) and compare between time periods. Therefore, producing historical fire cycle estimates for different time periods was considered both inappropriate (because this study's fire frequency data do not fit the negative exponential distribution) and outside the scope of this research (see Discussion and Conclusions section 6.1).

4.4.2 Estimating a Current Fire Cycle

The 2007 *Life Science Inventory* (Solomon, 2007) computed the fire cycle applying methods (i.e., burning rate) similar to Heinselman's (1973) utilizing data from Quetico's digital fire atlas. The digital fire atlas included all recorded fires, including areas that may have burned more than once for 1921-2007. Solomon (2007) estimated a fire cycle of 300 years. However, the area burned that is actually in lakes in the digital fire atlas was not accounted for in the computation. Since the lakes comprise 98,803 ha (or 21%) of the Park's total area, Solomon's (2007) computation underestimated the fire cycle. Therefore, for this research Heinsleman's (1973) methods were used to perform an updated computation by subtracting the lake area from the area burned. First, the digital fire atlas was clipped to the Park boundary layer as some of the fires included in the digital fire atlas were partially located outside the Park's limits. The Park's total land area was also calculated by subtracting the area of the lakes from the Park's total area. The burning rate was calculated by dividing the number of years in the reference period by the total area burned over the reference period, divided by the study area size (Heinselman 1973) (see Equation 1), where A is the size of the Park's land area (373,451 ha), B is the total area burned (94,906 ha) during T years, the amount of years covered by the study (87 years).

4.4.3 Time-since-fire and Fire Interval Dataset

A 2007 time-since-fire map was created by adding 41 years to the stand age attributes of the 1966 FRI (Figure 4.1). The forest stands (or portions of stands) that have burned since 1966 were updated with the digital fire atlas for 1967-2007 (Figure 4.2). The digital fire atlas recorded fires from 1921 to 2007. The 1966 FRI provided the time-since-fire from 1668 (the oldest stand encountered) to 1966 (where a fire occurred during the year of mapping). The two datasets were combined the using the FRI for 1668-1966 and the digital fire atlas for 1967-2007. This was considered acceptable as the FRI was assumed to be more indicative of stand-replacing fires.[9] The prescribed burn data provided by the Park were not included because these fires were human-caused. It was unusual that the primary time-since-fire dataset was historical. However, a current FRI is not available until 2009 and the 1966 FRI increased the temporal extent of the study.

The 1966 FRI time-since-fire attributes were updated by clipping the FRI to the digital fire atlas. The digital fire atlas time-since-fire attributes were assigned to the FRI by performing a spatial join. According to the digital fire atlas, there were seven forest stands in the FRI that burned three times since 1966. These seven stands were manually updated. *Second* and *third* area burned (ha), fire year, and time-since-fire attribute fields were thus updated to the FRI. To explain, the 1966 FRI forest stand area was equated with the *first* area burned. The FRI year of origin was equated with the fire year and the FRI stand age was equated with the time-since-fire. The *second* and *third* area burned, fire year and time-since-fire attributes were provided by the digital fire atlas. As discussed earlier, when mapping the time-since-fire dates provided by the 1966 FRI, 20 year age-classes were used to smooth the effects of approximate stand dates. However, grouping of this kind also accounted for irregularities in the fire frequency data caused by large fires (e.g., the 1936 fires).

Once updated, a current time-since-fire map was produced for the entire Park and fire intervals were available for 615 FRI forest stands, seven of which also fully burned or partially burned a third time according to the digital fire atlas.

[9] The FRI accounted for a burn only if the stand was killed and succession was initiated. Whereas the digital fire atlas would have included multiple fire severities and did not account for unburned areas within the fire perimeter.

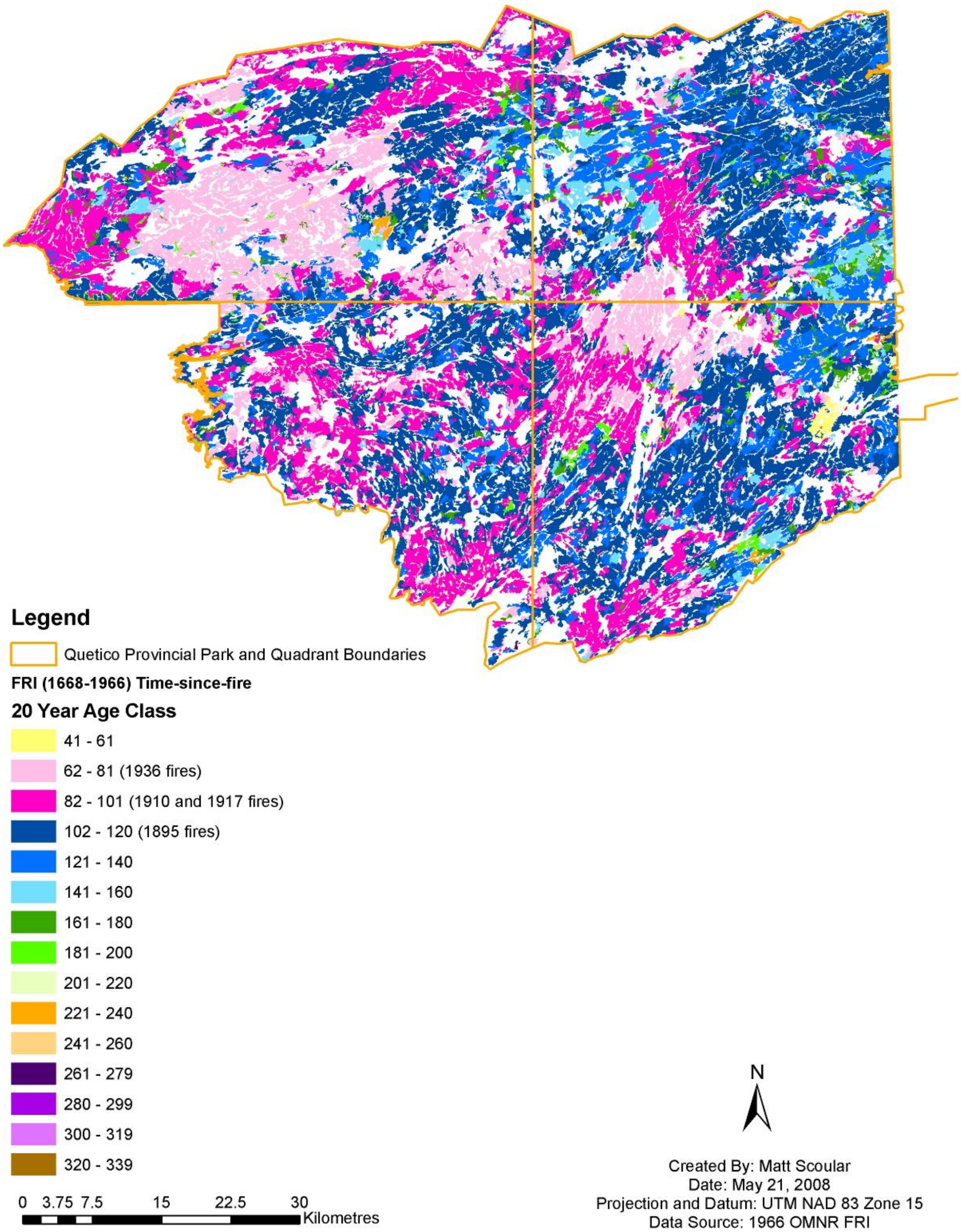

Figure 4.1 1966 FRI time-since-fire (updated to 2007).

Note: 20 year age classes are time-since-fire estimates in 20 year groupings which correspond to the 1936 fires, etc.

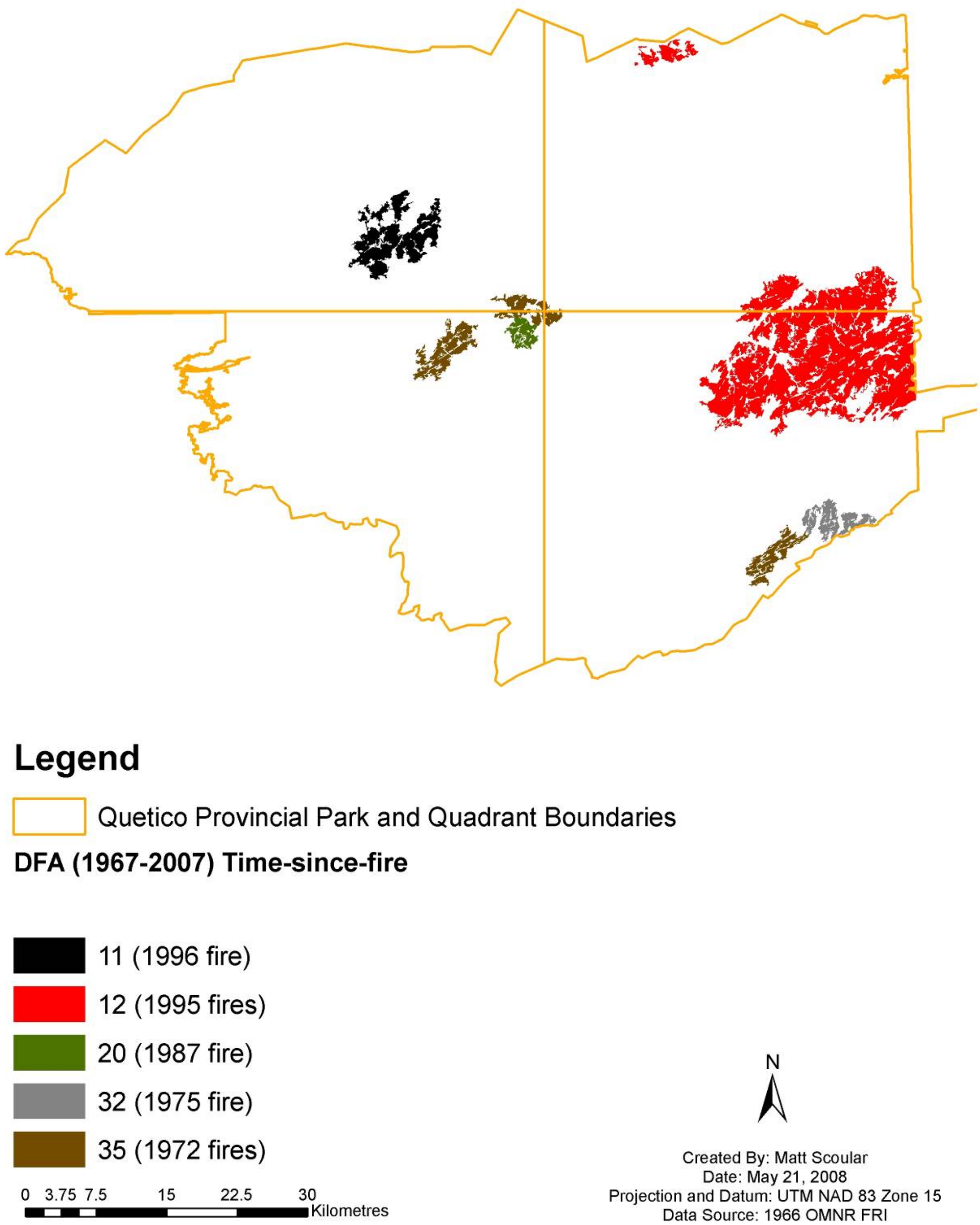

Figure 4.2 Digital fire atlas time-since-fire (1967-2007).

Note: 11, 12, 20, 32, 35 is the time-since-fire in years which correspond to the 1996, 1995, 1987, 1975 and 1972 fires

4.5 Survival Analysis

4.5.1 Estimating the Survival Function

The survival function was estimated by non-parametric Kaplan-Meier survival analysis using the statistical software package R (R, 2007). The FRI forest stands that had not burned since 1966 according to the digital fire atlas provided a minimum estimate on the fire interval (lower bound) and were treated as right-censored observations in further statistical analyses (D. Matthews, personal communication, 2008). Without at least one complete observation, data that provide a fire interval for any given stand, estimation of the survival function and mean fire interval through non-parametric survival analysis is impossible (D. Matthews, personal communication, 2008). However, once the FRI was updated with current fire data documented in the digital fire atlas for the period 1967-2007, fire intervals for six hundred and fifteen forest stands were available. Subsequently, the estimated survival function and mean fire interval estimates based on survival analysis were obtainable. 91.8% of the dataset was classified as right-censored observations on the age at which stand-replacing fires occur.

The survival function was estimated non-parametrically as comparative simulated samples (Figures 4.3 and 4.4). This confirmed that the empirical survival function was not a good fit to either the Weibull or negative exponential theoretical models. The same amount of censoring (91.8%) was used to create the simulated negative exponential and Weibull model comparative samples because the level of censoring could drastically distort the survival curves (D. Matthews, personal communication, 2008).

It is possible to group the data into five, 10 or even 20 year age classes. Recent studies justified this practice because smoothing the data in this manner can account for inaccuracies in dating stand ages and therefore in the estimated survival function (e.g., Drever et al., 2006; Grenier, 2005; Lauzon et al., 2007). The estimated survival function produced with the empirical data was compared to the estimate based on grouping the data into five-year and ten-year age classes. The resulting estimates were not different from those obtained from raw data (see Appendix C). Therefore the raw data were not grouped into age classes in subsequent analyses.

As the survival analysis in this research was not concerned with area burned, any potential temporal inaccuracy (and therefore spatial accuracy of area burned per year) of the FRI data were considered acceptable. Moreover, the temporal resolution of the digital fire atlas (1 year) could have been compromised if the data were grouped in age classes.

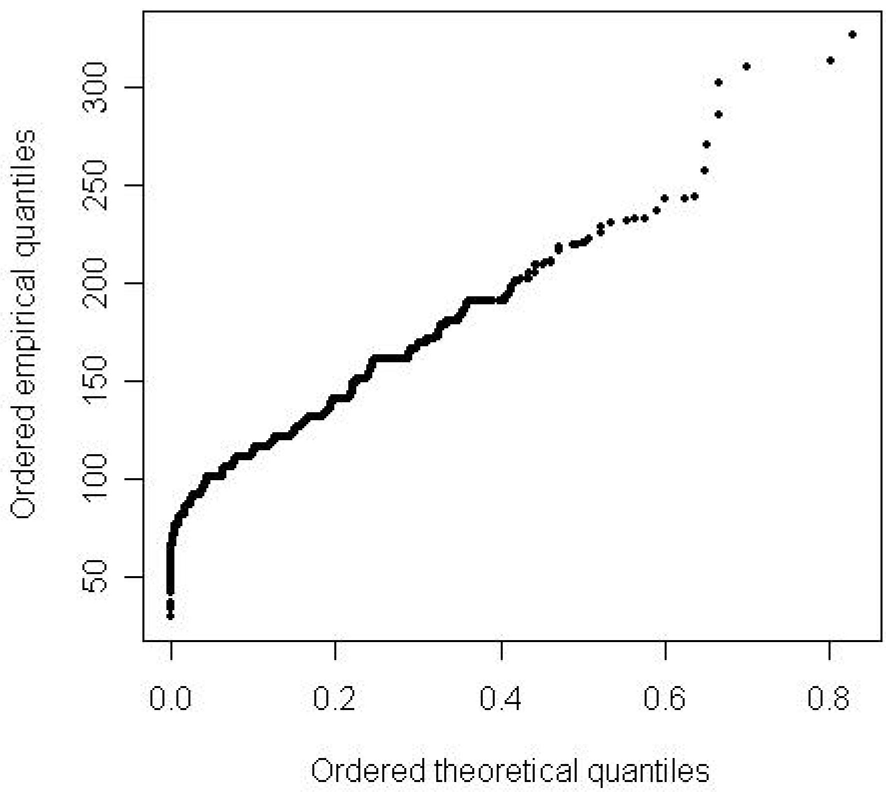

Figure 4.3 Negative exponential model vs. global empirical distribution.

Note: The negative exponential is not a good fit to the data because the plot deviates sharply from a linear pattern

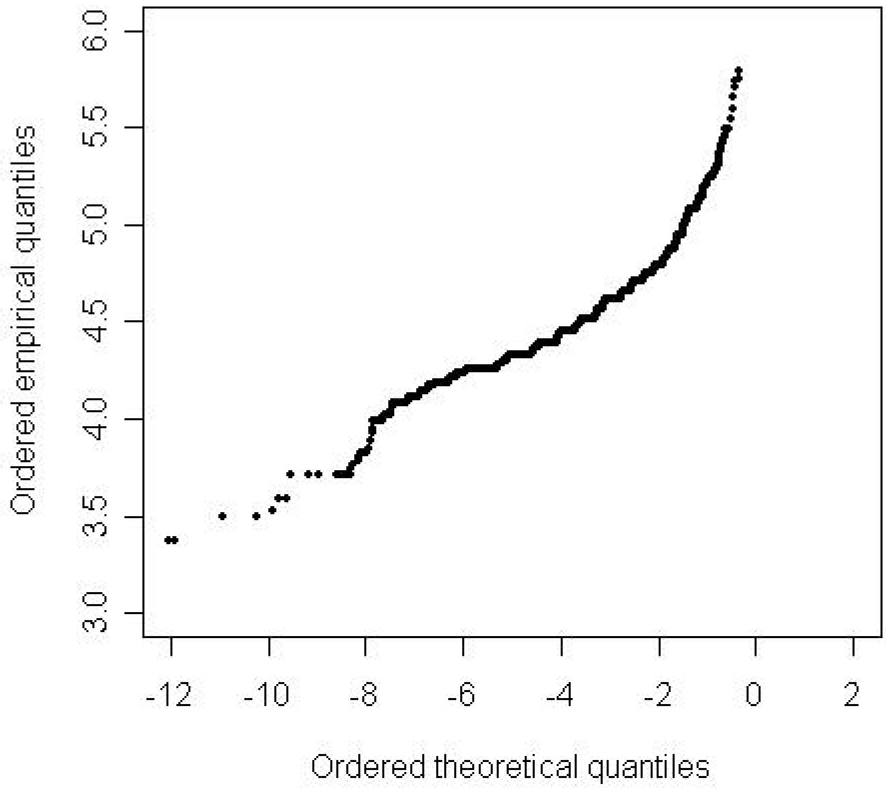

Figure 4.4 Weibull model vs. global empirical distribution.

Note: The Weibull is not a good fit to the data because the plot deviates sharply from a linear pattern

If either the negative exponential or the Weibull theoretical models were good fits to the observed data, the plots in Figures 4.3 and 4.4 would be roughly linear but they are not. Considerable curvature is evident for both of these parametric models compared to the empirical data for Quetico, so neither of the two models are a good fit. The negative exponential model may be suitable for stands from 100-250 years old; however, the probability of burning does appear to increase with increasing time-since-fire (see Discussion and Conclusions section 6.5).

4.5.2 Data Partitioning

4.5.2.1 Spatial partitioning

According to Hills (1959), Rowe (1972) and the OMNR, Quetico lies within a single ecoregion at the landscape level. Therefore, due to a lack of available ecological data and time constraints, spatial partitioning based on ecological factors that influence fire frequency (e.g., distance to fire break, surficial deposit, etc.) other than topography (upland vs. lowland) were not considered in this research.

