Pre- and post-European settlement fire history of red pine dominated forest ecosystems of Seney National Wildlife Refuge, Upper Michigan

Igor Drobyshev, P. Charles Goebel, David M. Hix, R. Gregory Corace III, and Marie E. Semko-Duncan

Abstract: To understand the dynamics of fire in red pine (*Pinus resinosa* Ait.) forest ecosystems that once dominated areas of the northern Lake States, we dendrochronologically reconstructed the fire regime prior to European settlement (pre-1860), after European settlement (1860–1935), and postrefuge establishment (post-1935) for different portions (wilderness and nonwilderness) and landforms (sand ridges and outwash channels) of the Seney National Wildlife Refuge (SNWR) in eastern Upper Michigan. Using data from 50 sites, we found that the cumulative number of fires showed a slow rate of accumulation from the 1700s to 1859, a steeper pattern suggesting higher fire occurrence from 1860 to 1935, and a return to fewer fires after 1935. Prior to European settlement, the fire cycle (FC) of sand ridge landforms interspersed within a poorly drained lacustrine plain in the Seney Wilderness Area was 91–144 years. This was longer than on glacial outwash channel landforms (53 years) and on sand ridge landforms interspersed within lacustrine plains located outside of the wilderness (47 years). The FC was also shorter (30 years) during this period and has subsequently increased (149–1090 years) after SNWR establishment. Differences in fire regimes among landform types were minor relative to the temporal variation in fire regimes among the three time periods.

Résumé : Pour comprendre la dynamique des feux dans les écosystèmes forestiers de pin rouge (*Pinus resinosa* Ait.) qui ont jadis dominé des régions du nord des États des Grands Lacs, nous avons reconstitué à l'aide de la dendrochronologie le régime des feux avant la colonisation par les européens (<1860), après la colonisation par les européens (1860–1935) et après l'établissement de refuges (>1935) pour différentes portions (zone protégée ou non protégée) et formes de relief (crêtes sablonneuses et dépôts d'épandage fluvioglaciaire) de la Réserve faunique nationale de Seney située dans l'est du Haut-Michigan. À partir de données provenant de 50 stations, nous avons trouvé que le nombre cumulatif de feux a augmenté lentement dans les années 1700 jusqu'en 1859, plus rapidement de 1860 à 1935 suggérant une occurrence plus élevée des feux et à nouveau plus lentement après 1935. Avant la colonisation par les européens, le cycle de feu dans le cas des crêtes sablonneuses parsemées à travers une plaine lacustre mal drainée dans la zone protégée de la Réserve de Seney était de 91 à 144 ans. Ce cycle était plus long que sur les dépôts d'épandage fluvioglaciaire (53 ans) et que sur les crêtes sablonneuses parsemées dans la plaine lacustre située à l'extérieur de la zone protégée (47 ans). Le cycle de feu était également plus court (30 ans) durant cette période et a par la suite augmenté (149–1090 ans) durant la période qui a suivi l'établissement de la réserve faunique. Les différences entre les régimes des feux parmi les types de formes de relief étaient mineures relativement à la variation temporelle dans les régimes des feux entre les trois périodes de temps.

[Traduit par la Rédaction]

Introduction

Red pine (*Pinus resinosa* Ait.) is one of the few fire-resistant tree species in the eastern United States (Starker 1934), with a range confined to the North American northern forest region and the southern fringe of the North American boreal forest region (Rudolf 1990). In the northern Lake States region of the United States, fire-maintained mixed-pine forests dominated by primarily red pine, but also containing eastern white pine (*Pinus strobus* L.) and jack pine (*Pinus banksiana* Lamb.) were common features of the pre-European settlement landscape (Whitney 1986). During this time, periodic surface fires played an important role in red pine regeneration by providing full sunlight conditions

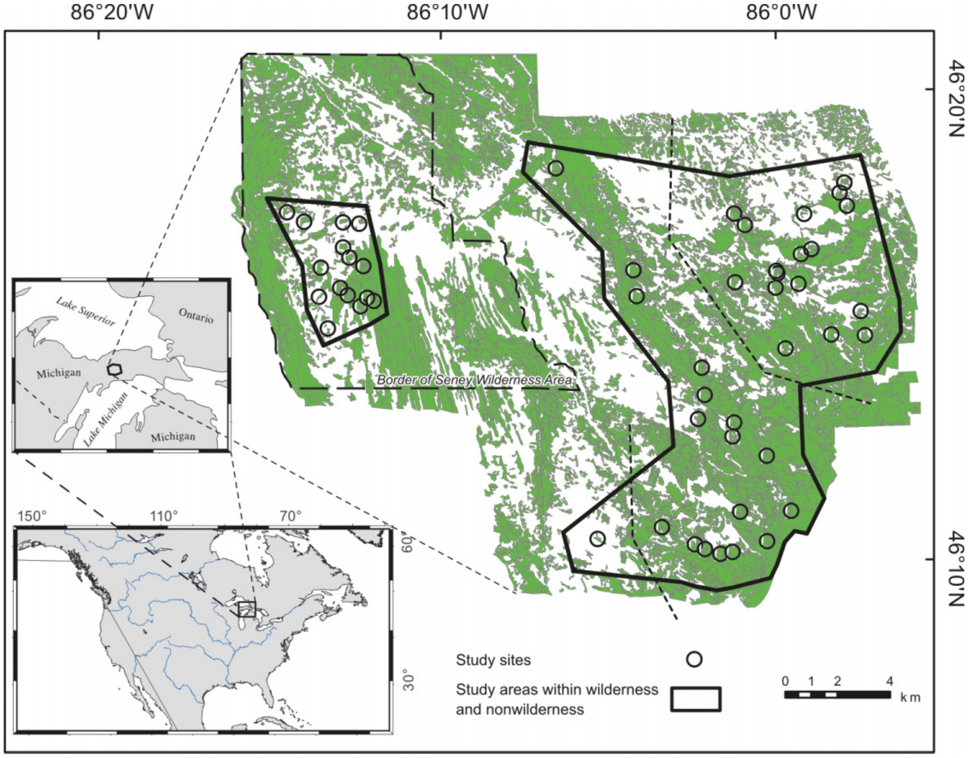
Received 12 October 2007. Accepted 18 June 2008. Published on the NRC Research Press Web site at cjfr.nrc.ca on 27 August 2008.