Since fire frequency could significantly vary from drier upland sites to wetter lowland sites, the data were partitioned into upland and lowland forest stands. The topography throughout the Park varies by only 164 m so this broad classification was acceptable. The partition was performed by separating the WGs (predominant species in the species composition string) that are known to occur on lowland sites. Therefore, all forest stands where either Tamarack or Black Ash was predominant were considered lowland sites. Black Spruce occurs on both upland and lowland sites. Therefore, site class and forest unit (FU) also had to be considered in the GIS selection query. Forest stands of site class 1 and 2 occur primarily on upland sites and those with site classes of 3 and 4 occupy the lowland or organic sites (W. Day, personal communication, 2007). However, site class 1 and 2 can occur on lowland sites when the site is nutrient rich with evident water flow and Tamarack and White Cedar are present (W. Day, personal communication, 2007). Therefore, all Black Spruce stands of site class 3 or 4 and that fell within the “Black Spruce Lowland” FU (Tamarack and White Cedar present) were considered lowland sites. White Cedar is also present on both upland and lowland sites. Therefore, when the stands Forest Unit (FU) was “Upland Cedar” it was considered upland, otherwise it was classified as a lowland site. As a result of this classification, 426 (6%) forest stands were categorized as lowland. However, as stated earlier, the treed and open muskeg and alder patches were not digitized; therefore some of the lowland sites in the Park were not captured in the dataset.

The data were partitioned into four quadrants of the Park (northwest, northeast, southeast, southwest) to characterise fire frequency spatial variability and potentially assess the influence of land use on fire frequency (see Figure 5.2). For example, the majority of industrial logging occurred within the northeastern quadrant of the Park.

A log-rank test (R, 2007) was used to identify significant statistical differences in the survival function among the four quadrants, based on the observed fire interval data. Log-rank tests are currently the technique of choice for comparing two or more survival curves derived from non-parametric Kaplan-Meier estimation. The two methods are based on the same assumptions: 1) treat right-censored and complete observations in the same way, 2) survival probabilities are the same for observations in early and late time periods and 3) fire intervals occur at the observed stand age (Bland and Altman, 2007; Matthews and Farewell, 2007). According to Bland and Altman (2007), the log-rank test is used to test the null hypothesis that there is no difference between the populations (here the four quadrants of the Park) with respect to the probability of an event (here a fire interval) at any stand age (time-since-fire). For each stand age at which an event occurs, the observed numbers of fire intervals, at that stand age, in each quadrant and the corresponding numbers expected if there truly were no difference among the quadrants are calculated using a contingency table. A 2 x 4 contingency table was be used to compare the four quadrants of the Park and a 2 x 2 contingency table was be used to compare just the two northern quadrants.

For example, if a total of 253 stands have survived to an age of at least 100 years and 5 of these stands burn at an age of 100 years in all four quadrants, the number of burned stands is divided by the number of stands that survived to age 100 (i.e., 5/253). This number is then multiplied by the corresponding number of stands surviving to age 100 or more in each of the four quadrants to obtain the expected numbers of fire intervals at age 100 in each quadrant. This calculation is performed every time a fire interval occurs. Then observed and expected numbers, by quadrant, are aggregated across all stand ages. The resulting quadrant-specific differences between the observed and expected event totals are compared across the four quadrants to obtain the observed value of the log-rank statistic. This observed value is then calibrated against an appropriate sampling distribution (i.e., a chi-squared distribution with three degrees of freedom if four quadrants are being compared simultaneously) to obtain the significance level (p-value) of the log-rank test for the given data.

The last spatial partition was based on the two fire management zones within the Park, the measured zone and the prescribed fire zone (refer to Figure 3.1). This partition was performed as fire management is considerably different within the two zones. For example, given certain criteria (e.g., fire weather), fires are allowed to burn freely within the prescribed fire zone, whereas fire is primarily reintroduced through smaller prescribed burns in the measured zone (refer to section 3.1.4 for a more detailed description of difference between these two zones). Therefore a current mean fire interval estimate was needed for both fire management zones.

4.5.2.2 Temporal partitioning

To test the generally accepted hypothesis that fire frequency has increased in northwestern Ontario during the most recent time period (1970-2007), the data were partitioned to exclude any stand ages observed prior to 1970 before estimating the global survival function and mean fire interval. When temporally partitioning fire frequency data, at least one fire interval (complete observation) must end within the time period of interest (D. Matthews, personal communication, 2008; Polakow and Dunne, 1999). Since the first fire interval ended in 1972, the most recent time period (1970-2007) was the only temporal partition suitable for survival analysis. Polakow and Dunne (1999) explained that previous studies that have fitted fire frequency data to a Weibull theoretical model appear to have failed to take this requirement into account and therefore have produced erroneous results. Likewise, recent studies that have used non-parametric methods (e.g., Cyr et al., 2007; Lauzon et al., 2007) have also made this mistake. This detail may have been overlooked as most survival analysis has been undertaken to estimate the fire cycle as opposed to the mean fire interval.

To compare the mean fire interval estimates obtained in this research with forthcoming OMNR research using BFOLDS, the data were partitioned from 1963-2003. BFOLDS runs on Fire Weather Index data for this time period.

To assess the influence of Fire #141 in 1995 on the mean fire interval estimates it was removed from the dataset. Fire #141 provided 75% of the fire intervals (complete data) and therefore influenced the survival analysis results the most. However, from an ecological perspective although it was a large fire, it only affected 5% of the Park. Therefore, from a fire management viewpoint, it was important to assess the Park's fire frequency with and without this fire.

To assess whether the probability of burning was constant for all stand ages (time-since-fire observations) the Kaplan-Meier estimator and scatter plot smoothing were used to estimate the probability of a stand-replacing fire for the period 1668-2007. Essentially, scatter plot smoothing is an averaging of the probability of burning between local events (fire intervals that are near to one another in stand age) weighted by the number of events at each stand age (Cleveland et al., 1992; D. Matthews, personal communication, 2008).

Confidence intervals were obtainable as the statistical package R provides standard errors for the mean fire interval estimates derived from the survival function. A standard error is multiplied by two and then added to/subtracted from the mean fire interval estimate to obtain an approximate 95% confidence interval. The various mean fire intervals and their confidence intervals are summarized in Table 5.3.

Chapter 5
Results

This chapter presents the results of the historical forest analysis based on the FRI (i.e., the forest age distribution, the forest composition as a function of time-since-fire and the time-since-fire distribution), the current fire cycle estimate inferred from the digital fire atlas, the time-since-fire map based on the FRI and the digital fire atlas and the survival analysis. The survival analysis includes the characterisation of fire frequency spatial and temporal variability and the probability of burning.

5.1 Historical Forest Analysis

5.1.1 Historical Forest Age Distribution

According to the 1966 FRI, the mean (area weighted) stand age (± Standard Deviation) in Quetico was 63 ± 25 years. Approximately 16% of the forest was older than 100 years, with the oldest stand being 298 years. The 1966 forest age distribution (proportion of forested area per bi-decade of origin) is presented in Figure 5.1.

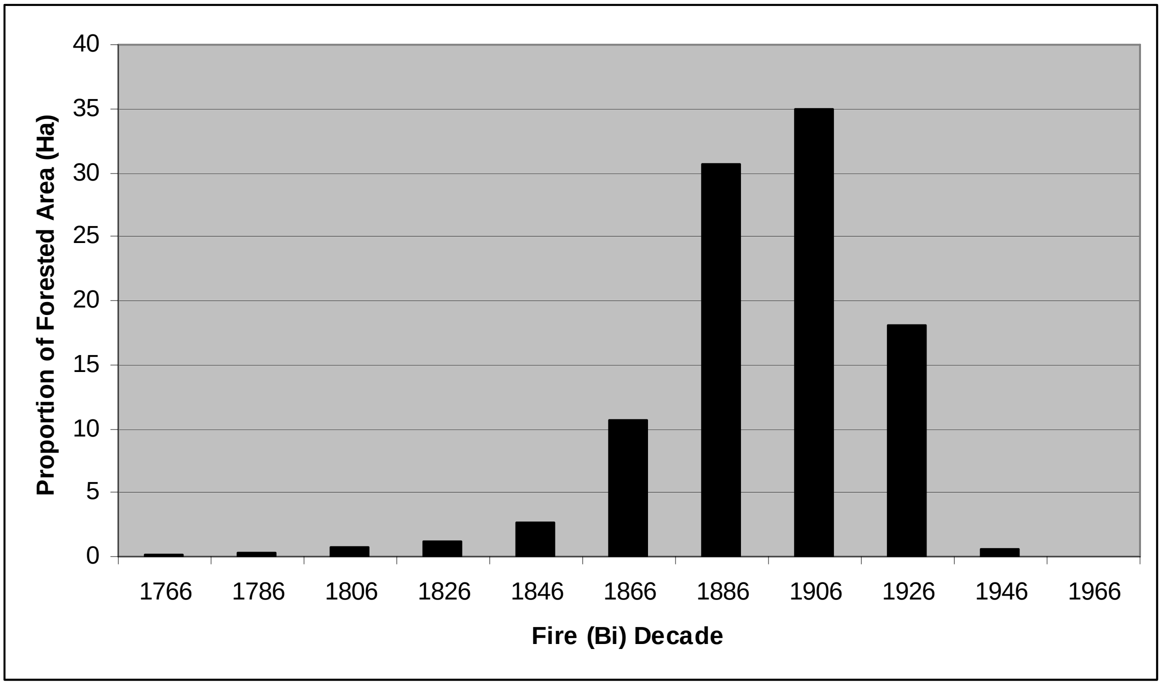

Figure 5.1 Historical forest age distribution.

Notes: Stand age was assumed to be equivalent to time-since-fire; therefore year of origin was equated with the fire year. The proportion of forest area burned prior to 1766 in each (bi) decade (20 year period) was less than 1% in 1966 and therefore was not plotted. Likewise, less than 1% of the forest area burned during the year of mapping (1966) and therefore does not appear on the graph. All stands are assumed to have originated from fire; however, in reality insects, wind throw and logging could have initiated stand recruitment.

Quetico's 1966 age distribution was typical for non-logged (or lightly-logged) areas of the Ontario boreal forest. According to the available historical records, roughly 3% of Quetico Park was logged (refer to Study Area section 2.8) (Peruniak, 1990; Solomon, 2007). There were no large stand-replacing fires from 1946-1966 and this was reflected in the 1966 age distribution and validated by the Park's digital fire atlas. It is evident that the fires of the late 1800s and early 1900s influenced Quetico's age distribution the most as stand recruitment increased more than three-fold from 1866 to 1906. For example, the 1917 fire that burned approximately 6,070 ha within Timber Berths 10[10], 41[11] and 38[12] (Ontario Parks, no date) (refer to Figure 2.2 for the location of the timber berths) corresponds to the 41-61 age class of the historical forest age map (Figure 5.2). Likewise, the 1891 fire in the northeast portion of the Park (M. Gluck, personal communication, 2008)[13] corresponds to the 61-80 age class. The forest age map also displays the spatial pattern including size, shape and configuration of the 1966 forest stand mosaic.

[10] Timber Beth 10 was totally burned
[11] Timber Berth 41 was partially cut
[12] Timber Berth 38 was to be cut the following winter
[13] Fire year 1891 was validated by a modelled fire return interval using BFOLDS (M. Gluck, personal communication, 2008). However, the Parks oral history fire records state that there was also a large fire in 1895 in the southern portion of the Park. This is likely the same fire that Heinselman (1996) refers to that burned in both Quetico and the BWCAW in 1894.

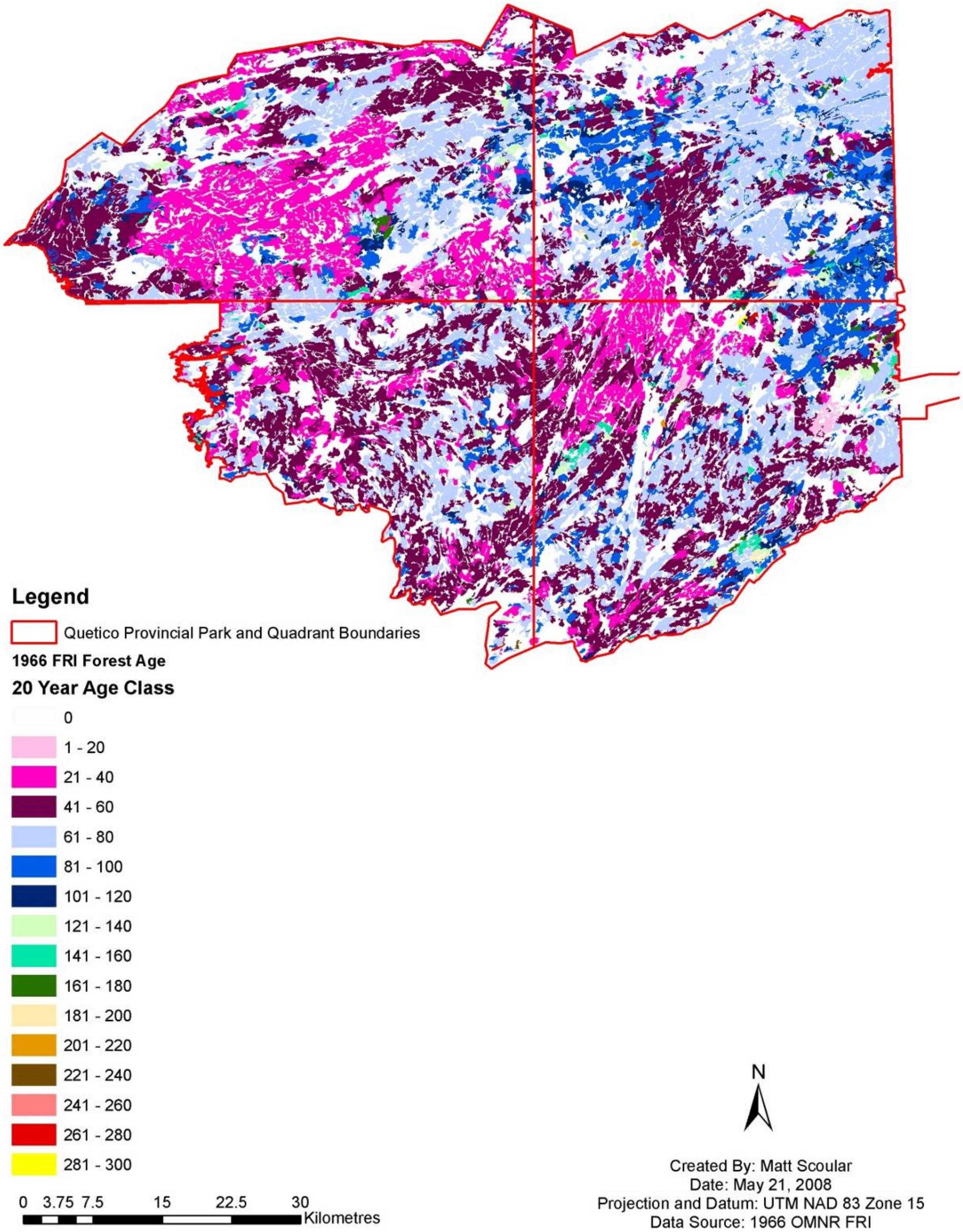

Figure 5.2 Historical forest age map.

It is evident when interpreting the 1966 forest age distribution map that the majority of Quetico's forest communities originated as a result of the large fires of 1891 (M. Gluck, personal communication, 2008), 1895 (and/or 1894), 1910, 1917 (Ontario Parks, no date) and 1936.

5.1.2 Historical Forest Composition as a Function of Time-since-fire

Figure 5.3 and 5.4 show the 1966 distribution of Quetico's stand-replacing fire-dependant working groups (WG), Black Spruce (upland) and Jack Pine, in terms of their relative area occupied in each decade of origin. Figure 5.3 confirms that the majority of Black Spruce dominated stands originated as a result of the large fires of 1891(M. Gluck, personal communication, 2008), 1895 (and/or 1894), 1910, 1917 (OMNR, no date) and 1936. The lack of large stand-replacing fires from 1945-1966 explains the lack of upland Black Spruce stands originating during this time period.

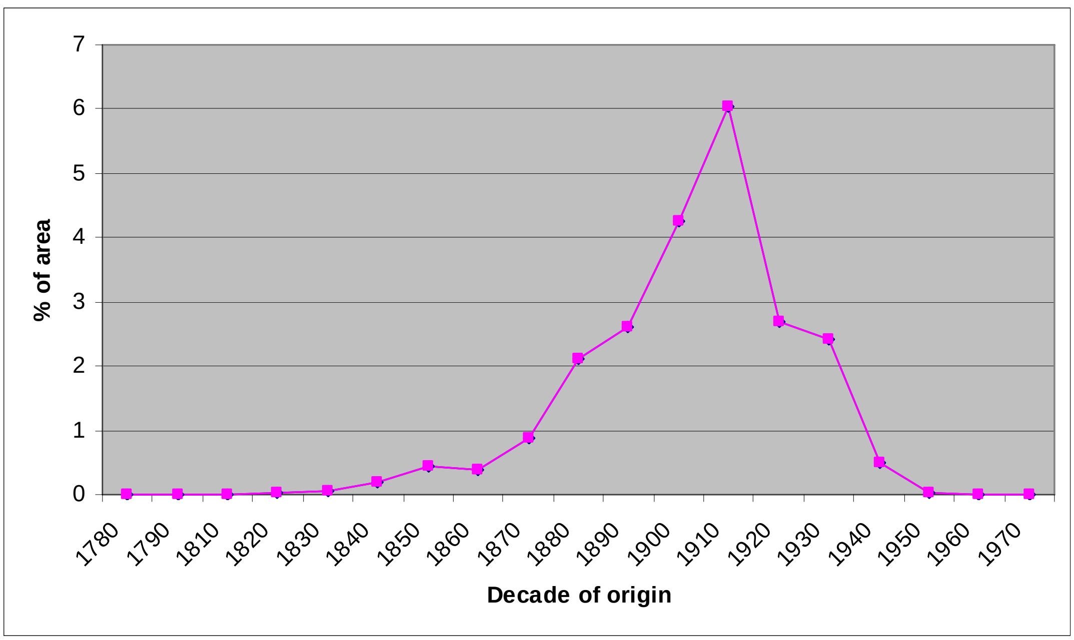

Figure 5.3 Relative area occupied by the Black Spruce (upland) Working Group as a function of time-since-fire.

Figure 5.4 confirms that the majority of Jack Pine dominated stands originated as a result of the large fires of 1891 (M. Gluck, personal communication, 2008), 1895 (and/or 1894), 1910, 1917 (OMNR, no date) and 1936. The lack of large stand replacing fires from 1945-1966 explains the lack of Jack Pine stands originating during this time period.

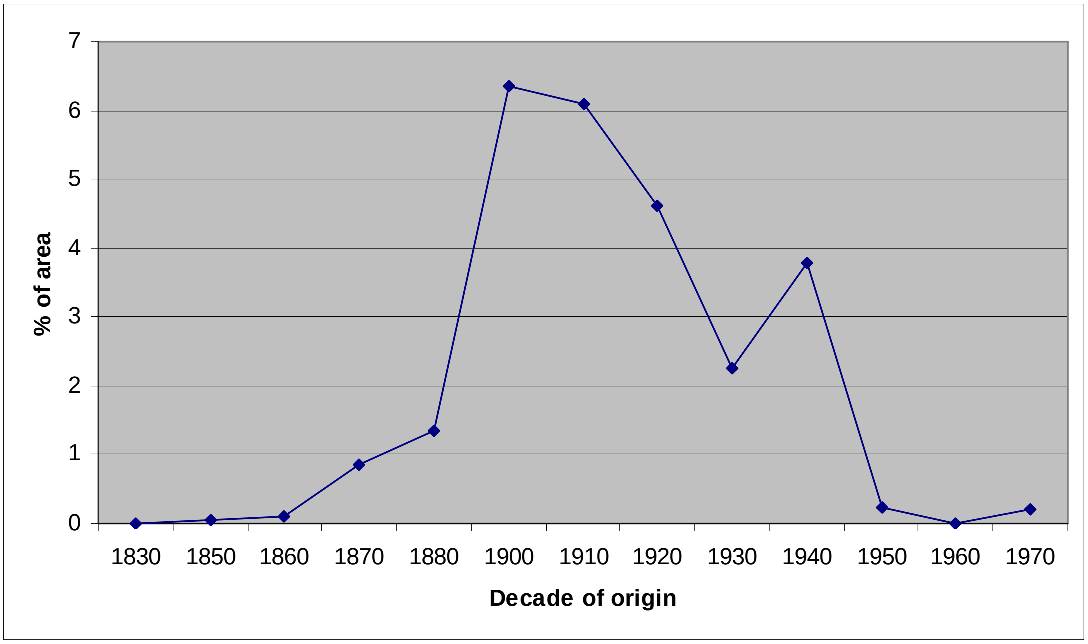

Figure 5.4 Relative area occupied by the Jack Pine Working Group as a function of time-since-fire.