I. Drobyshev,[1] P.C. Goebel,[2] and M.E. Semko-Duncan. School of Environment and Natural Resources, Ohio Agricultural Research and Development Center, The Ohio State University, 1680 Madison Avenue, Wooster, OH 44691, USA.
D.M. Hix. School of Environment and Natural Resources, The Ohio State University, 2021 Coffey Road, Columbus, OH 43210-1085, USA.
R.G. Corace, III. United States Fish and Wildlife Service, Seney National Wildlife Refuge, 1674 Refuge Entrance Road, Seney, MI 49883, USA.

[1] Present address: Université du Québec en Abitibi-Témiscamingue, 445 boulevard de l'Université, Rouyn-Noranda, QC J9X 5E4, Canada.
[2] Corresponding author (e-mail: goebel.11@osu.edu).

doi:10.1139/X08-082
© 2008 NRC Canada

Fig. 1. Wilderness and nonwilderness portions of Seney National Wildlife Refuge (SNWR) with stand locations (open circles). White areas on the refuge map indicate marshland areas, and darker areas indicate forest vegetation. Broken lines in the lower central part and in the upper right part of the SNWR map delineate the outwash channel (lower part) and sand ridge (upper) landforms of nonwilderness area.



for this shade-intolerant species, reducing competing vegetation, and exposing mineral seedbeds (Ahlgren 1976). However, many of these forests in the northern Lake States have been altered from their pre-European condition, first by extensive logging in the late 19th and early 20th centuries (Karamanski 1989) and, later, by fire suppression policies. Together, these activities have substantially decreased the presence of naturally regenerated red pine stands across the northern Lake States (Cleland et al. 2004).

The legacies of logging and of stand-replacing wildfires in the northern Lake States have also contributed to the subsequent increases in the abundance and distribution of more pyrophytic species such as jack pine and fire-sensitive hardwoods (e.g., aspen, *Populus* spp., and sugar maple, *Acer saccharum* Marsh.; Friedman and Reich 2005), leading to increased fuel loadings and fire hazards relative to a more natural condition (Woodall et al. 2005). These high fuel loadings present a challenge for land managers involved in ecological restoration of these forest ecosystems (Palik and Zasada 2003; Corace et al. 2008) because of potentially high risk associated with using prescribed fire as a restoration tool.

Natural fires have been recognized as the driving factor in the historic development of red pine forests, with surface fires being the most typical disturbance events (Engstrom and Mann 1991; Cleland et al. 2004). However, regional variation in fire regimes has proved to be an important factor in controlling the distribution of red pine across much of its range. While moderate-intensity surface fires are thought to regulate red pine regeneration and establishment across the southern portion of the species' range, Flannigan and Bergeron (1998) suggest that red pine is regulated by large stand-replacing fires across the northern extent of its range, including areas of northern Quebec. As a result, at the northern fringe of its range, red pine forests tend to be spatially associated with habitats associated with nonstand-replacing fires. Therefore, these refugia may function as seed sources of red pine over large landscapes where stand-replacing fires dominate the disturbance regime (Bergeron and Gagnon 1987; Bergeron et al. 1997). There is also a tendency for red pine to be more abundant on xeric sites across its range (Horton and Bedell 1960), likely reflecting the low probability of stand-replacing fires on such sites due to their low fuel loadings.

Both in more northern and central portions of the range of red pine, fire-return intervals (FRIs) are estimated to vary between 3 and 5 years (Bergeron and Gagnon 1987) to >300 years (Heinselman 1973). This variability results from differences in site and climatic conditions, as well as shifts in fire regime due to changes in land-use patterns and, possibly, the influence of changing climate on fire regimes (Flannigan et al. 2001; Bergeron et al. 2004). Therefore, knowledge of the factors controlling fire dynamics at the ecoregional scale are crucial for formulation of sound conservation and restoration strategies for red pine-dominated forest ecosystems.

In this paper, we quantify the characteristics of the pre- and of the post-European settlement fire regimes of the historically red pine-dominated landscape of Seney National Wildlife Refuge (SNWR; Fig. 1). This 38 544 ha area located in eastern Upper Michigan is dominated by two major landform types: glacial outwash channels and sand ridges (“islands;” sensu Heinselman 1965) interspersed in a matrix of patterned or spring bogs located on a lacustrine plain (Heinselman 1965). Additionally, SNWR forests reflect the legacy of natural disturbance regimes prior to European settlement of the area, the extensive logging and high-grading of the forests at the time of the Great Cutover (end of 19th and beginning of 20th century; Dickmann and Leefers 2003) and, finally, management oriented towards a set of multiple objectives, including wildlife habitat, fire protection, recreation, and ecosystem restoration (USFWS 2003; Corace et al. 2008). Our overall hypothesis is that glacial landforms mediated the influence of fire on the SNWR pre-European settlement landscape and that postsettlement harvesting practices and fire suppression activities have changed the pattern of natural fire activity associated with each landform type. Specifically, we (*i*) characterize FRIs, length of fire cycle, and seasonal and spatial patterns of forest fires over a period of 300 years (1707–2006) for the entire SNWR landscape, as well as for the two major landform types; (*ii*) examine the variation in fire regimes across three different time periods (namely, pre-European settlement, pre-1707–1859; European settlement, 1860–1935; and after SNWR establishment, post-1935) that reflects the different natural and anthropogenic disturbance regimes of the region; and (*iii*) discuss possible management implications with respect to the restoration of mixed-pine ecosystems and their fuel management.

Study area

The SNWR lies within the Seney Sand Lake Plain subsubsection of eastern Upper Michigan, where poorly drained landforms of lacustrine origin prevail (Albert 1995). The terrain is characterized by glacial outwash channels and a matrix of patterned fens interspersed by sand ridges (Silbernagel et al. 1997*a*). This matrix was formed by the deposition of Valders glacial outwash, subsequent inundation by earlier stages of the Great Lakes, and drainage of the area during the post-Algonquin period (i.e., 10 000 years B.P.) (Heinselman 1965). Across the Seney Sand Lake Plain, mixed-pine forest ecosystems tend to be limited to both the outwash channel and sand ridge landforms. At SNWR, outwash channels are typically linear landform types that are associated with streams and are approximately 6–8 km long and several hundred meters to 1 km wide. However, sand ridges are smaller features ranging from <0.5 to 5 ha, with most of them being approximately 0.5–2 ha in size and several metres in elevation above the surrounding patterned fen landscape.

The climate of this ecoregion is continental but also strongly influenced by both Lake Superior and Lake Michigan. The long-term (1971–2000) mean annual temperature is 6.2 °C (MRCC 2007), with considerable variation between the coldest (January, long-term mean minimum temperature, –13.6 °C) and the warmest (July, long-term mean maximum temperature, 26.7 °C) months. Total annual precipitation is 781 cm, most of which occurs in the form of rain (mean snowfall: 312 cm). Precipitation tends to peak in July (93.7 cm) and be lowest in February (30.2 cm). The mean length of growing season is 119 days, with evaporation of 64 cm. Mean humidity varies from 50% to 60% from spring through fall. Finally, long-term wind data suggest that during the spring and summer months, the predominant winds are from the southwest and northwest (USFWS 2003).

Sedge- and shrub-dominated ecosystems, along with mixed coniferous–deciduous forests, dominated the pre-European landscape at SNWR (Comer et al. 1995; Zhang et al. 2000). Particularly, many areas of SNWR were historically dominated by forest ecosystems comprised primarily of red pine but with a lesser component of eastern white pine and jack pine (Heinselman 1965; Comer et al. 1995). Presently, SNWR is characterized by a wetland–upland mosaic of forest and non-forest cover types (USFWS 2003; Corace et al. 2008). The wetland vegetation is typically a sedge- and shrub-matrix. Upland areas are dominated by mixed forests with varying proportions of deciduous species (e.g., American beech, *Fagus grandifolia* Ehrh.; sugar maple; and yellow birch, *Betula alleghaniensis* Britt.), as well as several coniferous species (e.g., red pine; eastern white pine; jack pine black spruce, *Picea mariana* (Mill.) BSP; and balsam fir, *Abies balsamea* (L.) Mill).

Prior to European settlement, fires in pine forests of the upper Great Lakes originated from both lightning and anthropogenic sources (Loope and Anderton 1998). However, by 1851, the General Land Office had completed its surveys in eastern Upper Michigan (Barnett 1982), and land sales began shortly thereafter. Europeans moved into the region, large areas were harvested for timber as well as to clear the land for agricultural land uses, and there were subsequent intense slash fires. Maybee (1960) suggests that the approximate onset of logging at SNWR was 1860, focusing first on red and white pine sawtimber. Following this exploitive period, many areas were drained and cleared for agriculture; however, poor soil quality and the large-scale intense forest fires fueled by logging debris thwarted these efforts (Losey 2003). This period lasted until the early 1930s, and much of this land was abandoned and tax-reverted to the State of Michigan. In 1935, SNWR was established and forest management practices have been undertaken to foster wildlife and waterfowl management.

Methods

Field data collection

Field sampling was designed to allow reconstruction of the fire histories of the two major landform types of the SNWR landscape: (*i*) sand ridges interspersed within patterned fens and (*ii*) glacial outwash channels. The majority of the former type was located within the federally designated Seney Wilderness Area (hereafter wilderness), a 10 178 ha portion of the 38 544 ha SNWR, where remnant old-growth red pine forest ecosystems remain and where wildfires have occurred within the last 30 years. This area is one of the best remaining tracts of old-growth red pine forest ecosystems in the region. In the red pine dominated sand ridges of the wilderness, the extensive wetland shrub matrix precluded logging because access for horses or machinery was difficult. Conversely, the terrain of the nonwilderness (i.e., level outwash channels) did not constrain logging to the same degree as in the wilderness, and records of timber harvesting exist for this part of SNWR since 1935. Our fire history sampling took place in the areas with differ-

© 2008 NRC Canada

ent levels of human impact ("low" in the wilderness vs. "high" in the nonwilderness) and different landform types. In searching for stands to be sampled, we avoided sites where the absence of fire-scarred trees, logs, and stumps precluded fire history reconstruction (Swetnam and Baisan 1996).

Nested within the wilderness, the area sampled included the most remote cluster of sand ridges that comprise portions of the Strangmoor Bog National Natural Landmark and its associated research natural area. We sampled all major sand ridges that rose above the patterned fen matrix by >2 m, as well as a number of randomly selected smaller sand ridges. We sampled the small sand ridges (0.2–2 ha) to increase the detection probability for small ($\sim$10 ha) fires (as explained later) and to obtain a dataset representative of the whole population of sand ridges in the wilderness. In the nonwilderness (14 220 ha), sampling was constrained by the low number of fire-scarred trees, stumps and snags, some of which had been removed or destroyed during past logging operations. Finally, because we were interested in the legacy of past harvesting activities, we searched for sites that represented different combinations of vegetative cover and the recent harvesting history (over the last 70–100 years) across the area.