As expected, the fire dependent species WGs (Black Spruce and Jack Pine) recruitment was most abundant in years following large stand-replacing fires. However, the longevity of these tree stands and fire history must also be considered when interpreting these graphs. For example, the lack of abundance in the earliest decades may be the result of a lack of surviving stands that originated in this time period. This can be primarily attributed to over burning but perhaps also to mortality other than fire (wind throw, insect outbreak, and disease) or stand succession to a different WG. Appendix D includes graphs for the rest of the WGs found in Quetico in 1966.

5.1.3 Historical Time-since-fire Distribution

Using Van Wagner's (1978) method of the negative exponential distribution, the inverse of the 1966 mean (area weighted) stand age (63 years) would give a probability of a stand burning in any given year as 0.0158. The equivalent is that roughly 1.58% of the Park or 5,225 ha would have to burn every year to produce the expected negative exponential distribution. When fire frequency data fit (or are fitted to) a negative exponential model, the general fire cycle is the reciprocal of the hazard of a stand burning in a given year, which equates to the mean stand age (Agee, 1993; Finney, 1995; Van Wagner, 1978). However, when this theoretical distribution is plotted against the 1966 empirical distribution of Quetico's forest area in each time-since-fire class, the results show that the data do not fit the expected curve (Figure 5.5). Therefore, the mean stand age is not a good estimate of the fire cycle. The change in slope at approximately 50 years corresponds to the change in fire frequency due to the large stand-replacing fires of the late 1800s and the early 1900s.

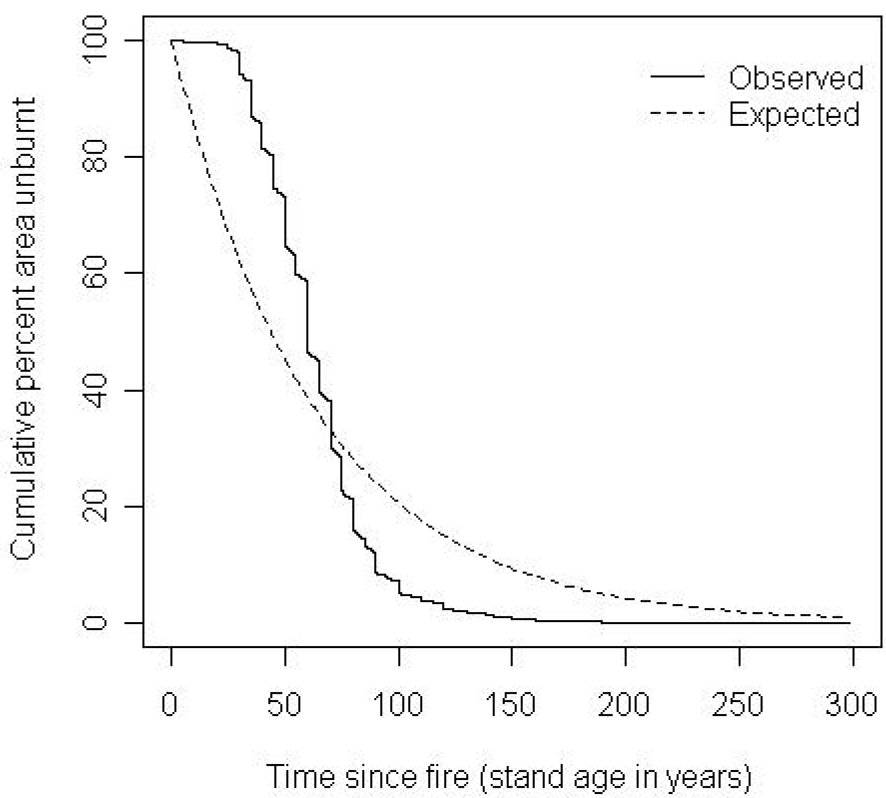

Figure 5.5 1966 time-since-fire distribution vs. expected negative exponential.

5.2 Current Fire Cycle

The fire cycle for the time period between 1921 and 2007, using the digital fire atlas, was 342 years when the lakes are subtracted from the area burned. Two very large fires, one that occurred in 1936 that burned approximately 29,022 ha (light pink area of Figure 5.6 below), and another in 1995 that burned approximately 27,900 ha (red area in Figure 5.6 below), account for the majority of the area burned during this time period, and therefore influence the burning rate the most.

5.3 Time-since-fire Map

The time-since-fire map, created by combining the 1966 FRI age distribution map (updated to 2007) and the digital fire atlas, is presented in Figure 5.6. The light pink area of the map represents forest stands that originated after the large fires of 1936. The red area represents the area burned by the 1995 fires. The majority of stands that do not originate during these two fire years, according to the time-since-fire map, came from fires between the late 1800s and the early 1900s, which correspond with the Park's oral history fire records of the 1895 (and/or 1894), 1910, 1917 fires (Ontario Parks, no date) and the BFOLDS modelled fire return interval fire year of 1891 (M. Gluck, personal communication, 2008). Stands with time-since-fire dates from 201-220 likely originated after the 1803-1804 fires[14] (Ontario Parks, no date).

[14] Grace Lee Nute, "Voyageur's Highway" p. 61, J.D. Cameron, Hudson's Bay Company trader at Rainy Lake wrote that in 1803 and 1804, "the whole country almost from one extremity to the other was in a continual blaze and stopped only by the snow of autumn".

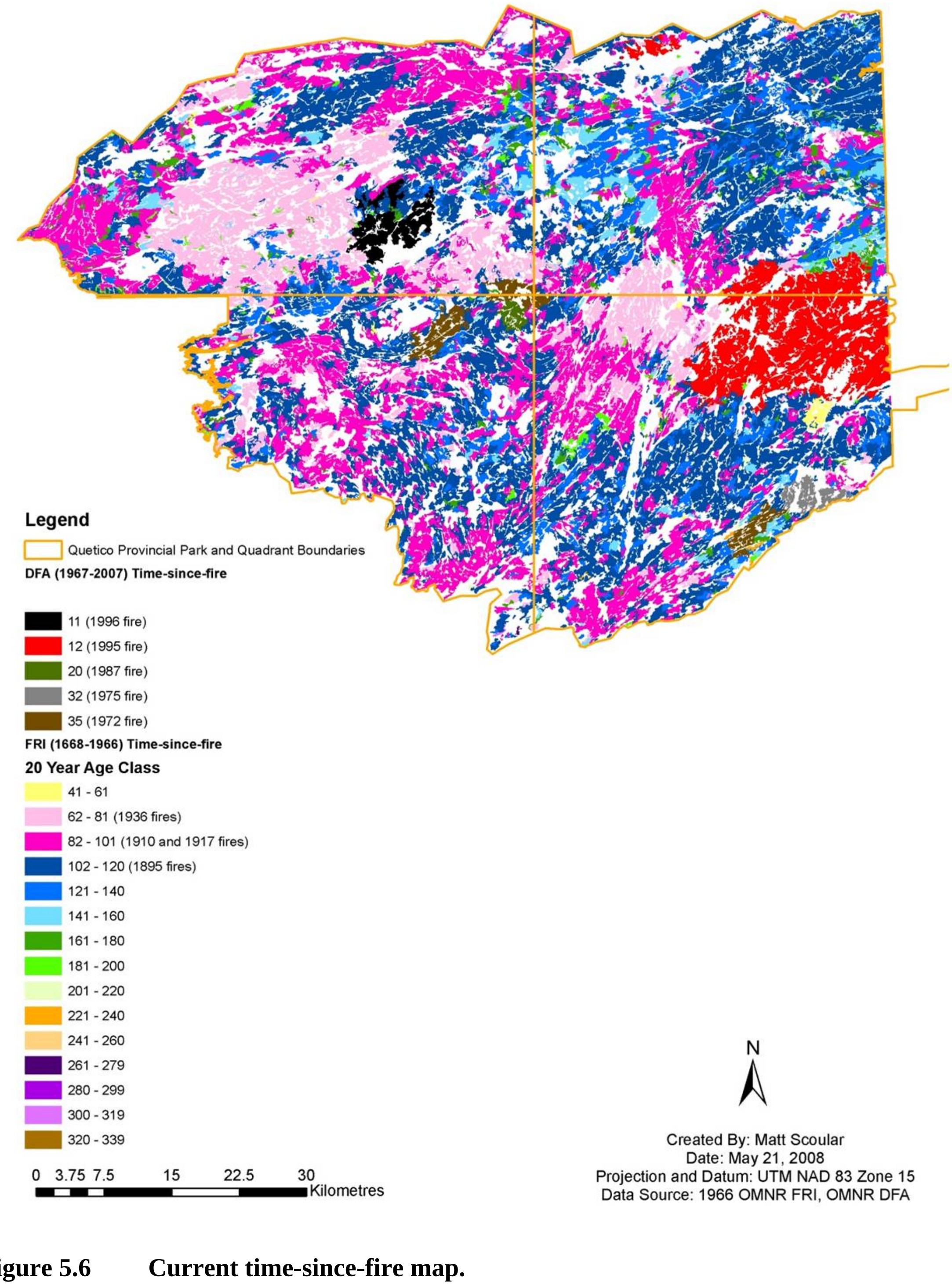

Figure 5.6 Current time-since-fire map.

Note: the 1966 FRI provided historical fire years within 20 year age-classes (for the time period 1668-1966) whereas the digital fire atlas provided the exact year of the fire (for the time period 1967-2007).

5.4 Survival Analysis

5.4.1 Survival Function Spatial Variability

The survival function can be estimated using only the information concerning the age at which a stand burned and can incorporate area weighting. Area weighting accounts for the area burned by using the size (in ha) of the most recent fire on a forest stand to attach greater statistical importance to larger stand-replacing fires in the non-parametric estimation of the time-to-fire survival curve. Given that the research goal was to estimate the stand-replacing mean fire interval for any given forest stand in Quetico, a point-based concept, accounting for area burned was not considered necessary. However, to validate this choice the survival function for the entire Park (global) was estimated with and without area weighting (Figure 5.7).

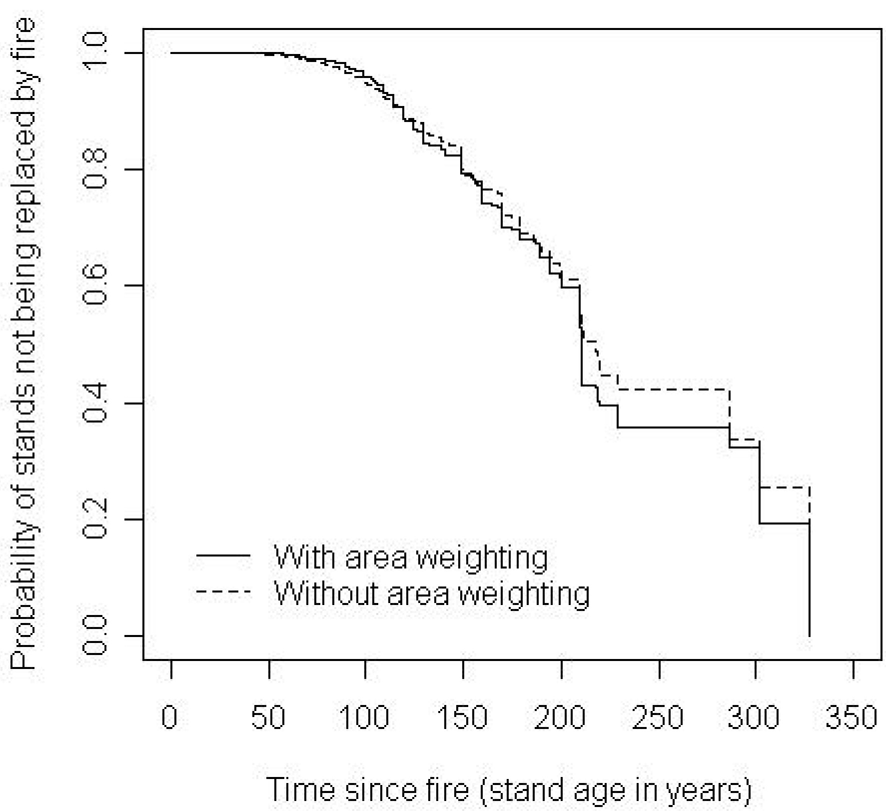

Figure 5.7 Global survival function with and without area weighting.

The two estimated survivor functions are virtually identical until 200 years. Where the estimates vary after 200 years, statistical precision becomes low due to small sample sizes (i.e., the small proportion of old growth stands) and therefore should not necessarily be interpreted as important.

The global (entire Park) survival function during 1668-2007 is presented in Figure 5.8. The raw data results of the Kaplan-Meier survival analysis are included in Appendix E.

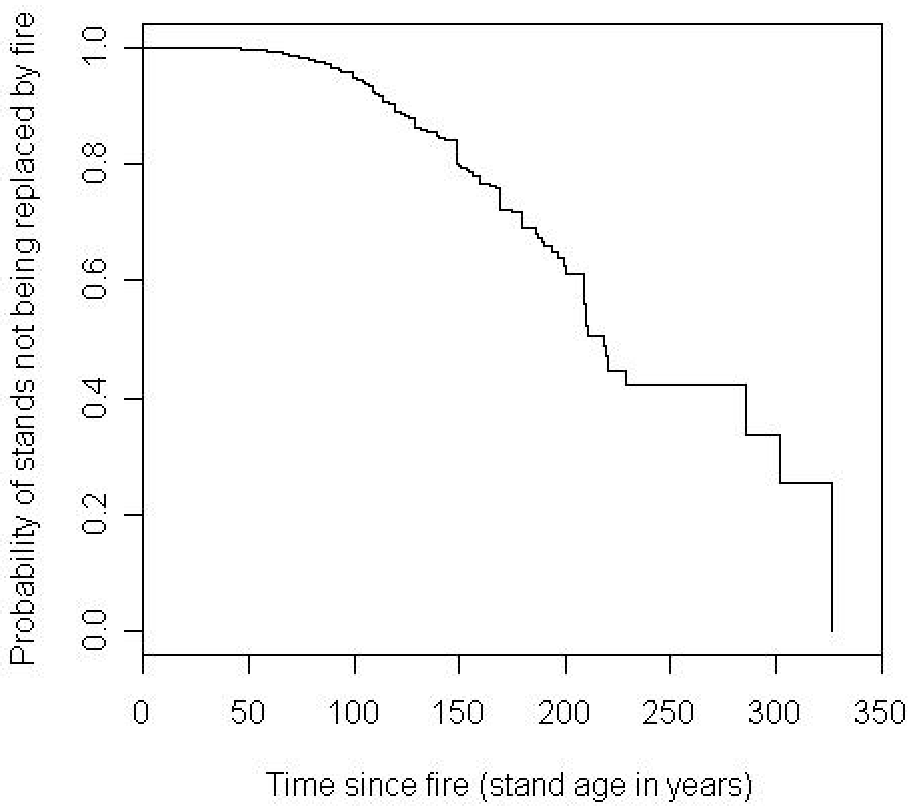

Figure 5.8 Global survival function.

There is a 5% probability that stands will burn before they reach 100 years of age (60% of the stands are younger than 100 years). There is a 20% probability that stands will burn before they reach 150 years. There is a 40% probability that stands will burn before they reach 200 years. However, at 200 years only 46 stands remain out of 7,493. Therefore, the probability of stands not being replaced by fire should not be interpreted as being precise beyond 200 years due to a small sample size for these age classes.

The global survival functions for the four quadrants of the Park during the time period 1668-2007 are presented in Figure 5.9.

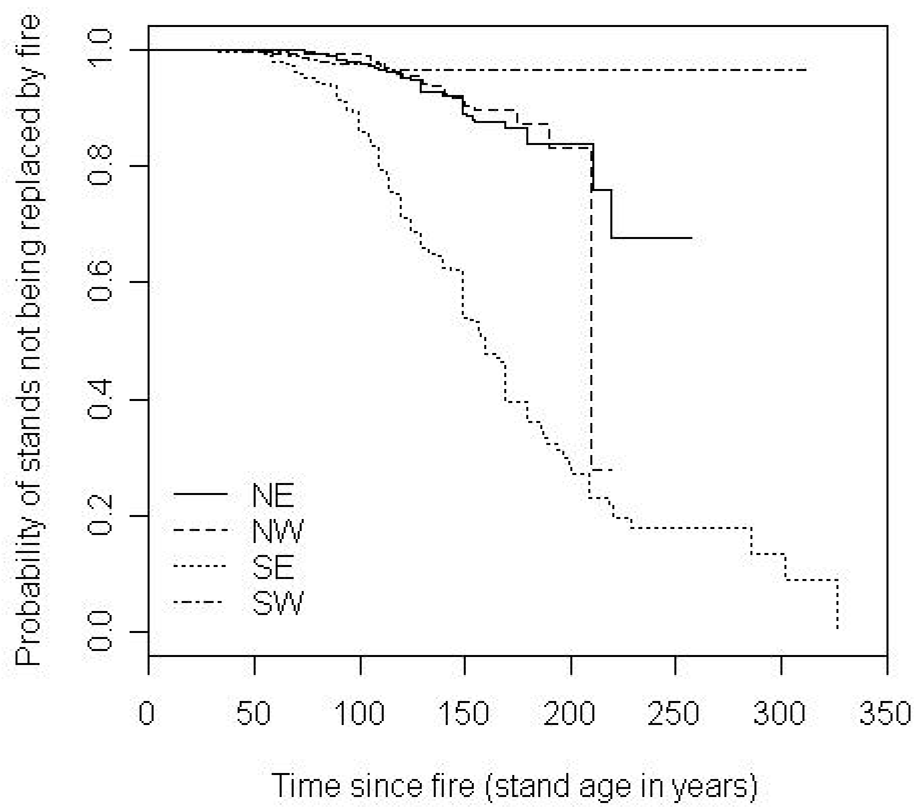

Figure 5.9 Survival functions for the Park's four quadrants.

Except in the southeastern quadrant, the survivor function estimates do not reduce to zero because the oldest stands in these other three quadrants are incomplete observations (i.e., we do not have a standing-replacing fire interval for the oldest stand). Therefore, subsequent estimates of the mean fire interval are not considered as statistically reliable for these quadrants as opposed to the southeastern quadrant or the entire Park. However, they represent the only mean fire interval statistical estimates currently available for Quetico. The survival function for the southwestern portion of the Park (a horizontal straight line after 120 years) indicates the lack of fires in older growth forest stands in this quadrant. The southwest is renowned for its old forest. The northwestern and northeastern quadrant estimates appear to be virtually identical to one another.