In each sampled site, we searched for fire-scarred live trees, stumps, and deadwood and used the method of "wedge sampling" (Swetnam 1996) to extract a wedge from the bole of trees containing scars. Trees representing the oldest cohorts within the site were typically sampled to recover scars not visible from outside of the trees (so-called overhealed scars). At each site, we searched over an area up to 1 ha (or smaller in the islands with the total area <1 ha) and also limited our maximum search time for potential samples to 2.5 h to ensure that our sampling effort did not lead to differences in the probability of detecting past fires among sites. However, as the size of study sites did not exceed 1 ha, 2.5 h was ample time to explore the site, identify all fire-scarred trees, and then select the presumably most informative live or dead trees with respect to fire history reconstruction. This approach, which is similar to that of Swetnam and Baisan (2003), allowed us to focus on the trees (often the oldest trees within stand), snags, and stumps, that exhibited multiple noneroded fire scars at each site so as to develop the longest possible stand fire chronology. As the red pine dominated forest ecosystems are distributed along a narrow, xeric environmental gradient in this landscape, the likelihood that there were differences in the preservation times of deadwood, which may influence estimates of fire frequency, across our sampled sites was low.

In the majority of cases (97%), the samples were collected from red pine; other tree species from which samples were taken included jack pine and eastern white pine. In the field, we recorded tree diameter at 1.37 m above the ground (hereafter DBH), species, type of sample (e.g., living tree, stump, snag, or log), and azimuth of the scar. The last parameter was analyzed to infer the direction of the fire which produced the first scar of that sample. Due to the temperature pattern around a tree during a fire, fire scars typically form on the side of the tree 180° to the direction of fire (Dickinson and Johnson 2001), given similar fuel conditions around the base of the tree.

Sample preparation and dating

Samples were mounted on wood and progressively polished with up to 400 grit sandpaper to allow clear recognition of annual rings and fire scars under a binocular microscope (using up to 40× magnification). All samples were crossdated using a visual approach (following Stokes and Smiley 1968), which included skeleton plots for each sample. We developed local chronologies for sites at SNWR, and the master chronology (also referred to as a pointer-year chronology; Schweingruber et al. 1990), utilized ring widths, earlywood and latewood widths, and latewood and earlywood densities. The resulting pointer-year chronology provided a basis for, and was used for, accurately aging each sample. Additionally, we used a number of existing red pine chronologies, which were helpful in verification of our pointer year chronology, that were available for the region from the International Tree-Ring Data Bank (www.ncdc.noaa.gov/paleo/treering.html). Such an approach has been used successfully in several North American and European studies of fire history (e.g., Caprio and Swetnam 1995; Niklasson and Granström 2000; Brown 2006). After cross-dating of tree rings was completed on a fire-scarred sample, calendar dates and seasonal information were obtained for all fire scars. Finally, after all samples were cross-dated and aged, it became possible to use known fire dates from the studied stands (and especially the dates of large fires in the area such as the Seney Fire of 1976; Anderson 1982) as an additional step to verify crossdating accuracy.

Seasonal dating of fires was accomplished through identification of the position of each fire scar within an annual ring. We classified fires into three groups: (*i*) earlywood fires, those fire scars located in dormant earlywood to late earlywood; (*ii*) latewood fires, those fire scars located in early latewood to dormant latewood; and (*iii*) fires with unclear seasonal dating.

Methods of fire history reconstruction

To deduce the size of past fires, we mapped and analyzed the locations of recording sites for each fire year reconstructed. We considered recording sites as those sites that included a fire chronology covering a fire year in question, either with a fire dated in a year (based on a specific fire scar associated with a tree ring) or no history of fire during that year (based on the lack of a fire scar associated with a tree ring which indicates a fire-free year for that site). A site was considered nonrecording for a year in question if any of the wedge samples collected on that site were missing a ring from the year in question. In identifying the periods when a site was not a recording site, we considered not only susceptible trees (e.g., already scarred) but all trees irrespective of whether they were scarred or not. During spatial delineation, we also used topographic and vegetation maps to identify natural fire breaks (e.g., streams, local depressions in the terrain, and shrub-dominated areas) that were considered as suggested borders of the fires between two recording sites with and without fire recorded for the year in question. In this process, we adopted a conservative approach to determining fire history and considered any of the above features of the terrain as a potential fire break. During delineation, we also utilized seasonal information on fire occurrence to

Table 1. Detection probabilities for fires of different size classes within wilderness and nonwilderness portions of Seney National Wildlife Refuge during three time periods.

	Wilderness			Nonwilderness		
Fire size class (ha)	1707–1859	1860–1935	1936–2006	1707–1859	1860–1935	1936–2006
10	0.43	0.43	0.43	0.03	0.04	0.04
100	1	1	1	0.26	0.35	0.37
200	1	1	1	0.37	0.48	0.5
300	1	1	1	0.6	0.73	0.75
400	1	1	1	0.71	0.83	0.84
500	1	1	1	0.79	0.89	0.9
600	1	1	1	0.84	0.93	0.94
700	1	1	1	0.88	0.95	0.96
800	1	1	1	0.91	0.97	0.97
900	1	1	1	0.94	0.98	0.98
1000	1	1	1	0.95	0.99	0.99
1100	1	1	1	0.97	0.99	0.99
1200	1	1	1	0.98	0.99	1

Note: Fires >1200 ha in size had detection probability of 1 in both areas and for all time periods. Fire size column refers to the upper limit of the respective size class; the center of each class was used for calculation of the detection probability (e.g., 250 ha for 200–300 ha size class).

separate different fires occurring in the same calendar year. We chose not to calculate fire areas as a simple function of proportion of the number of sites recording fire to the total number of sites. This calculation approach would assume equal probability for fire spread across the landscape, which would not be expected in our study landscape as many pine-dominated ecosystems are located on landforms surrounded by extensive wetlands.

To adjust for the variation in detection probability of fires of different sizes and across periods with different sampling coverage, we calculated the probability of a fire of a certain size to be recorded given a certain number of recording sites. Therefore, probability of a fire of a size A to go undetected can be written as $e^{-\lambda A}$, and the detection probability for a fire of size A is $1 - e^{-\lambda A}$, where λ is the number of stands covering that year divided by the study area (Niklasson and Granström 2000). Detection probabilities for fires within 11 fire size classes (Table 1) were calculated for each calendar year over the span of our data set (1707–2006). In these calculations, we used the midpoint of the fire size interval (e.g., 5 ha for the first fire size interval: 0–10 ha). Prior to these calculations, we reduced the total area comprising the wilderness and the nonwilderness portions of our study landscape by subtracting the amount of nonforest vegetation (marshland) and open water. By doing so, we assumed a negligible probability of fire initiated in these areas. However, within areas covered by forest vegetation, we assumed randomness of both fire occurrence and spatial distribution of study sites. Consequently, our estimates of the total area of the wilderness and the nonwilderness portions of SNWR were 610 ha and 9510 ha, respectively. To test for differences in the central tendency and shape of the fire-size distributions between the wilderness and the nonwilderness, we used a nonparametric Kolmogorov–Smirnov test, a goodness-of-fit test used to determine whether two underlying one-dimensional probability distributions differ (Sokal and Rolf 1995). To correct for decreasing detection ability with decreasing fire size, we used the frequency distribution of fires within each of the three periods divided by the detection probability for that fire size class. For example, the frequency of fires within the 100–200 ha size class for the 1860–1935 was divided by 0.48, the period-specific detection probability for that fire size class.

Finally, at each site, one to three 50 m × 10 m plots were established as a part of an accompanying study on stand composition and fuel loadings (Drobyshev et al. 2008), and time since the last stand-replacing event was estimated for each site through aging of the three dominant cohorts of red pine. In the field, we visually separated different cohorts on the basis of tree DBH and height and cored several overstory trees per site at the height of 10–30 cm to obtain cohort origination dates. To filter out possible effects of timber harvesting on stand age structure, we report cohort origination data only for the wilderness portion of SNWR.

Analysis of FRIs and site survivorship functions

It has been shown that the distribution of FRIs; (the mean number of years between successive fires) for a single site often may be represented by the Weibull distribution (Grissino-Mayer 1999). We tested the FRIs for goodness-of-fit with respect to the Weibull distribution using the Hollander–Proschan test, using both uncensored and censored observations (Dodson 1994). In the context of our analyses, uncensored FRIs were those between two fires dated on a site; censored FRIs were intervals between date of a fire event and a nonfire date (e.g., the first year covered by fire chronology of that stand or the year of sampling). Differences in FRIs between both periods and areas were checked with the Cox–Mantel test (StatSoft Inc. 2001). It has been previously shown that this test is more powerful than other alternatives for comparison of survivorship functions drawn from populations that follow Weibull or exponential distributions (Lee et al. 1975).

To assess site survivorship function, describing the probability for a site to escape fire during a given time interval, we used Kaplan–Meier estimator (Kaplan and Meier 1958):

$$S(t) = \prod_{j=1}^{t} \left(\frac{n-j}{n-j+1}\right)^{\delta_{(j)}}$$

where $S(t)$ is the site survivorship function estimated for a time t, n is the total number of observations, Π is the geometric sum across all cases $\leq t$, and j is a dummy variable that is either 1 if the jth case is uncensored (complete) or 0 if it is censored (incomplete). For the analysis of temporal variation in fire statistics we examined three time periods and analyzed properties of the fire regime separately for each period including: pre-European settlement (pre-1860), European settlement (1860–1935), and post-SNWR establishment (post-1935).

Calculation of fire cycles

Fire cycle (FC; also known as fire rotation) is the length of time (years) required to burn over an area equal to the study area (Van Wagner 1978). We calculated FC as

$$\mathrm{FC} = \frac{T\,\mathrm{Area}_{\mathrm{study}}}{\mathrm{Area}_{\mathrm{burned}}}$$

where T is the length of the time period under consideration (years) and $\mathrm{Area}_{\mathrm{study}}$ and $\mathrm{Area}_{\mathrm{burned}}$ are the study area and the area burned over this time period (hectares), respectively. In these calculations, we adjusted the area of each fire by dividing its size by the detection probability of a fire of that size category. In this way, we compensated for the decline in our ability to detect smaller fires. We then obtained 10% and 90% confidence intervals for the FC distributions by the bootstrap method (Efron and Tibshirani 1994). During bootstrapping, the original FC distribution was randomly resampled with replacement 1000 times, and confidence limits were calculated on the generated distributions of FC.

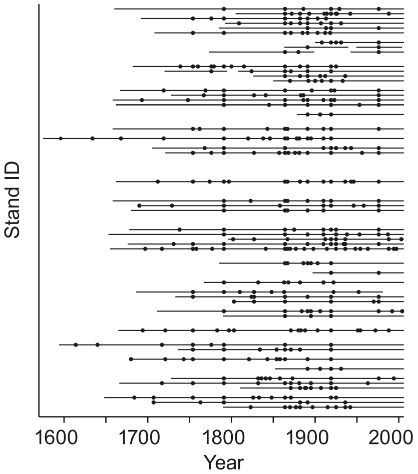
Results

Detection probability of past fires

Detection probabilities varied greatly across time periods and fire size classes (Table 1). For the wilderness, detection probability was 0.43 for fires <10 ha and 1.00 for fires >10 ha for all three time periods (Table 1). For the wilderness, no correction was needed to adjust the number either of fires detected or for the area burned. In the case of the first size class (0–10 ha), we assumed detection probability of the next fire size class. We adopted this conservative approach, which may lead to underestimation of the number of small fires, because of the lack of information on the distribution of very small fires. For the nonwilderness, the number of sites covering a particular year varied from a mean of 22.6 stands for the period from 1707 to 1859 to 33.6 for the period from 1936 to 2006.