A log-rank test was used to test whether or not survival functions in the four quadrants were significantly different statistically (Table 5.1).

Table 5.1 Four quadrant comparison of survival functions.

Quadrant	**N (sample size)**	**Observed**	**Expected**	**(O-E)^2/E**	**(O-E)^2/V**
NE	2,084	105	229.3	67.4	111.2
NW	2,036	53	139.1	53.3	70.1
SE	1,955	421	159.0	432.0	599.5
SW	1,418	26	87.6	30.4	36.5

Chisq = 600 on 3 degree of freedom, p = 0

Chisq 0.05 critical value = 7.81

This test clearly indicates that survival functions for the southwestern and southeastern quadrants of the Park are significantly different from one another and from the northwestern and northeastern quadrants of the Park. Based on the similarities between the survival curves of the northwestern and northeastern quadrants, a second log-rank test was performed and the results are presented in Table 5.2.

Table 5.2 Quadrant comparison within the northern half of the Park.

Quadrant	N (sample size)	Observed	Expected	(O-E)^2/E	(O-E)^2/V
NE	2,084	105	100.5	0.200	0.563
NW	2,036	53	57.5	0.394	0.563

Chisq = 0.6 on 1 degree of freedom, p = 0.453

Chisq 0.05 critical value = 3.84

The test verifies that the survival functions for the northwestern and northeastern quadrants of the Park are not statistically significantly different.

5.4.2 Upland and Lowland Survival Functions

Kaplan-Meier estimates of the survivor function for the upland and lowland forest stands during the time period 1668-2007 are presented in Figure 5.10.

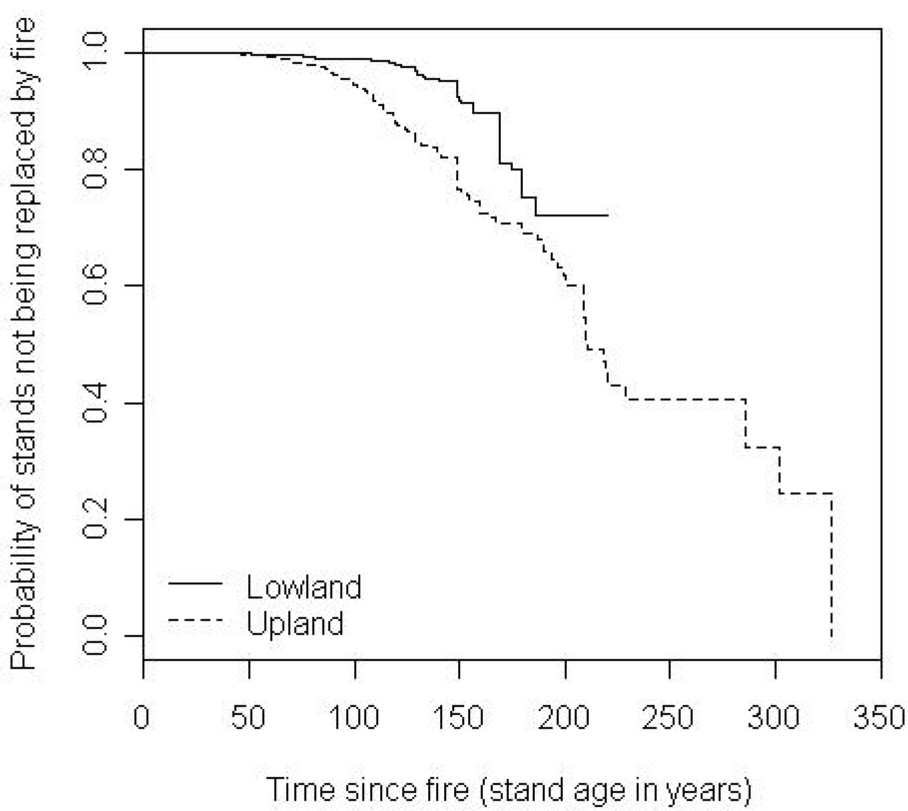

Figure 5.10 Survival functions for the Park's upland and lowland forest stands.

It is evident that the two distributions in Figure 5.10 are very different from one another. The two distributions were not compared statistically due to the marked difference in the sample size (only 6% of the forest stands were classified as lowland). Survival functions for all stands versus only the upland stands are presented in Figure 5.11.

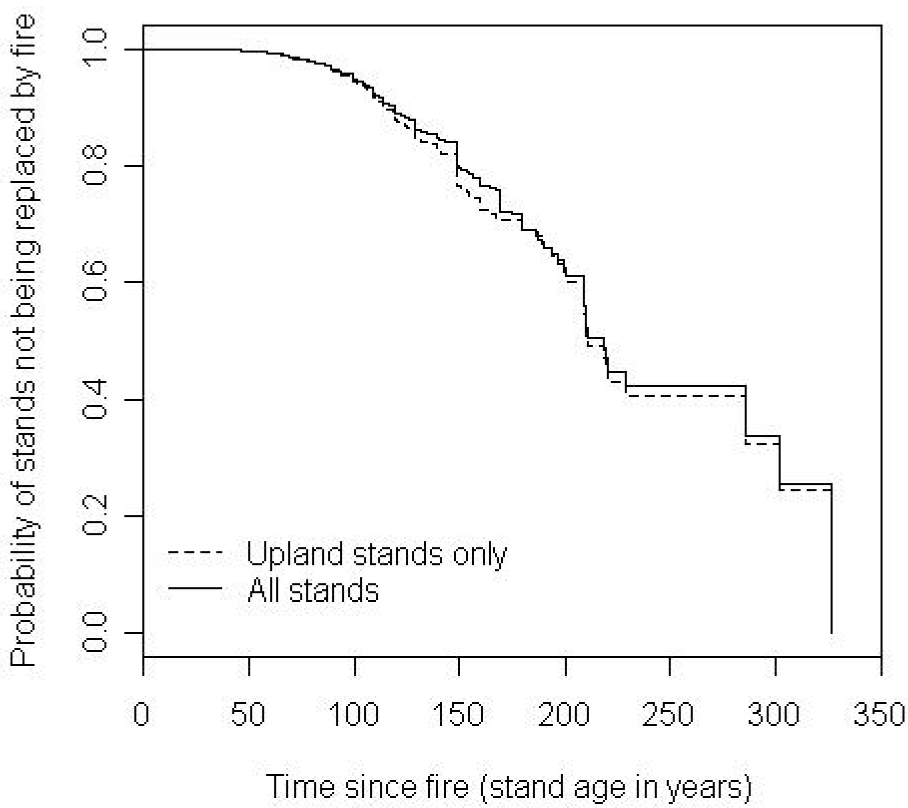

Figure 5.11 Survival functions for the Park's forest stands versus only the upland stands.

The distribution of the total Park's forest stands is virtually identical to the distribution of upland stands only. This can be attributed to the fact that most of Quetico forest strands are upland. Only FRI forest stands were digitized which means some lowland treed areas were not captured (i.e., nutrient rich swamps). However, these areas make up a small proportion of the Park's total land area (Scoular and Suffling, 2008; Solomon, 2007; Woods and Day, 1976). Heinselman (1996) asserted that non-lethal (surface) fires at long fire intervals burned these swamps, rather than stand-replacing fires, as wetland areas are last to dry out even during severe drought. Since restricting the data to upland stands had only a minimal effect on the estimated survival function, mean fire interval estimates for the area weighted vs. non-area weighted and the spatial and temporal partitions were based on the complete dataset (all forest stands) .

5.4.3 Mean Fire Interval Spatial and Temporal Variability

Once the FRI was updated with the digital fire atlas (for fire years 1972, 1975, 1995 and 1996) fire intervals for 615 of the 7,493 forest stands were obtained. The mean fire intervals estimated from the Kaplan-Meier survival analysis are presented in Table 5.3 for the whole Park, and various spatial and time partitions, using two methods of calculation.

The stand-replacing mean fire intervals for the entire Park based on data derived from 1668-2007 with and without area weighting are 223 and 230 years respectively. However, the corresponding confidence intervals overlap and therefore are not statistically different from one another. All other mean fire interval estimates were therefore computed without area weighting.

The mean fire interval for the entire Park based on data derived from 1970-2007 is noticeably shorter, at 198 years, than the mean fire interval for the entire temporal extent of the study.

Although the corresponding confidence intervals for the mean fire intervals of the northern quadrants of the Park did not overlap, their survival functions were not significantly different from one another. According to (D. Matthews, personal communication, 2008) comparing two survivor functions (northeastern quadrant vs. northwestern quadrant) provided more latitude for sampling variability than in comparing two mean fire intervals, which were point estimates. Therefore, the northern quadrants fire frequency should not be considered significantly different statistically. The southeastern quadrant had the shortest mean fire interval, at 175 years. The southwestern quadrant had the longest mean fire interval, at 306 years.

The upland mean fire interval was 3 years shorter than the mean fire interval for the entire Park (upland and lowland) but the corresponding confidence intervals were overlapping substantially. Therefore, these mean fire intervals were not statistically significantly different. Although the mean fire interval for the lowland forest stands was shorter than the mean fire interval for the upland forest stands (contrary to what one would expect), this result should not be interpreted with caution due to small sample size (fewer than 426 observations out of 7,493) relative to the upland forest stands. Therefore, the only statistically reliable comparison was the mean fire interval estimates for all forest stands versus the upland stands. However, as discussed earlier they were not statistically different. Removing the lowland stands from the dataset did not have a significant effect on the mean fire interval estimate.

Table 5.3 Mean fire interval spatial and temporal variation.

Spatial Extent	Time Period	Mean Fire Interval (years)	Standard Error (years)	95% Confidence Interval
Entire Park (upland and lowland) without area weighting	1668-2007	*230	7	216-244
Entire Park (upland and lowland) with area weighting	1668-2007	*223	1	221-225
Entire Park	1970-2007	*198	8	182-214
NW Quadrant	1668-2007	202	3	196-208
NE Quadrant	1668-2007	231	5	221-241
SW Quadrant	1668-2007	306	2	302-310
SE Quadrant	1668-2007	*175	6	163-187
Entire Park (upland only)	1668-2007	*227	7	213-241
Entire Park (lowland only)	1668-2007	203	2	199-207
Measured Fire Zone	1668-2007	263	8	247-279
Prescribed Fire Zone	1668-2007	*216	7	202-230
Entire Park (BFOLDS comparison)	1963-2003	*201	6	198-213
Entire Park (Without Fire #141)	1668-2007	318	5	308-328

Note: Survival function and mean fire interval estimates derived from 1966 FRI (1668-1966) and digital fire atlas (1967-2007)

** statistically reliable mean fire interval estimate (i.e., oldest stand a complete observation)*

The mean fire interval for the Measured Zone was 263 years, whereas the mean fire interval for the Prescribed Fire Zone was 216 years.

The mean fire interval for the BFOLDS Fire Weather Index time period was 201 years.

The mean fire interval for the Park without Fire #141 was 318 years.

5.4.4 Probability of Burning

Another way of interpreting the survival function estimated through survival analysis is by plotting the probability of burning function (the probability of a stand being replaced by fire). Figure 5.12 presents the probability of burning function for the entire Park based on the data from the period 1668-2007.

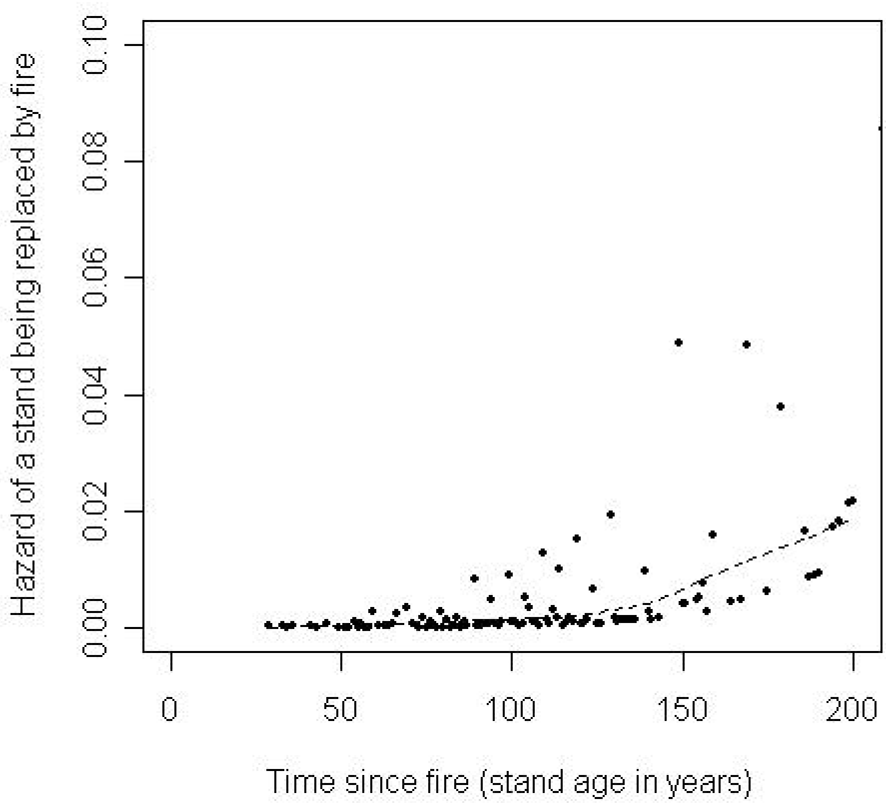

Figure 5.12 Probability of burning function.

When interpreting the scatter plot line on Figure 5.12, the probability of burning appears relatively constant (independent of stand age/time-since-fire) for stands younger than 125 years (see Methods section 4.5.2.2 for a discussion on how the scatter plot smooth line was derived). However, among older stands there is an increasing trend in the average probability of burning. To illustrate the interpretation of this plot, the point at (149, 0.05) indicates that stands which are 149 years old have a 5% estimated probability of burning at that age. This estimated probability is based on 25 fire intervals at 149 years (complete observations) and all 487 right-censored time-since-fire observations (i.e., minimum fire interval estimates) involving observed stand ages of at least 149 years.

Chapter 6
Discussion and Conclusion

6.1 Fire Frequency Study Statistical Methods

Two methodological problems were observed in the application of non-parametric Kaplan Meier survival analysis in two recent fire frequency studies[15] (Cyr et al., 2007; Lauzon et al., 2007). The first was introduced when incomplete (time-since-fire) observations were treated as complete (fire interval) observations. Time-since-fire observations represent a minimum estimate on the time between two stand-replacing fires (fire interval). All that is known is that the forest stand has survived to its current age and therefore such data must be treated as a lower bound (right-censored) on the fire interval (D. Matthews, personal communication, 2008; Polakow and Dunne, 1999). As discussed in the methods section, complete observations refer to forest stands where a fire interval is known. For this research, the complete observations were created by updating the initial 1966 FRI time-since-fire map (1668-1966) with the digital fire atlas for 1967-2007. As a result, fire intervals were available for FRI forest stands that overlapped with the fires recorded in the digital fire atlas. Studies that have failed to right-censor incomplete data (if fire frequency data were fitted to a Weibull or non-parametric methods were used) have produced erroneous results and should be reanalyzed using proper statistical techniques (D. Matthews, personal communication, 2008; Polakow and Dunne, 1999).

The second problem was introduced when partitioning fire frequency data temporally (D. Matthews, personal communication, 2008); usually after spatial partitioning was attempted. To assess the influence of temporal parameters on fire frequency, data have been partitioned at hypothesized change points where fire frequency has previously been observed to increase or decrease (e.g., a decrease after the end of the Little Ice Age in 1850 or following effective fire suppression starting in the 1950s). After partitioning the data, separate distributions are achieved and the change in fire frequency can be linked to the parameter of interest (Johnson and Gutsell, 1994). However, if the distributions are not homogeneous, other change points or spatial factors (e.g., distance to fire break, ecological site type, topography, surficial geology, etc.) need to be considered. The problem involved in this approach is that survival analysis (Weibull or non-parametric methods) cannot be performed on temporal partitions unless at least one fire interval (complete observation) ends within the time period

[15] Cyr et al. (2007) and Lauzon et al. (2007) did not explicitly state that they right-censored time-since-fire observations. The censoring criteria used can be found in Bergeron et al. (2001) and Bergeron et al. (2004).

of interest. It is easy to see how this problem occurred, as the first problem led to the incorrect distinction between time-since-fire and fire interval observations. Another possible reason could be that the two most commonly cited fire frequency study methodology review papers (Johnson and Gutsell, 1994; Johnson and Van Wagner, 1985) do not discuss this requirement for applying the Weibull theoretical model nor do they discuss non-parametric survival analysis techniques.

Another limitation in previous fire frequency studies (except Cyr et al., 2005 and Lauzon et al., 2007) was that they fitted fire frequency data to either the negative exponential or Weibull theoretical models without first exploring the empirical distribution. This step may have been performed, but was not reported in the literature. Parametric models should not be assumed without first testing the fit of the empirical data to the negative exponential and Weibull models. Empirical estimators of the survival function provide reliable fire frequency estimates without potentially problematic ecological assumptions (e.g., constant probability of burning).

Fire frequency studies should explicitly state how data are censored. As discussed earlier, some studies may not have right-censored time-since-fire observations, thus leading to incorrect temporal partitioning and so producing errors. To avoid this in future studies, the OMNR should create standards for fire frequency estimation using survival analysis through collaboration with statisticians. Given this research employed non-parametric Kaplan Meier survival analysis methods, time-since-fire (incomplete) observations were right-censored and fire interval (complete observations) included those stands that had burned at least twice. The first fire interval ended in 1972. Therefore the temporal partition could start before 1972 but had to end after 1972 for at least one fire interval to be included in the analysis.

6.2 Historical Forest Age and Composition

The forest age distribution and age distribution map for 1966 confirmed that the majority of the Park's forest originated after the large stand-replacing fires of the late 1800s to early 1900s. These fires were responsible for the size, shape and configuration of the forest patch mosaic. Likewise, analysis of the Park's 1966 forest composition (by OMNR WG) as a function of time-since-fire, confirmed that Jack Pine and Black Spruce stands originated from these fires which resulted in the rejection of hypothesis #1 (i.e., *in 1966, the relative area occupied by fire dependant species (Black Spruce and Jack Pine) was not a function of time-since-last stand-replacing fire*). The Park's 1966 time-since-fire distribution (cumulative percent area unburned as a function of time-since-fire) also did not fit the expected negative exponential theoretical model. In support of this result, several authors have observed nonequilibrium forest age structures in boreal mixed-wood ecosystems (Baker, 1989; Cumming et al., 1996) and ecosystems characterised by large stand-replacing fires (Suffling, 1988; Suffling, 1990; Turner and Romme, 1994).

6.3 Current Fire Cycle

The Park uses Heinselman's "natural" (1727-1910) fire cycle estimate (100 years) as a guide to reintroducing fire into the Park. The current fire cycle estimated in this study can be used to evaluate the deviation from Heinselman's natural estimate. If the Park decides to continue to use the fire cycle concept in fire management, this research has provided an up-to-date fire cycle. Prior to this research, the Park estimated the fire cycle using the same general methods as this research (computing a burning rate using the digital fire atlas). However, they did not subtract the area covered by lakes from the area burned and therefore underestimated the length of the fire cycle (by approximately 12%). Bridge (2001) asserted that although fire cycle estimates derived using a burning rate approach were almost always lower than those derived using Maximum Likelihood Estimator (MLE) survival analysis (Reed, 1997), the estimates are on average within 5% of one another. The fire cycle estimates are likely lower than the MLE estimates because averaging accounts for all fires that burned within a certain time period, including areas of over burning, whereas MLE only accounts for the date of the most recent fire (Bridge, 2001). This suggests that the simpler of the two methods (burning rate) may be used with confidence when estimating the current fire cycle.