The number of recording sites (sites recording fire or recording "no fire" in a year in question) varied between 24 (1707) and 49 (since 1970), with the mean and mode being 44.1 and 47, respectively. Because of this variation, raw data might have increasingly underestimated the number of fires at the start of the study period (1707). We adjusted the number of fires by calculating the inverse ratio of the total number of sites (49) to the number of active sites for each year. This coefficient, ranging from 2.4 (in 1707) to 1.0 (1970–2006), was multiplied by the actual number of fires recorded during an individual year.

Fig. 2. Time span of each of the 49 fire history sites within SNWR. Each chronology is based on a mean of five samples. Fires are recorded as "points" and blank spaces indicate a hiatus in a chronology (i.e., stand was not recording during that period).

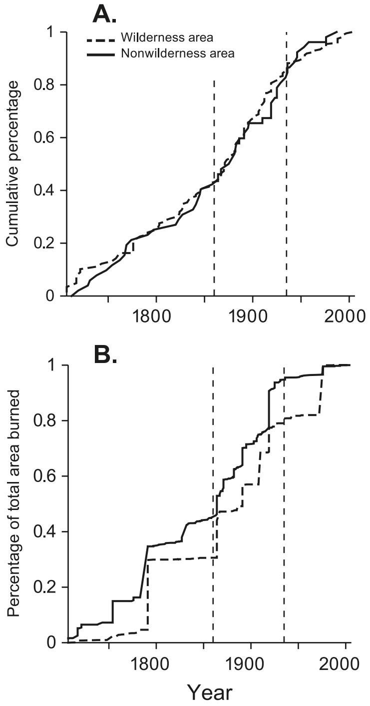


General dating and fire history information

In total, 97 wedge samples were collected in the wilderness; the mean, minimum, and maximum number of samples per island were 6.5, 1, and 12, respectively. Among the nonwilderness sites, the total number of samples was 151. The mean number of samples per nonwilderness site was 4.6 samples, with the minimum and maximum values being 1 and 12, respectively.

We dated 686 fire scars corresponding to a total of 410 site-level fire events. The earliest fire in the SNWR landscape based on our data occurred in the late growing season of 1596. During the 1600s, 17 fires were recorded. However, sampling coverage for the time period prior to 1707 was low: on average, there were 7 sites (minimum 2 and maximum 23) and a total of 56 samples (mean 17 samples per year) from 1596 to 1706 (Fig. 2). Spatial analyses of fire occurrence across the landscape and results of delineation of fire areas documented 217 individual fires from 1707 to 2006. Over this period, the slope of the cumulative number of fires first increased slowly in the 1700s and the first half of 1800s, followed a steeper increase in the second half of 1800s and early 1900s, and finally leveled off in the second half of the 1900s (Fig. 3A). The cumulative area burned revealed a generally similar temporal pattern (Fig. 3B), disrupted by single large fires. We identified seven major fire years for the entire SWNR, defined as years where the area burned exceeded 20% of the study area (either wilderness or nonwilderness). The major fire years were 1754, 1791, 1864, 1891, 1910, 1919, and 1976. Of these, 1910 and 1976 were recorded as major fire years only within the wilderness. The year 1754 was a major fire year only in the nonwilderness portion of SNWR. The FRIs for these larger fires averaged 37 years, with a range of 9–73 years. Shorter FRIs corresponded to the period before

Fig. 3. Cumulative percentage of the number of fires in the wilderness (15.5 km^2) and nonwilderness (153.5 km^2) portions of SNWR (A) and cumulative area burned as a percentage of the total area burned in these two subareas (B) during 1707–2006. The broken vertical lines are the boundaries (1860 and 1935) of three time periods identified for the statistical analyses.



SNWR establishment and the Great Cutover (Dickmann and Leefers 2003).

Small (<100 ha) fires dominated the fire regime in both the wilderness and nonwilderness portions of SNWR over the entire period studied (1707–2006) and within each of the time periods (Fig. 4). Large fires (>1100 ha) were the main contributors to the total area burned, with the exception of 1860–1935 period in the wilderness where several fires with sizes ranging from 400 to 900 ha burned most of the area. A similar pattern was also observed in the 1936–2006 period in the nonwilderness, where fires <500 ha equaled more than one-half of the area burned during that period.

Aging of the dominant cohorts indicated that most of the sampled red pine associated with these sites at SNWR originated from two periods: 1870–1890 and 1920–1930 (Fig. 5). Age of the most abundant cohort within a site was 169 ± 97.1 years (mean ± SD) in the wilderness and 150 ± 75 years in the nonwilderness with age ranges of 31–356 years and 23–325 years for overstory red pine in the wilderness and nonwilderness, respectively.