6.4 Fire Frequency Spatial and Temporal Variability

For the first time, this research estimated a current stand-replacing fire frequency (mean fire interval) for the entire Park over a long time period (1668-2007) using current survival analysis methods. The mean fire interval (230 years) for the entire period of study was influenced by natural (e.g., climate variation) and human (e.g., fire suppression, recent climate warming) factors over the last three centuries. Hypothesis #3 (i.e., *stand-replacing fire frequency does not vary between a recent time period (1970-2007) and the entire time period of study (1668-2007)*) was rejected as considerable temporal variability in fire frequency was observed between the entire time period of study and a recent time period (1970-2007). This suggests that recent climate warming within the North American boreal forest (Flannigan et al., 1998, Stocks et al., 1998) and northwestern Ontario specifically (Lemieuk, 2007; Racey, 2004; Thompson, Flannigan, Wotton & Suffling, 1998), has influenced fire frequency in the last thirty to forty years. This trend is expected to continue (Lemieuk, 2007; Racey, 2004; Colombo, et al., 1998). However, other potential reasons for the observed trend could be a decrease in fire suppression during this time period or the data weighted most heavily in the analysis (Fire #141 fire interval data) came from the most recent time period and could have skewed the results. The survival function estimates, and therefore mean fire interval and probability of burning estimates of this research were primarily influenced by complete observations (fire intervals) provided by Fire #141. This 1995 fire provided 75% of the fire intervals used in the survival analysis. Future studies should correlate fire frequency data to weather data (e.g., Fire Weather Indices) to confirm the influence of recent climate variation (1963-present) on the temporal variability of fire frequency (Suffling, 1992; Lauzon et al., 2007). Fire Weather Indices are reconstructed using precipitation, temperature, relative humidity, wind direction and wind speed data (Canadian Forestry Service, 1984).

Although not tested with empirical data for Quetico, the Park's present fire frequency is likely an artifact of over 50 years of fire suppression. Once the OMNR BFOLDS research is complete, a comparison between it and the results of this research can be used to assess the influence of fire suppression on the Park's fire frequency. The BFOLDS mean fire interval estimates will be derived from fires that have not been suppressed (M. Gluck, personal communication, 2008). This research has estimated mean fire intervals from fires that were suppressed and those that were allowed to burn. Therefore, the influence of fire suppression on the Park's fire frequency can be inferred.

Considering the possible effects of recent climate warming and fire suppression, fires that are allowed to run their course should not be expected to behave "naturally". Natural variation in fire frequency over time, which appears to follow distinct trends driven by large-scale climate processes (e.g., end of the Little Ice Age), complicates the goal of restoring fire to ecosystems. Consequently, it will likely be ineffective to manage for a particular "natural" fire frequency (i.e., 100 year fire cycle) and therefore fire management must be adaptive to accommodate future knowledge generated about the complex interactions between fire, climate and landscapes. For example, once the spatial and temporal variability of fire frequency is characterised at multiple scales, it can be incorporated into fire management goals and operations (e.g., the timing and location of prescribed burns).

Hypothesis #2 (i.e., *stand-replacing fire frequency does not vary between different areas within the Park*) was rejected as fire frequency spatial variability within Quetico was confirmed by producing survival functions and obtaining mean fire interval estimates for the four quadrants of the Park. Although between 1909 and 1946 White and Red Pine were cut in the northwest section of the Park, the majority of industrial logging (of Jack Pine, Black Spruce, Balsam Fir and Trembling Aspen) occurred between 1961 and 1971 in the northeast corner of the Park (Peruniak, 1990; Solomon, 2007). Because the survival function was not significantly different between the northwestern and the northeastern quadrants, logging can be considered not to have greatly influenced the Park's stand-replacing fire frequency. However, this does not mean that past logging practices have not influenced lower-severity fire regimes or will not influence the Park's stand-replacing fire frequency in the future. For example, outbreaks of spruce budworm have occurred in logged areas where Balsam Fir has regenerated leading to a buildup of dead material which may be conducive to large catastrophic fires (Fleming et al., 2002; Stocks and Walker, 1973; Stocks, 1985; Stocks, 1987). Conversely, other studies suggest that logging fragments the landscape patch mosaic leading to a decrease in fire frequency due to the creation of a discontinuous fuel load and less fire-adapted (and therefore fire prone) species (Bergeron, 2006; Grant, 2007; Suffling, Grant and Feick. (in press)). This situation presents considerable challenges for Quetico's FMP. Large stand-replacing fires are natural in Quetico's forests. However, forestry lands adjacent the eastern border of the Park must be protected from fire.

The survival functions of the northern and southern sections of the Park may differ as a result of the transition from the mixed-wood (southern) to the boreal (more northern) forest ecoregions. However, the southeastern and southwestern quadrants survival functions were different as well. Bergeron et al. (2004) found that fire frequency did not differ between mixed-wood and boreal ecoregions of northwestern Quebec. Another hypothesis is that the abundance of lakes and wetlands (distance to fire break) (Table 6.1) may be responsible for fire frequency variability throughout Quetico. Therefore, the lack of area burned in the southwestern quadrant may be correlated to the fragmentation of the forest landscape by these fire breaks. Further studies are needed to test this hypothesis.

Table 6.1 Percent abundance of water and wetlands (fire breaks) by Park quadrant.

Quadrant	**Area of Land and Water/Wetlands (Ha)**	**Area of Water/Wetlands (Ha)**	**Proportion of Water/Wetlands (%)**
Northwest	132,038	28 384	22
Northeast	116,330	19,897	17
Southwest	**96,554**	**23,053**	**24**
Southeast	127,333	27,469	22
Totals:	**472,255**	**98,803**	**21**

Note: summary statistics based on OBM NRVIS water GIS shapefile

The spatial pattern (size, shape and configuration) of fire breaks could also have influenced fire frequency spatial variability (Figure 6.1). The complex spatial pattern of the lakes in the southwest quadrant (longest mean fire interval at 306 years) may have prevented the spread of fires. The southeast to northeast orientation of the lakes in the southeastern quadrant (shortest mean fire interval at 175 years) may have facilitated the spread of fire. Further studies are needed to test these hypotheses.

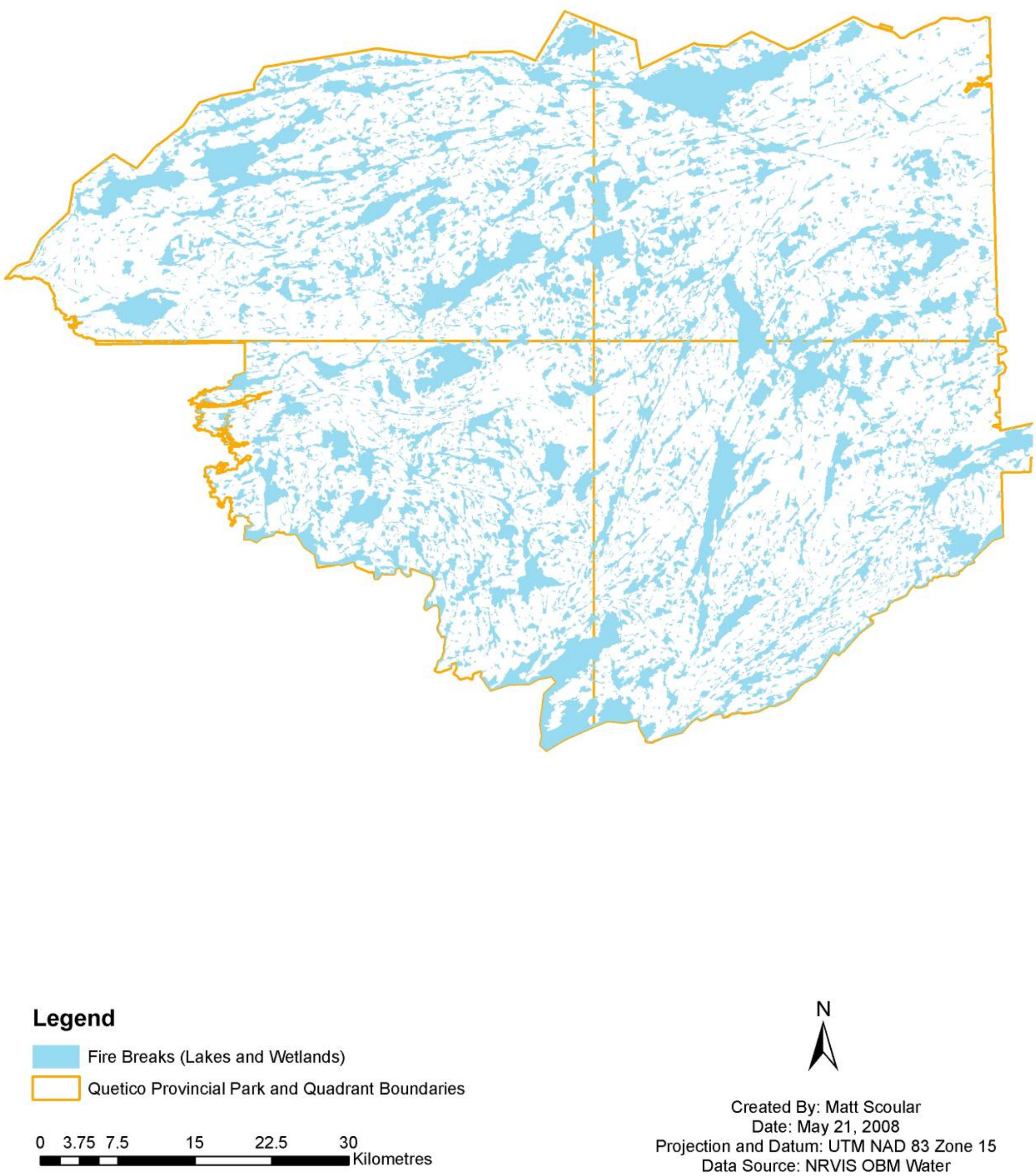

Figure 6.1 Spatial pattern of fire breaks by Park quadrant.

6.5 Probability of Burning

The probability of burning in boreal forests has been explained by both the weather and fuels hypotheses. The weather hypothesis suggests that large stand-replacing fires are a function of extreme fire weather and burn irrespective of fuel conditions (i.e., stand age/time-since-fire) (Bessie and Johnson, 1995; Johnson et al., 1998). The fuels hypothesis suggests that variation in fuel types affects fire spread and severity (Cumming, 2001). Studies in the boreal forests of Scandinavia and western North America suggested that the probability of burning increases with time-since-fire, distance to natural fire breaks and conifer dominance of stand composition (Schimmel and Granstrom, 1997; Cumming, 2001; Hellberg et al., 2004; Tanskanen et al., 2005).

Anecdotal field observation (L. Solomon, personal communication, 2008) and scientific study in Quetico (Lynham and Curran, 1998) and the region (Frelich and Reich, 1999), have suggested that many old growth White and Red Pine stands have been killed by large stand-replacing fires (e.g., Fire #141) and have regenerated to either Jack Pine (Lynham and Curran, 1998) or Trembling Aspen (Frelich and Reich, 1999) instead of their pre-fire composition. This observation is contrary to what was observed in practice. Stand-replacing fires have been known to reduce to surface fires when entering old growth Pine stands due to a lack of ladder fuels (R. Suffling, personal communication, 2008). It is likely that fire suppression has been effective in reducing the frequency of low severity surface fires within the Park and could have led to an increased amount of fuel in its old growth Pine stands. This is one hypothesis as to why hypothesis #4 (i.e., *the probability of burning is independent of time-since-last stand-replacing fire*) was rejected whereby an increasing probability of burning trend in stands over 125 years was observed. This contradicts the assumption that the probability of burning in boreal forests is independent of time-since-fire (constant). However, Johnson and Gutsell (1994) stated that "the increasing hazard (probability of burning) only describes a relationship between survival time and chance of burning; it does not define the nature (cause) of this relationship" (p. 250). Stand-level studies based on empirical data are required to test the potential causes (e.g., random ignition and fuel accumulation). Further studies are also needed to assess whether the observed increasing probability of burning trend is in fact naturally inherent to Quetico's forest ecosystem.

6.6 Implications for Fire Management

As discussed in the literature review, fire management within Quetico's two fire zones is considerably different. Fires are normally allowed to run their course in the Prescribed Natural Fire zone and fires are suppressed in the Measured zone. Therefore, current mean fire interval estimates were derived for both of these zones to provide context for the comparison between "natural" estimates of previous studies (Heinselman, 1973; Woods and Day, 1977b) and future studies (OMNR BFOLDS study).The mean fire interval for the Measured Fire zone (263 years) was longer than the mean fire interval of the Prescribed Natural Fire zone (216 years) suggesting that fire frequency in Measured Fire zone may have been influenced by fire suppression.

Although useful for broad fire management planning and comparison with spatially explicit simulation models (e.g., BFOLDS), this study's results should not be used to set specific fire management prescriptions. However, the results of this research suggest that natural and human factors have led to stand-replacing fire frequency spatial and temporal variability within Quetico. Research on "natural" fire frequency spatial, temporal and stochastic variability within the Park is being completed by the Forest Policy Section of the OMNR in the summer of 2008 (using this study's FRI GIS database and BFOLDS). The spatially explicit results of the OMNR research can be used to set landscape-scale fire management goals (e.g., the appropriate timing, location, size and spatial pattern of prescribed burns). Correlation between fire frequency and potential spatial biases (e.g., distance to fire break, surficial geology, etc.) can be also studied using the BFOLDS results.

6.7 Conclusion

Estimates of fire frequency, the confidence intervals of these estimates and their variability in space and time are critical to understanding the effects of fire across scales (Keane et al., 2004). Fire frequency spatial and temporal *variability* is likely more important than *mean* fire frequency as this variability has allowed Quetico's ecosystem to adapt to change (e.g., climate variation). Therefore, further studies into Quetico's fire frequency variability should be undertaken. For the effective restoration and maintenance of the ecological integrity of Quetico's forests and managed boreal forests generally, fire management should acknowledge fire frequency spatial and temporal variability. The results of this research will provide Park managers with an increased awareness of fire frequency spatial and temporal variability within Quetico to support new management policies and direct future fire research. The observed spatial variability suggests that fire frequency studies should be undertaken at larger (coarser) scales to characterise the regions fire regime in support of landscape level fire management planning. The observed temporal variability suggests that ecological renewal targets should not be based on static historical fire frequency estimates (i.e., Hiensleman's 100 year fire cycle).

This thesis identified eight research objectives and Table 6.2 summarizes how these objectives were met and where they are addressed in the thesis.

Table 6.2 Research objectives, how objectives were met and relevant thesis section(s)

Research Objective	How Objective Met	Relevant Thesis Section(s)
1. **Conduct a literature review of fire management in Ontario, its provincial parks and specifically, Quetico Provincial Park**	Fire management literature review provided	3.1
2. **Conduct a literature review of fire frequency study concepts, data collection and statistical approaches and summarize the results of fire frequency studies relevant to the Park and the region**	Fire frequency study concepts, data collection and statistical approaches literature review provided Literature review of fire frequency studies relevant to Quetico provided	3.2 and 3.3
3. **Georectify and digitize the 1966 OMNR FRI**	The 1966 FRI was georectified and digitized and updated with the digital fire atlas using GIS	4.1, 4.2, 4.4.3 and 5.3
4. **Perform a historical forest analysis using the 1966 FRI**	1966 age distribution and map produced 1966 forest composition as a function of time-since-fire assessed 1966 time-since-fire distribution plotted against the negative exponential theoretical fire model	4.3 and 5.1
5. **Estimate survival functions, mean fire intervals and the probability of burning for the Park using contemporary methods**	Survival curves and mean fire intervals were estimated using Kaplan-Meier survival analysis The Park's probability of burning was estimated	4.5, 5.3 and 5.4
6. **Characterise fire frequency spatial and temporal variability within Quetico**	Survival functions and mean fire intervals for different parts of the Park and different time periods were estimated	5.4.1, 5.4.3

Research Objective	How Objective Met	Relevant Thesis Section(s)
7. **Make recommendations for future fire frequency research**	Based on the rejection of the research hypotheses, further hypotheses were recommended	Chapter 6 and Chapter 7
8. **Evaluate the significance of the research findings**	The significance of the research findings were evaluated in terms of fire management planning, fire management, the scale of fire frequency data collection and analyses, survival analysis statistical methods and current fire frequency knowledge	Chapter 6 and Chapter 7

The time-since-fire and fire interval dataset created during this research from combining the 1966 FRI and the digital fire atlas allowed for the characterisation of Quetico Provincial Park’s historical vegetation-fire dynamics, stand-replacing fire frequency spatial and temporal variability and probability of burning. These results:

- Provided context to landscape-level fire management;
- Synthesized existing knowledge and identified knowledge gaps regarding stand-replacing fire frequency spatial and temporal variability within Quetico;
- Can be used to test and validate future fire frequency models for Quetico and compare to ecosystem simulation model (e.g., BFOLDS) fire frequency estimates;
- Provided current stand-replacing fire frequency estimates as references for evaluating ecosystem change (when compared to the natural variability estimates to be predicted using BFOLDS); and
- Characterised the general trend in the probability of burning during the time period 1668-2007.

Chapter 7
Recommendations and Future Research

The following **general and standards/protocol** recommendations are presented based on the results of this research:

1. Ontario Parks and/or OMNR should rectify and digitize historical FRI for parks (if available) and areas surrounding parks (especially Wilderness class parks) for similar research to support fire and vegetation management activities.
2. The OMNR Geographic Information Branch and Forest Resource Inventory (FRI) experts should establish provincial standards and protocol for geo-referencing (e.g., Root Mean Square Error to map scale ratio) and digitizing historical FRI mapping.
3. The OMNR Science and Research Branch, in concert with Ontario Parks, should establish standards and protocol (in cooperation with statisticians) for fire frequency studies based on survival analysis so that results can be properly integrated into existing fire and vegetation management processes. For example:
    a. Parametric models should not be assumed in fire frequency studies based on survival analysis without first testing the fit of empirical data to theoretical models (i.e., negative exponential and Weibull). Non-parametric (e.g., Kaplan- Meier) methods should be considered first.
    b. Fire frequency studies should explicitly state how data are censored (e.g., right-censored) to avoid confusion and/or the use of improper methods.

The following **fire management planning** recommendation is presented:

1. Quetico's Statement of Fire Intent should be updated to reflect the findings of this research and the findings of the OMNR BFOLDS study. The current Statement of Fire Intent discusses fire frequency studies by Woods and Day (1977a) and Frech, Caputo & McCulloch (1999). However, the Park uses Heinselman's (1973) 100 year fire cycle as an annual area burned target (L. Solomon, personal communication, 2008). The Park's fire management planning document should be consistent with the Park's fire management objectives (annual area burned target) and should be routinely updated when further knowledge is generated on the Park's fire regime.