Wilderness versus nonwilderness comparison

The FRIs closely fit Weibull distributions, with the exception being the nonwilderness in the most recent period (i.e., after SNWR establishment; 1936–2006). The FRI showed wide variation across sites and time periods (Fig. 6A). For example, in the wilderness mean FRI was 28 years for the whole study period (1707–2006) with considerable variation observed in among periods (15–33 years). Moreover, within each of the three time periods, FRI varied greatly, although rarely exceeding 80 years (Table 2, Fig. 6). The variation in uncensored FRI was highest in the pre-European settlement time period (1707–1859) compared with the European settlement period (1860–1935) in both wilderness and nonwilderness, as indicated by the higher values of the scale parameter of the respective Weibull distributions: 74.17 and 50.64 versus 20.82 and 20.62 for wilderness and nonwilderness, respectively (Table 2). For both areas, we may expect even higher variation in the uncensored FRIs in the future because the majority of most recent intervals for the study area were apparently exceeding FRI ranges of the previous intervals and were not complete (censored) at the time of sampling (summer 2006). Corresponding with this observation, scale parameters were generally much higher for FRI distributions in the most recent period. Except for the after SNWR establishment period in the nonwilderness area, the shape parameter of Weibull distribution was between 1.06 and 1.46, indicating a moderately increasing probability over time for a site to experience a fire.

In the population of FRIs, we observed both short and long intervals. Overall, 9.5% of all uncensored intervals in the wilderness (12 intervals) and 4.9% (15 intervals) in nonwilderness were <5 years, indicating that relatively short intervals were not extremely rare at SNWR. However, we also recorded censored intervals that were >100 years: they comprised 3% (five intervals) of the FRI in the Wilderness and 0.3% (one interval) in the non-Wilderness (Fig. 6A).

Changes in FRIs were tested for significance in pairwise comparisons of respective site survivorship functions, representing probability for a site to escape a fire event. The FRI changes were compared with estimated site survivorship functions. In between-area comparisons, significantly longer FRIs were found only in the pre-European settlement time period (1707–1859) with no differences observed in the two other periods and for the whole study period (Table 3). During this same time period (1707–1859), the survivorship function revealed about a 50% chance for a site to escape fire after 50 years of fire-free period in the Wilderness, with this percentage declining just below 40% after 80 years. For the non-Wilderness portion of SNWR, corresponding analysis suggested a more active fire regime with half of the sites being burned after only 40 years and 20% of sites escaping fire for 80 years.

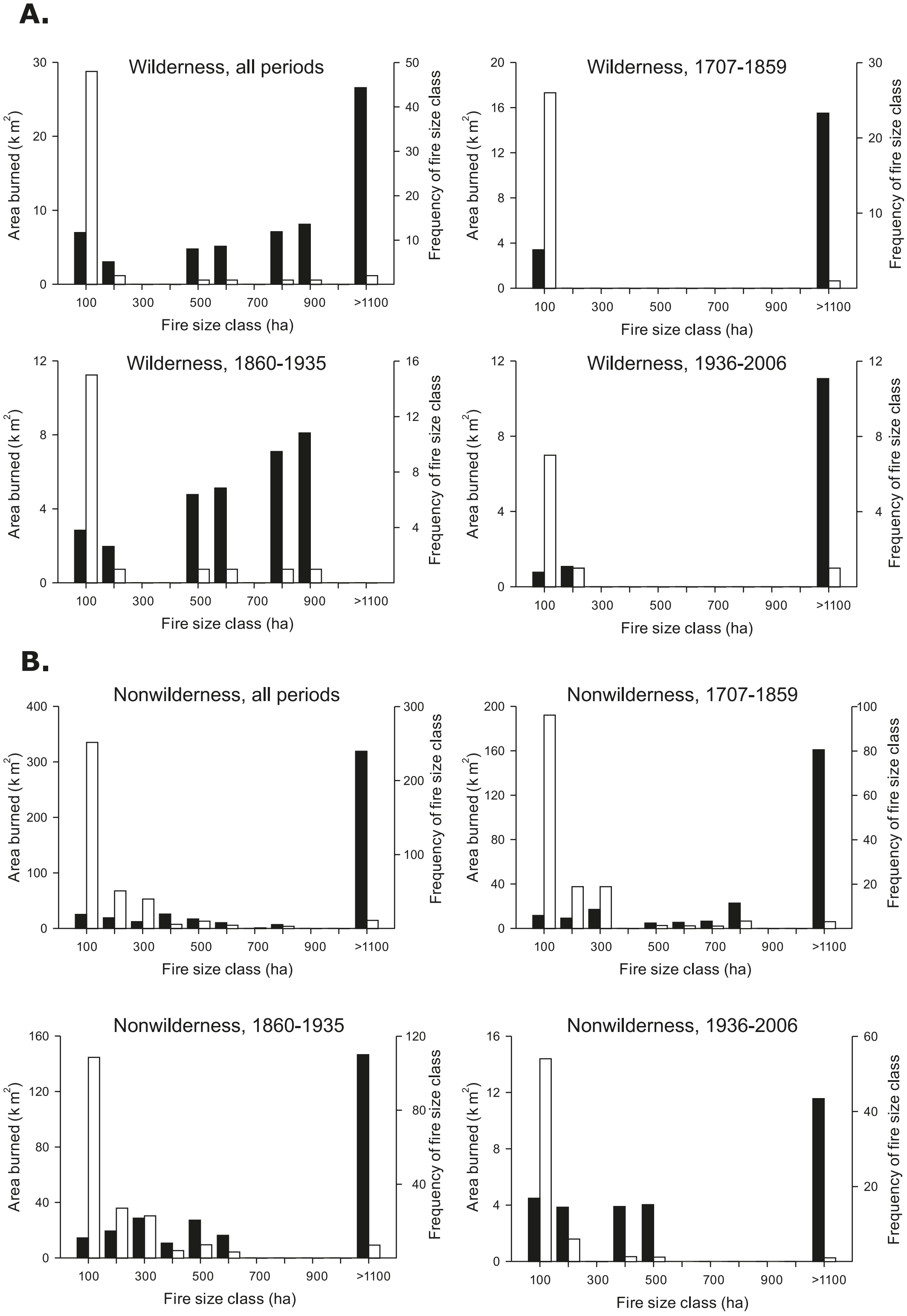
In contrast with between-area comparisons, most of the between-period comparisons showed highly significant differences (Table 3). The FRI was shortest during the European settlement period in both areas and site survivorship functions indicated that sites had a <10% chance of escaping fire after 40 years without fire in the wilderness and after 50 years in the nonwilderness. A completely different pattern was observed in the most recent period: even after 80 years since the last fire, there was a >60% probability for a site to remain fire free in both the wilderness and nonwilderness portions of SNWR.