The following **fire management** recommendations are presented:

1. Woods and Day's (1977b) "natural" fire frequency estimates should not be used as a benchmark for the reintroduction of fire throughout the entire Park as their study covered only 21% of the Park and their fire frequency estimates relied heavily on community break-up estimates, a different ecological process than fire (i.e., successional and self-organizational change, as well as other kinds of disturbance like wind throw, insect outbreak and disease).
2. "Natural" fire frequency estimates (mean fire interval and/or fire cycle) are a research priority for provincial parks (Kingston and Mosley, 2007). The use of either Heinselman's (1973) or Lertzman's (1999) "natural" (pre-European settlement) fire frequency estimates as annual area burned (ecological renewal) targets should be reconsidered in light of this study's fire frequency estimate for the time period 1970-2007. Given the context of a predicted warmer future climate, fires should not be expected to behave "naturally" and studies that assess how fire will affect succession should be incorporated into long-term fire and vegetation management. A static historical annual area burned (fire frequency) target should not be used as fire frequency temporal variability is inherent in Quetico's and northern (boreal) fire-dependant ecosystems generally. Fire frequency spatial and temporal variability should be acknowledged in Quetico's revised Fire Management Plan. Further studies are required to prescribe specific goals for Quetico with respect to fire frequency spatial and temporal variability. The forthcoming natural variability estimates (fire cycle and mean fire interval) of the BFOLDS simulation can be used to guide landscape scale ecological renewal targets.

To achieve these fire management recommendations, **further research** into the following are needed:

1. Since considerable fire frequency spatial variability within Quetico was observed in this research, the study area for future research should extend beyond Park boundaries in attempt to characterise the region's fire regime.
2. Once the spatial, temporal, and stochastic bounds of natural variability are characterised (using BFOLDS) spatial variability should be correlated to potential spatial biases such as proximity to fire break (e.g., lakes, streams and wetlands), surficial deposits, etc. Once complete, spatial biases and temporal variability can be incorporated into the appropriate location and timing of prescribed burns.
3. Once the OMNR BFOLDS research is complete, it can be compared to the results of this research to assess the influence of fire suppression on the Park's fire frequency. The BFOLDS mean fire interval estimates will be derived from fires that have not been suppressed

(M. Gluck, personal communication, 2008). This research has derived mean fire interval estimates from fires that have been suppressed and those that have been allowed to burn. Therefore, the influence of fire suppression on the Parks fire frequency can be inferred.

4. Stand-level studies based on empirical data are required to test the potential causes (i.e., random ignition and fuel accumulation) of the observed increasing probability of burning trend in stands over 125 years old.
5. The temporal accuracy of the 1966 FRI is approximately 20 years. Therefore, time-since-fire estimates can be validated/calibrated using Quetico Foundation and other dendroecological data (i.e., tree core stand ages and fire chronologies) to improve the temporal (fire year estimate) accuracy of this GIS database. However, fire-scarred trees are typically indicative of non stand-replacing fires and high landscape level variability requires the sampling of each stand to correct age (time-since-fire) underestimation for landscape level studies (i.e., studies involving multiple stands).

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Appendix A

Assumptions, Advantages and Limitations of Fire Frequency Study Statistical Methods

Method	Relevant Literature	Assumptions	Advantages	Limitations
Burning Rate (Fire Cycle/Fire Rotation)	Heinselman, 1973; Bridge, 2001	The spatial extent of every fire has been accurately determined	Do not need to fit data to theoretical model (negative exponential or Weibull) Do not need to assume spatial or temporal homogeneity (i.e., probability distributions are the same for all regions or time periods) Simple computation Accurately reconstructs size of fires and their distribution in space and time; therefore characterises the spatial and temporal variability of fire regimes	Lacks statistical model to test hypotheses (e.g., differences in fire frequency between regions or time periods) Evidence of fires partially erased by subsequent fires; therefore data not always available Requires detailed data about fires (e.g., either annual fire maps (digital fire atlases) or time-since-fire maps based on air photos, stand origin and fire scars to determine the boundary of every historical fire)
Negative Exponential Model	Van Wagner, 1978	Study area composed of many equal sized forest stands) (Van Wagner, 1978) A relatively small number of these stands burn each year (Van Wagner, 1978) Stable climate and uniform site conditions throughout study area and over time (Van Wagner, 1978)	Only requires most recent fire (time-since-fire) for each stand in study area (do not need to reconstruct extent of historical fires) (Johnson and Gutsell, 1994; Johnson and Van Wagner, 1985; Van Wagner, 1978)	Requires a homogeneous spatiotemporal dataset (Johnson and Gutsell, 1994; Johnson and Van Wagner, 1985; Van Wagner, 1978) Bias introduced if ecosystem not in steady-state (i.e., the age class distribution different at any one point in time) “Missing tail”: theoretical models tail extends to infinity; fire frequency dataset truncated at oldest stand; old growth harder to age and can be missed because small patches (Finney, 1995)

		Ignitions (lighting) distributed fairly uniformly through time (cause approximately equal number of fires per year) (Van Wagner, 1978) Ignition locations distributed at random throughout study area (Van Wagner, 1978) Hazard of burning independent of stand age (time-since-fire) (Van Wagner, 1978)		Model is aspatial (doesn't characterise size or spatial distribution of fires in study area); assumes fire effects are uniform; however, fire contagious; model does not distinguish between 200 different patches evenly distributed over landscape or 200 spatially aggregated patches (Reed, 1998) Not accurate over short time periods (geometric properties of negative exponential); 20 years minimum (reasonable for stand-replacing fire regimes) Hard (impossible?) to map time-since-fire of low severity (and mixed severity) fire regimes Overestimates fire frequency in most recent time period (because older stands subject to different hazard of burning since they have survived through recent time periods) (Reed, 1998)
Weibull Model	Johnson, 1979	Same as negative exponential model Ignition locations drawn at random from polynomial distribution (power function) based on time-since-fire) (Johnson, 1979) Hazard of burning dependent on stand age (time-since-fire) (Johnson, 1979)	Same as negative exponential model	Same as negative exponential model Cannot treat incomplete observations (time-since-fire) as complete observations (fire interval) (D. Matthews, personal communication, 2008; Polakow and Dunne, 1999) Only allows for comparison of fire frequency for different time periods if one fire interval ends within period of interest (D. Matthews, personal communication, 2008; Polakow and Dunne, 1999)

MLE Survival Analysis	Reed, 1994; Reed, 1998	Same assumptions as negative exponential model Time periods derived independently from data (to avoid change point selection bias) (Reed, 1998) Change in fire frequency occurs instantaneously between periods (Reed, 1998)	Accounts for changing hazard of burning (Reed, 1994) Accounts for spatial autocorrelation in pattern of burning (contagion) (Reed, 1994) Produces confidence intervals for fire frequency estimates (Reed, 1994) Allows for comparison of fire frequency for different time periods (Reed, 1998)	Requires a homogeneous spatiotemporal dataset (Johnson and Gutsell, 1994; Johnson and Van Wagner, 1985; Van Wagner, 1978) Requires fitting data to a negative exponential theoretical model Bias introduced if ecosystem not in steady-state (the age class distribution different at any one point in time) (Fall and Lertzman, 1999) Hard (impossible?) to map time-since-fire of low-severity (and mixed-severity) fire regimes
Non-parametric Survival Analysis (Kaplan-Meier, 1958)	Cyr et al., 2007; Lauzon et al., 2007; ***this thesis***	Stable climate and uniform site conditions throughout study area and over time (spatiotemporal homogeneity assumed) (D. Matthews, personal communication, 2008) Probability of burning derived from empirical data	Does not require fitting data to theoretical model Produces confidence intervals for fire frequency estimates (Cyr et al., 2007; Lauzon et al., 2007)	If distribution not fit theoretical model, censored data may not have same weight in analysis as non-censored data and can therefore lead to underestimation of length of fire cycle (Lauzon et al., 2007) Cannot treat incomplete observations (time-since-fire) as complete observations (fire interval) (D. Matthews, personal communication, 2008; Polakow and Dunne, 1999) Only allows for comparison of fire frequency for different time periods if one fire interval ends within period of interest (D. Matthews, personal communication, 2008; Polakow and Dunne, 1999)

Appendix B

1966 OMNR FRI GIS Database Data Dictionary

Attributes	**STANDNUMF**	**YRSOURCE**	**SOURCE**	**SPCOMP**	**OMNR_FU**	**FIREYR1**	**TSF1**
Definitions	final GIS database stand number	year source	source	forest stand species compositio n	forest unit	fire year 1	1966 age (time-since-fire) plus 41 (to update to 2007)
Cell Format	Short integer	Short integer	Text, String length is 8	Short integer	Text	Short integer	Short integer
Explanation	Polygons that were along the edge of a map sheet were given new ID values, when they were joined with polygons from an adjacent map sheet.	year the map sheets were created	the map sheets we digitized using GIS	The tree species which make up the individual forest stand. (In groups of 6, the first three characters are letters, the next three characters are numbers) 3-3	The OMNR performed GIS queries to assign these attributes	YRSOURCE minus AGE	Forest stand age (time-since-fire)
Examples	1030	always 1966	always DIGITALA	SB 40PW 20BF 20 SW 100	ConMx, SbSha	1936	44

Clarification				SB40 SB = Black Spruce dominates 40% of the forest stand			

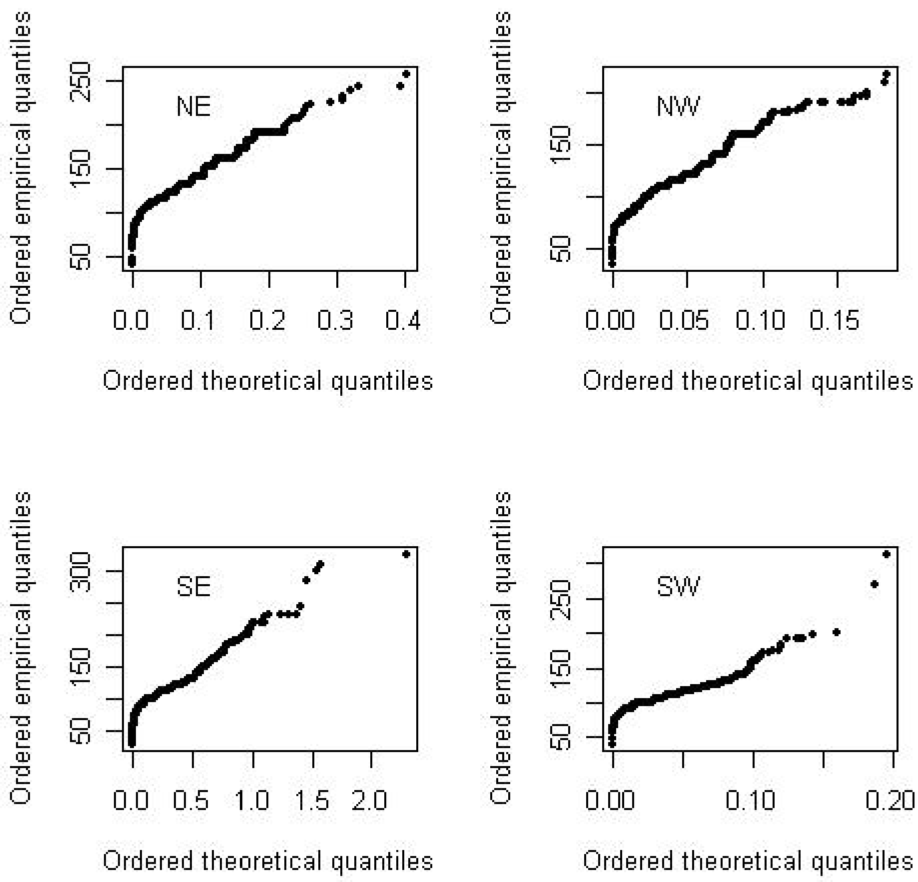
Attributes	**CCLO**	**SITECLASS**	**POLYTYPE**	**FORMOD**	**WG**	**LOGGED**	**AREA1**	**HT**	**UP_LOW**
Definitions	canopy closure	site class	FOR	Protection Forests	First species in SPCOMP string	flagged as logged in harvest ledger	area (ha)	height of tree stand	upland or lowland
Cell Format	Double	Short integer	Text, String length is 3	Text, String length is 2	Text, String length is 1	Short integer length is 1	Double	Short integer	Text
Explanation	Forest stand stocking in percent				PJ = Jack Pine working group SB = Black Spruce working group	1 = logged 0 = not logged	area of 1966 FRI forest stand	Forest stand height in feet	Survival analysis spatial partition. A GIS query was performed using the SPCOMP, SITE CLASS and FU fields (see Methods section)
Examples	0.6, 1.0 etc.	X, 1, 2, 3 or 4	FOR	PF	H, C	1, 0	123.543	45', 75' etc.	U or L
Clarification		In some cases PF forests did not have a value of 4; captured "as is"		If PF not present captured as RP					

Attributes	CRUISED	MapID1	StandNum1	FIREINT1	TSF2	QUADRANT
Definitions	Cruise line	Map ID (OBM ID)	1966 FRI stand number	1st fire interval	second time-since-fire	quadrant of Park
Cell Format	Short integer	Short integer	Short integer	Short integer	Short integer	Text. String length is 2
Explanation	Tagged the tree stands that had been cruised (field inventoried) in this field	The file name of the .pdf FRI map. The OBM identifier.	The 1966 FRI stand number	The time between successive stand-replacing fires (area of over burning)	The digital fire atlas provided this time-since-fire	Survival analysis spatial partition
Examples	1	486912E	13	8	20	NW, NE, SE, SW
Clarification	cruise lines are clustered (near one another)					

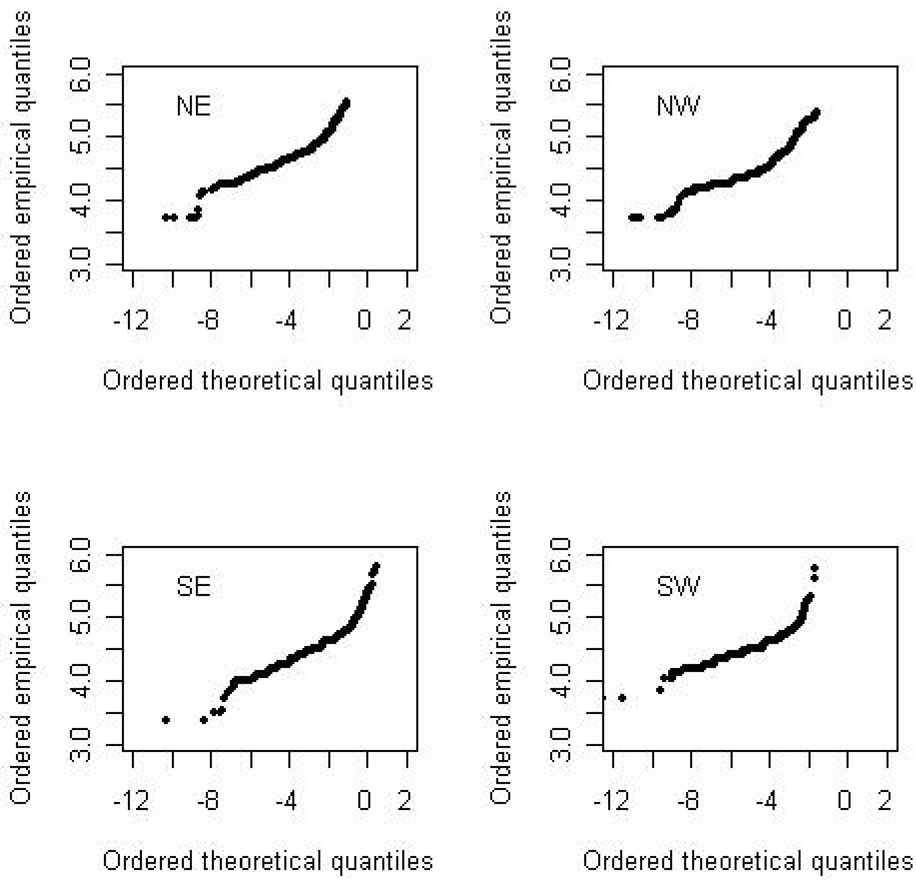
Notes: Once the FRI was updated with the digital fire atlas, there were three time-since-fire (TSF), fire year (FIREYR) and area burned (AREA) fields. Therefore two fire interval (FIREINT) fields were provided. However, not all of these fields are shown in this table. Likewise, there were four MaptID and ForestStandID attribute fields.

Appendix C

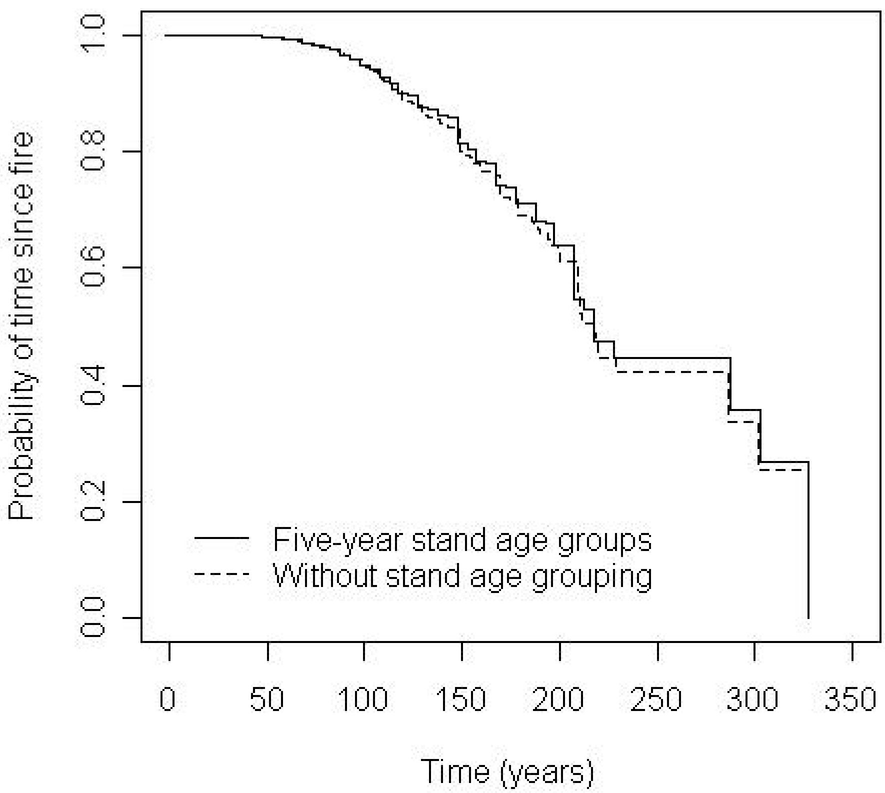
Other Survival Analysis Figures



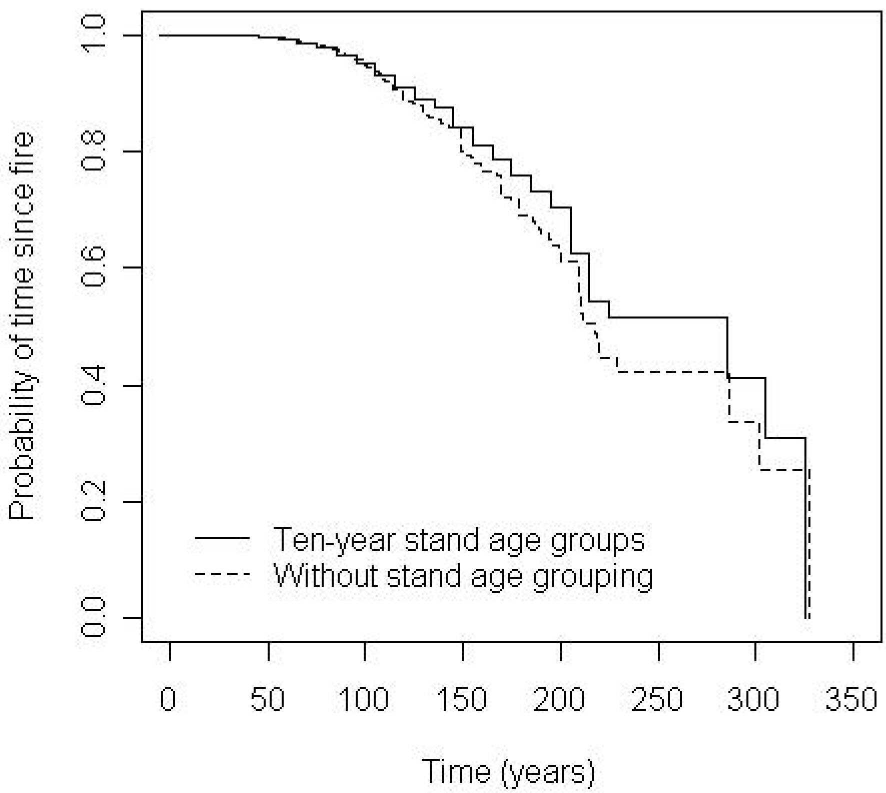
Negative exponential model vs. empirical distribution for four Park quadrants.



Weibull model vs. empirical distribution for four Park quadrants.



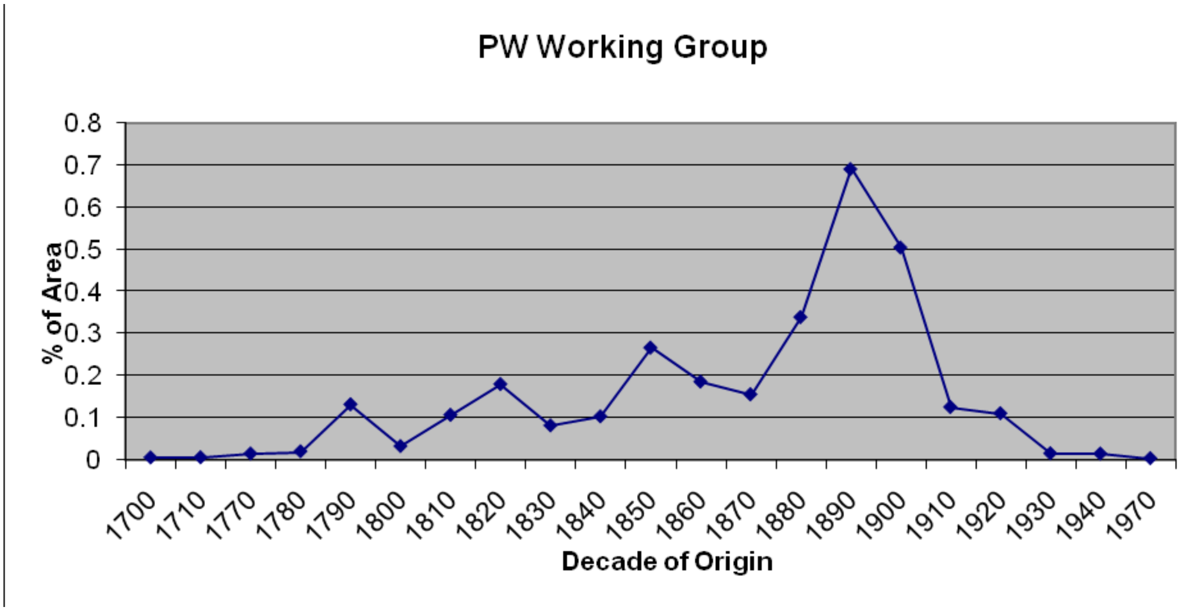
Global survival function (5 year age-class grouping)

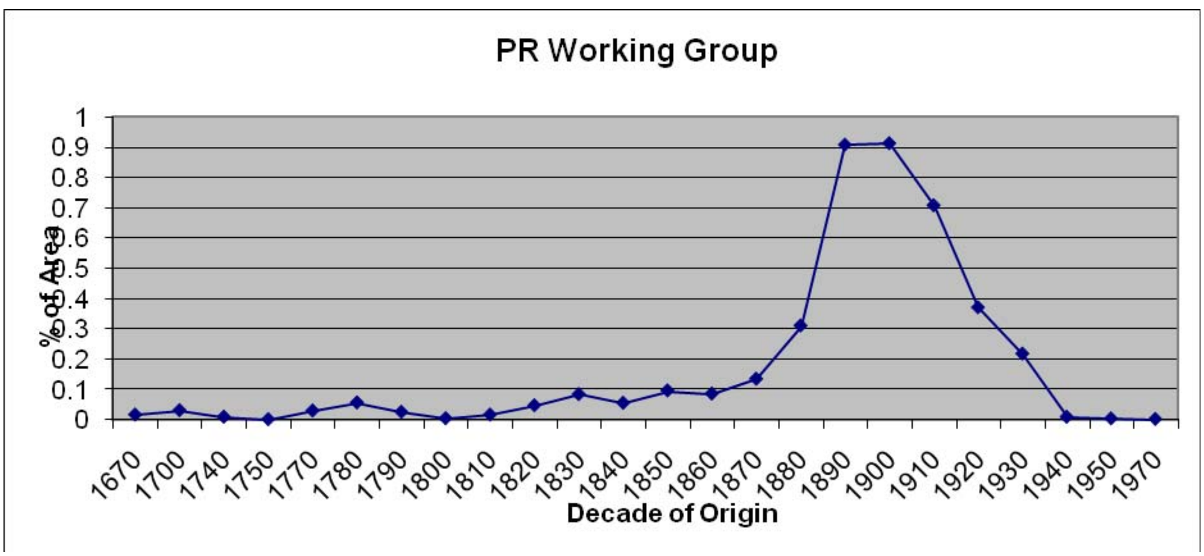


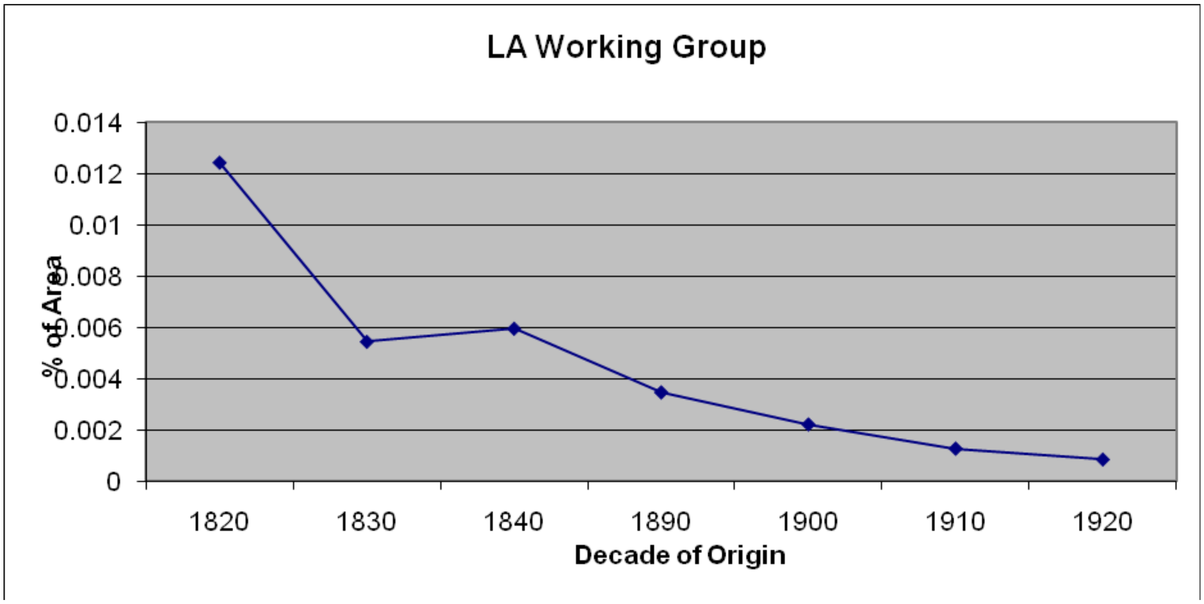
Global survival function (10 year age-class grouping)

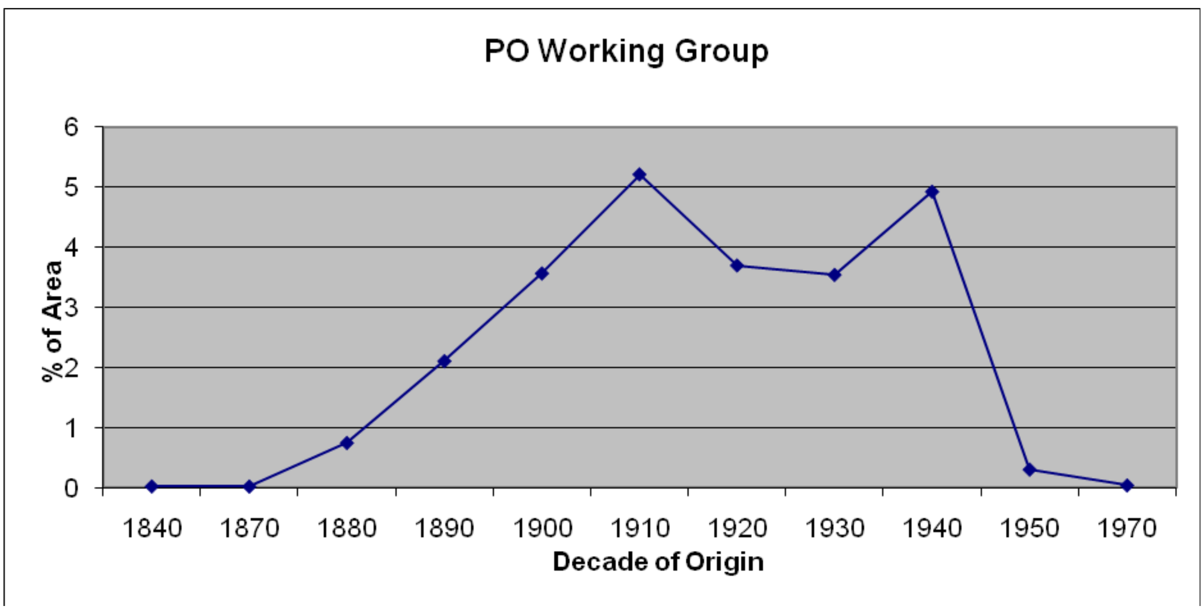
Appendix D

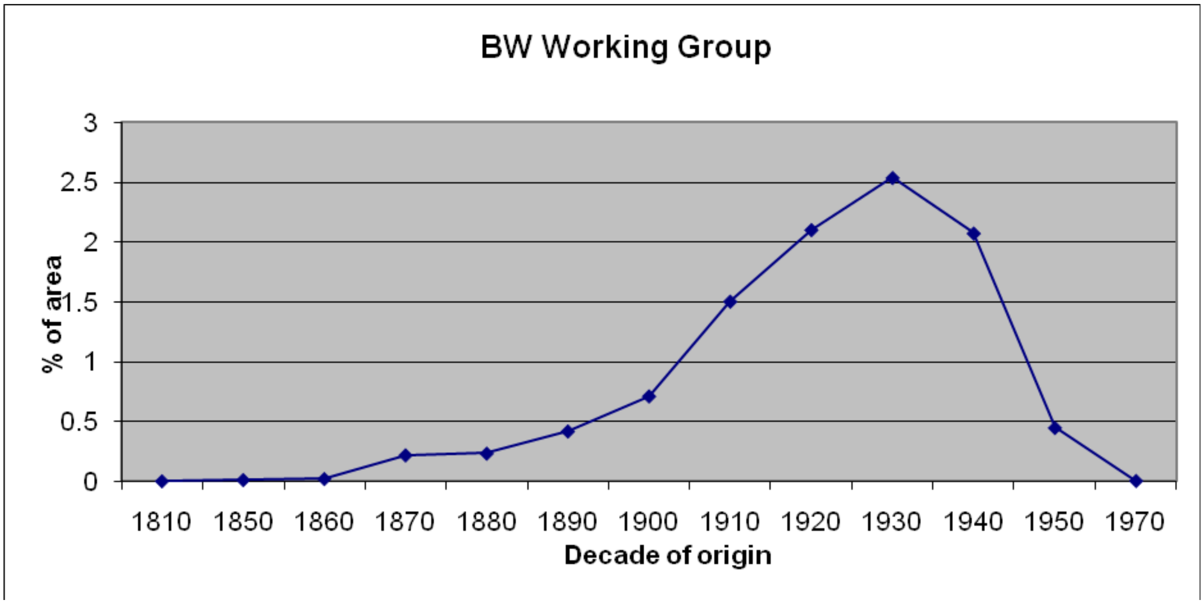
1966 FRI Forest Composition as a Function of Time-since-fire

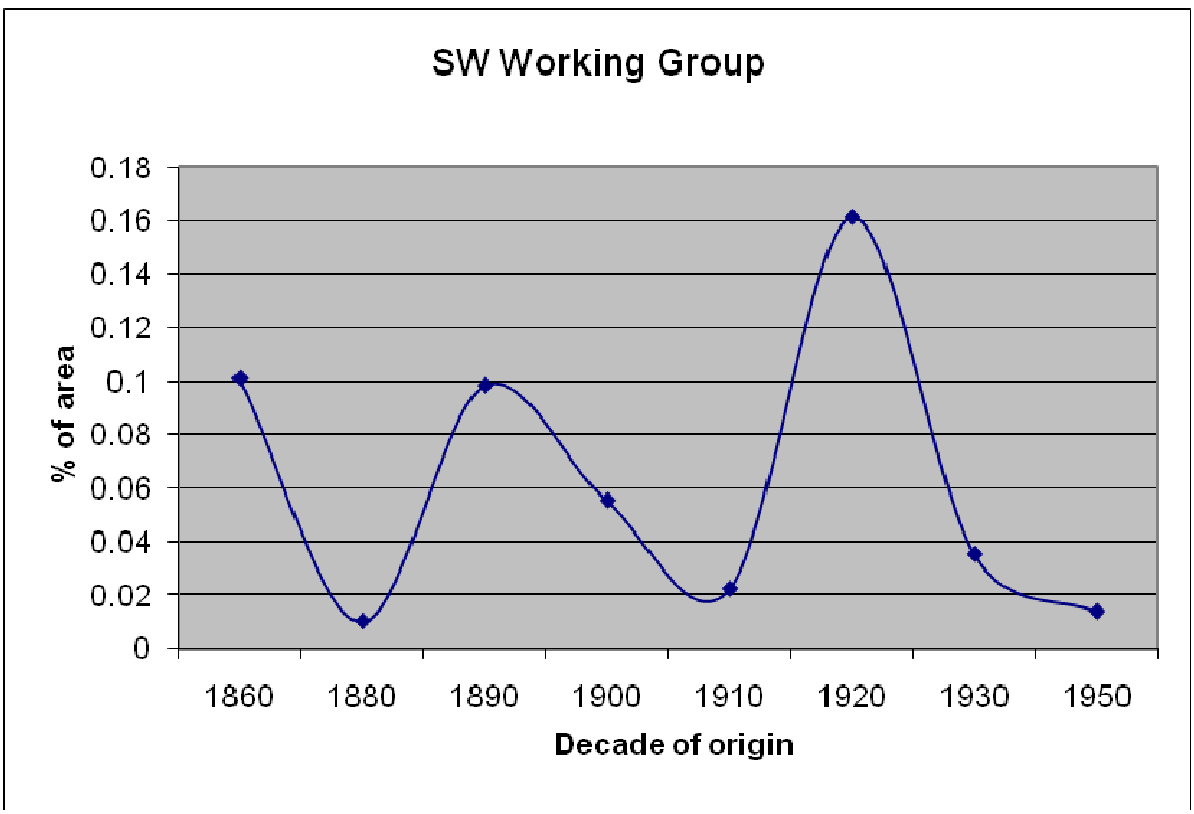


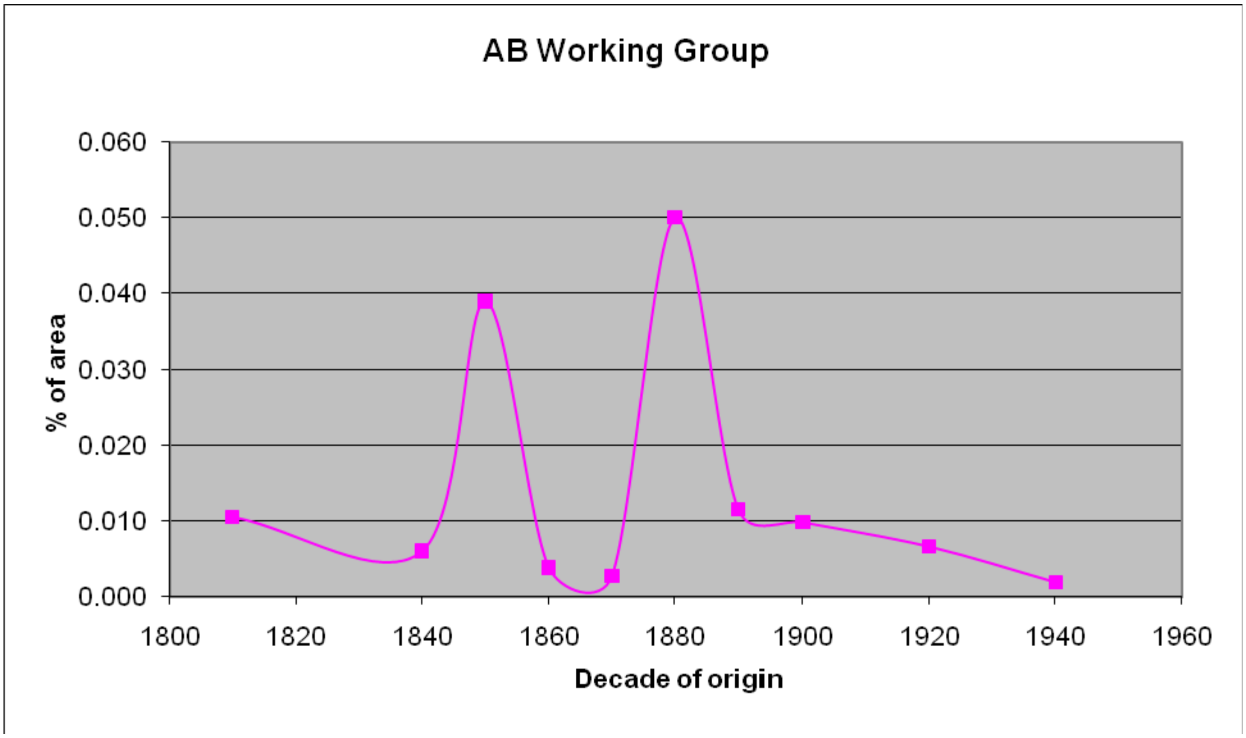


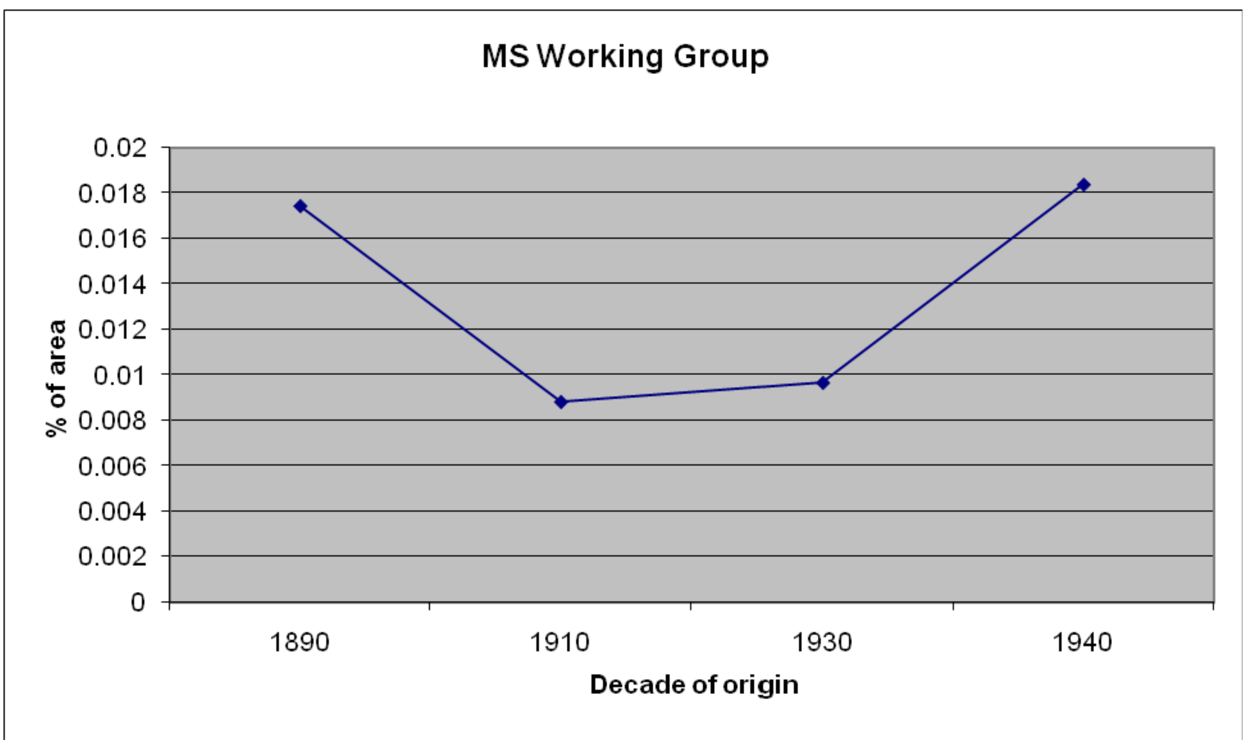

LA Working Group
0.014
0.012
0.01
0.008
0.006
0.004
0.002
0
% of Area
1820
1830
1840
1890
1900
1910
1920
Decade of Origin

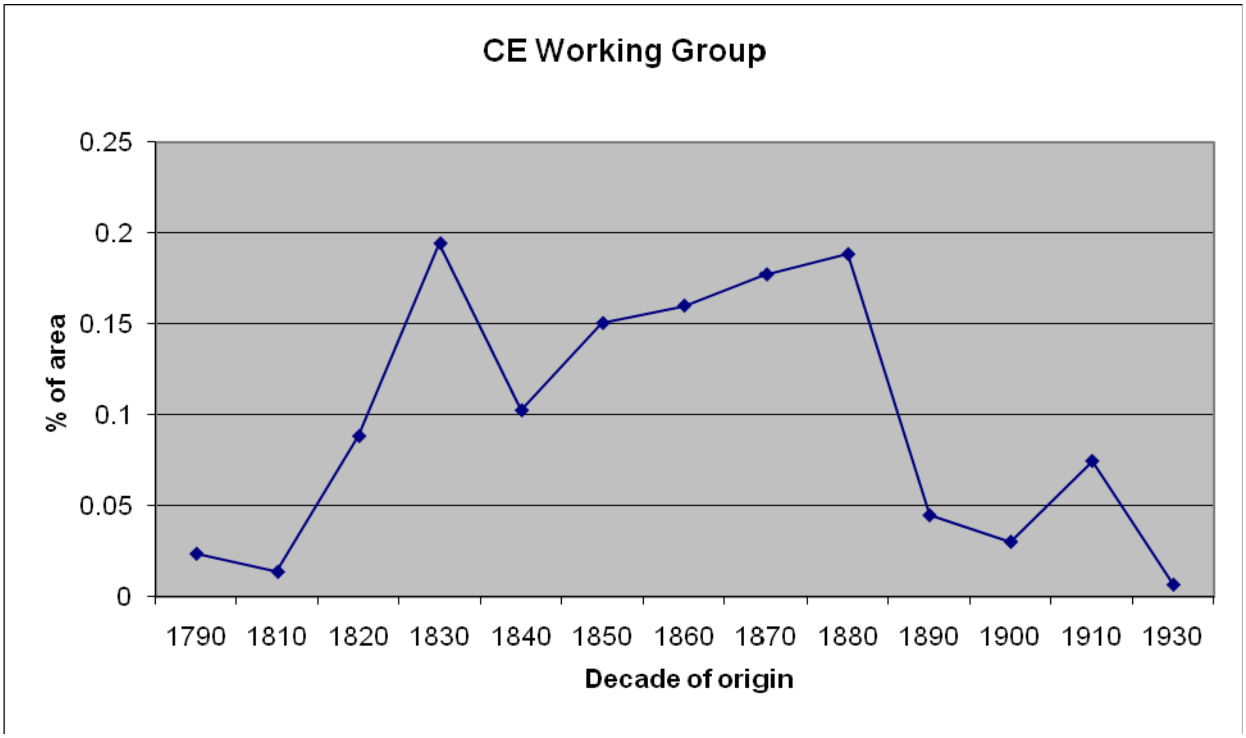

PO Working Group
6
5
4
3
2
1
0
% of Area
1840
1870
1880
1890
1900
1910
1920
1930
1940
1950
1970
Decade of Origin

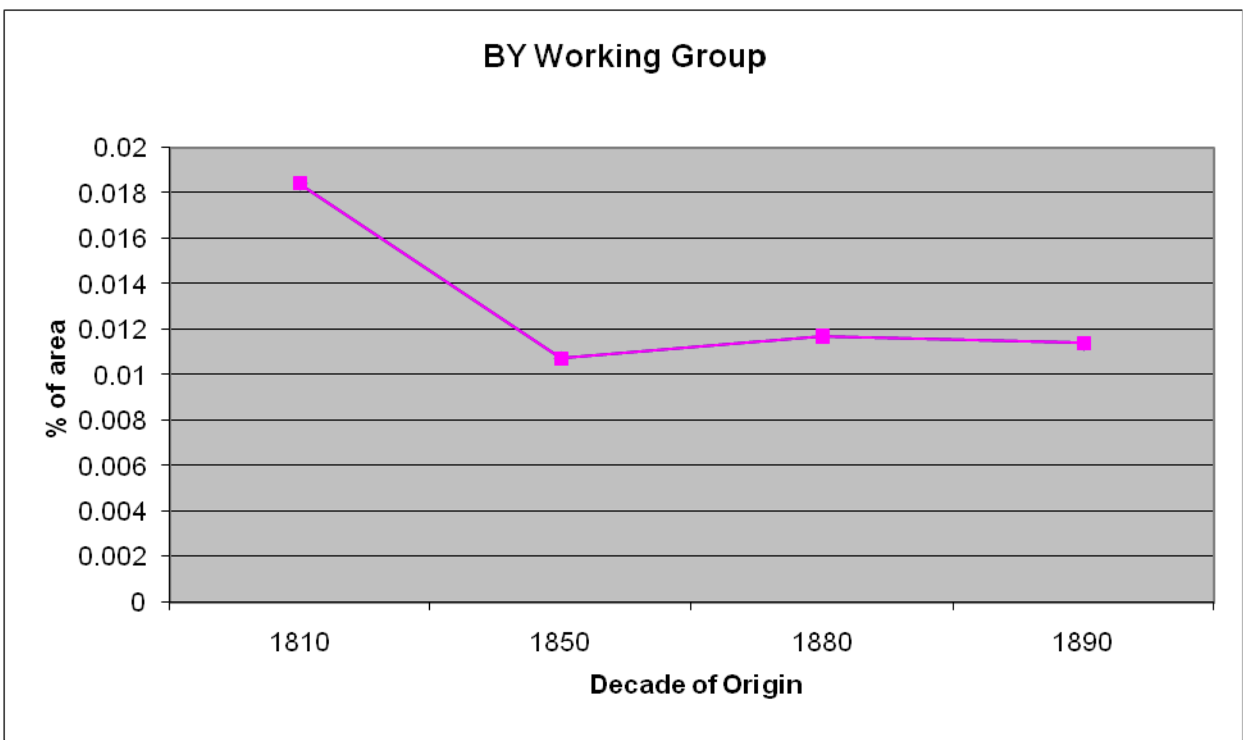

BW Working Group
% of area
3
2.5
2
1.5
1
0.5
0
1810 1850 1860 1870 1880 1890 1900 1910 1920 1930 1940 1950 1970
Decade of origin

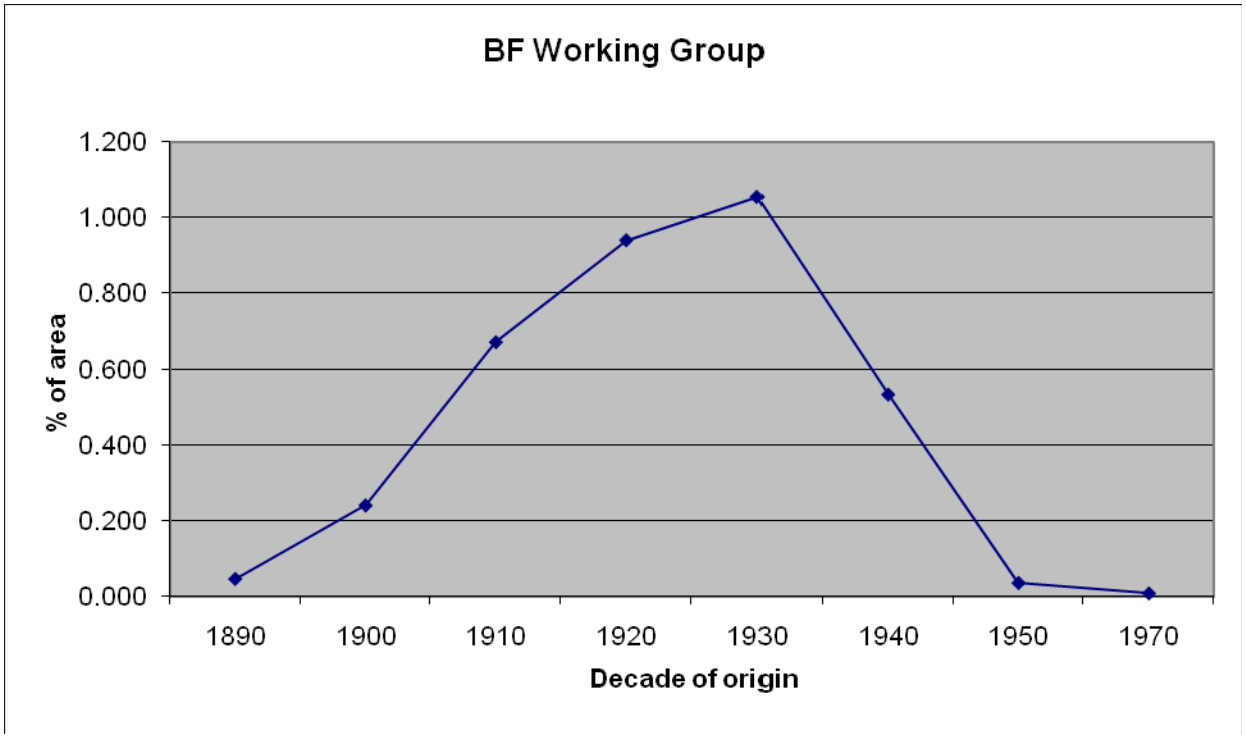

SW Working Group
% of area
0.18
0.16
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
1860 1880 1890 1900 1910 1920 1930 1950
Decade of origin


AB Working Group
0.060
0.050
0.040
0.030
0.020
0.010
0.000
% of area
1800
1820
1840
1860
1880
1900
1920
1940
1960
Decade of origin


MS Working Group
0.02
0.018
0.016
0.014
0.012
0.01
0.008
0.006
0.004
0.002
0
% of area
1890
1910
1930
1940
Decade of origin


CE Working Group
0.25
0.2
0.15
0.1
0.05
0
% of area
1790 1810 1820 1830 1840 1850 1860 1870 1880 1890 1900 1910 1930
Decade of origin


BY Working Group
0.02
0.018
0.016
0.014
0.012
0.01
0.008
0.006
0.004
0.002
0
% of area
1810 1850 1880 1890
Decade of Origin



BF Working Group
% of area
1.200
1.000
0.800
0.600
0.400
0.200
0.000
1890
1900
1910
1920
1930
1940
1950
1970
Decade of origin

Appendix E

Kaplan-Meier Survival Analysis Results

Call: survfit(formula = Surv(time, delta) ~ 1)

time	n.risk	n.event	survival	std.err	lower 95% CI	upper 95% CI
29	7493	2	1.000	0.000189	0.999	1.000
33	7491	2	0.999	0.000267	0.999	1.000
34	7489	1	0.999	0.000298	0.999	1.000
36	7488	2	0.999	0.000353	0.998	1.000
41	7486	3	0.999	0.000422	0.998	0.999
43	7469	1	0.999	0.000442	0.998	0.999
46	7466	4	0.998	0.000517	0.997	0.999
49	7459	1	0.998	0.000534	0.997	0.999
51	7458	1	0.998	0.000550	0.997	0.999
52	7457	1	0.998	0.000566	0.996	0.999
54	7456	8	0.997	0.000680	0.995	0.998
55	7448	1	0.996	0.000693	0.995	0.998
56	7447	5	0.996	0.000755	0.994	0.997
57	7440	1	0.996	0.000766	0.994	0.997
58	7438	1	0.995	0.000778	0.994	0.997
59	7437	20	0.993	0.000979	0.991	0.995
61	7412	2	0.993	0.000997	0.991	0.994
63	7392	2	0.992	0.001015	0.990	0.994
64	7386	2	0.992	0.001032	0.990	0.994
65	7383	4	0.991	0.001066	0.989	0.994
66	7374	17	0.989	0.001199	0.987	0.992
69	7299	24	0.986	0.001367	0.983	0.989
71	7257	6	0.985	0.001405	0.982	0.988
73	7031	1	0.985	0.001412	0.982	0.988
74	7002	13	0.983	0.001498	0.980	0.986
75	6966	1	0.983	0.001504	0.980	0.986

76	6946	7	0.982	0.001549	0.979	0.985
77	6641	3	0.982	0.001569	0.978	0.985
78	6616	1	0.981	0.001576	0.978	0.984
79	6595	18	0.979	0.001693	0.975	0.982
80	6558	1	0.979	0.001700	0.975	0.982
81	6529	8	0.977	0.001750	0.974	0.981
82	6143	1	0.977	0.001756	0.974	0.981
83	6112	2	0.977	0.001770	0.973	0.980
84	6089	10	0.975	0.001839	0.972	0.979
85	6044	1	0.975	0.001845	0.972	0.979
86	6024	7	0.974	0.001892	0.970	0.978
87	5618	3	0.973	0.001915	0.970	0.977
89	5560	45	0.966	0.002231	0.961	0.970
90	5482	2	0.965	0.002244	0.961	0.970
91	5440	2	0.965	0.002257	0.960	0.969
92	4855	3	0.964	0.002282	0.960	0.969
93	4826	3	0.964	0.002306	0.959	0.968
94	4792	23	0.959	0.002489	0.954	0.964
95	4743	3	0.958	0.002512	0.954	0.963
96	4719	1	0.958	0.002519	0.953	0.963
97	4513	5	0.957	0.002561	0.952	0.962
99	4449	40	0.949	0.002877	0.943	0.954
100	4375	5	0.947	0.002914	0.942	0.953
101	4342	5	0.946	0.002951	0.941	0.952
102	3555	1	0.946	0.002962	0.940	0.952
103	3527	2	0.946	0.002985	0.940	0.951
104	3502	18	0.941	0.003182	0.935	0.947
105	3448	12	0.937	0.003308	0.931	0.944
106	3406	4	0.936	0.003350	0.930	0.943
107	3033	3	0.935	0.003389	0.929	0.942
108	2998	1	0.935	0.003402	0.928	0.942

109	2965	38	0.923	0.003874	0.916	0.931
110	2886	4	0.922	0.003922	0.914	0.930
111	2856	2	0.921	0.003945	0.914	0.929
112	2311	7	0.918	0.004072	0.910	0.926
113	2269	4	0.917	0.004144	0.909	0.925
114	2237	22	0.908	0.004528	0.899	0.917
115	2189	1	0.907	0.004544	0.899	0.916
116	2162	2	0.907	0.004579	0.898	0.916
117	1777	3	0.905	0.004656	0.896	0.914
118	1740	2	0.904	0.004708	0.895	0.913
119	1709	26	0.890	0.005353	0.880	0.901
120	1663	1	0.890	0.005377	0.879	0.900
121	1639	1	0.889	0.005401	0.879	0.900
122	1304	2	0.888	0.005478	0.877	0.899
124	1249	8	0.882	0.005800	0.871	0.894
125	1229	1	0.881	0.005840	0.870	0.893
126	1206	1	0.881	0.005880	0.869	0.892
129	1095	21	0.864	0.006826	0.850	0.877
130	1056	2	0.862	0.006910	0.849	0.876
131	1041	1	0.861	0.006953	0.848	0.875
132	813	1	0.860	0.007024	0.847	0.874
133	795	1	0.859	0.007098	0.845	0.873
134	778	1	0.858	0.007175	0.844	0.872
135	769	1	0.857	0.007252	0.843	0.871
136	752	1	0.856	0.007331	0.842	0.870
139	721	7	0.847	0.007904	0.832	0.863
140	703	2	0.845	0.008063	0.829	0.861
141	697	1	0.844	0.008142	0.828	0.860
143	533	1	0.842	0.008279	0.826	0.859
149	512	25	0.801	0.011241	0.779	0.823
150	485	2	0.798	0.011435	0.776	0.821

151	478	2	0.795	0.011628	0.772	0.818
154	403	2	0.791	0.011900	0.768	0.814
155	399	2	0.787	0.012166	0.763	0.811
156	394	3	0.781	0.012555	0.756	0.806
157	383	1	0.779	0.012687	0.754	0.804
159	378	6	0.766	0.013451	0.740	0.793
164	227	1	0.763	0.013809	0.736	0.790
167	212	1	0.759	0.014205	0.732	0.788
169	207	10	0.723	0.017629	0.689	0.758
175	163	1	0.718	0.018070	0.684	0.754
179	158	6	0.691	0.020529	0.652	0.732
186	121	2	0.679	0.021719	0.638	0.723
187	114	1	0.673	0.022332	0.631	0.719
189	110	1	0.667	0.022953	0.624	0.714
190	108	1	0.661	0.023557	0.617	0.709
194	58	1	0.650	0.025762	0.601	0.702
196	55	1	0.638	0.027871	0.586	0.695
199	47	1	0.624	0.030404	0.568	0.687
200	46	1	0.611	0.032633	0.550	0.678
209	35	3	0.558	0.041540	0.483	0.646
210	32	2	0.524	0.045691	0.441	0.621
211	30	1	0.506	0.047384	0.421	0.608
218	28	1	0.488	0.049018	0.401	0.594
219	26	1	0.469	0.050599	0.380	0.580
220	22	1	0.448	0.052603	0.356	0.564
229	17	1	0.422	0.055718	0.325	0.546
286	5	1	0.337	0.087603	0.203	0.561
302	4	1	0.253	0.098227	0.118	0.541
327	1	1	0.000	NA	NA	NA