Sand ridge versus outwash channel comparison

On both landform types, FRIs followed Weibull distributions for the first two time periods (Fig. 6B, Table 2). In the pre-European settlement time period, mean values of the FRI on the two landform types were similar (25 years on

© 2008 NRC Canada

Fig. 4. Distribution of fires among 11 sizes classes (open bars) and relative contribution of each fire size class to the total area burned (solid bars) in wilderness (A) and nonwilderness (B) portions of SNWR for three time periods. Note the different scales on the *y* axes. The largest fire size class (>1100 ha) also included fires which exceeded the size of the area sampled.

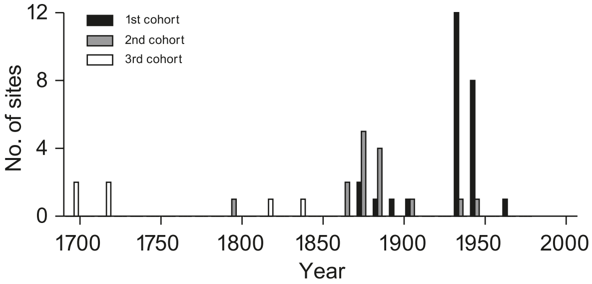


sand ridges and 23 years on outwash channels). Both the mean and variation of FRI decreased from the pre-European to the European settlement period, as indicated by decreasing values of Weibull scale parameter (Table 2).

Comparison of site survivorship functions between wilderness and nonwilderness (Table 3, Fig. 7) sites showed a significant difference for the pre-European period only, with sites in the wilderness having a higher probability of

© 2008 NRC Canada

Fig. 5. Age structure of sites in the Seney Wilderness Area. The first, second, and third cohorts refer to three of the most abundant cohorts recorded within a site. A single site might have one to three cohorts identified.



escaping fire than stands in nonwilderness. For example, a stand had a 37% probability of escaping fire after 80 years without fire in the wilderness and only a 20% probability in the nonwilderness.

Temporal changes in site survivorship function were similar in wilderness and nonwilderness (Table 3, Fig. 7). In the pre-European settlement time period, the probability of a site escaping a fire for >80 years was approximately 20%–30%. In the European settlement time period, sites in the wilderness and nonwilderness portions of SNWR had only a 10% chance of escaping fire after 40 years without fire. In the period after SNWR establishment, the probability of escaping fire increased to 60% after 80 years without fire.

Comparisons of site survivorship between landform types (Table 3, Fig. 8) showed a significant difference between sand ridges and outwash channels within the nonwilderness only during the European settlement time period. During that time period, sites on sand ridges had a <20% chance of escaping a fire after 46 years without fire. The corresponding time for the sites on outwash channels was significantly shorter (18 years). In the period after SWNR establishment, a decline in fire activity resulted in a 60% chance for a site to escape fire after 80 fire-free years on sand ridges, with a corresponding value for outwash channels even higher (80%; Fig. 8).

On the sand ridges, FRI declined from the pre-European time period to the postsettlement time period and then showed a tendency to increase ($p = 0.11$; Table 3, Fig. 8). Similarly, FRI on outwash channels showed a tendency to shorten from the pre-European period to the European period ($p = 0.07$) and then increased significantly in the latest period ($p < 0.0001$). As in the case with the between-area comparison (previous section), the variation between time periods was generally more pronounced than between landform types (Table 3).

Fire areas and fire cycle

In the wilderness, the size distribution of fires differed between the pre-European settlement and the European settlement time periods (Kolmogorov–Smirnov test, $p = 0.025$), primarily because of more frequent large fires in the European settlement time period (Fig. 4). No difference was found in the period after SNWR establishment and the two previous periods (in both cases, Kolmogorov–Smirnov test, $p > 0.100$). In the nonwilderness portion of SNWR, no difference in fire size distribution was found between the pre-European settlement and the European settlement time periods ($p > 0.100$). However, the fire size distribution during the period after SNWR establishment was significantly different from the previous two time periods ($p = 0.005$ for both pairs of contrasts), as a result of smaller fires during 1936–2006. In the nonwilderness, fires were significantly larger in outwash channels as compared with the area dominated by sand ridges (Kolmogorov–Smirnov tests, all three pairs of contrasts $p < 0.050$).

During the pre-European settlement period, the FC was longer in the wilderness than in the nonwilderness (means 144 vs. 47 years, respectively). There was no difference in the European settlement time period (means 20 vs. 19 years). The period after SNWR establishment showed the opposite pattern with a shorter fire cycle in the wilderness than in the nonwilderness (148 vs. 196 years).

Within both the outwash channel and sand ridge landform types, the FC decreased from the pre-European settlement time period (53 and 91 years, respectively) to the European settlement time period (30 and 22 years, respectively) and then dramatically increased in the most recent time period (179 and 1090 years, respectively). All of the above-mentioned differences in the bootstrapped FC distributions were significant (Kolmogorov–Smirnov tests, $p < 0.001$). Across all contrasts, the bootstrap-assessed variation in FC was high (Table 4), which was due to the corresponding variation in the fire sizes and, specifically, the combination of many small fires and few large fires (Fig. 4).

Seasonality and direction of fires

Our observations of earlywood and latewood development in red pine at SNWR indicated that the transition from earlywood to latewood typically occurs in mid-July. Seasonal distributions of fires revealed a moderate prevalence of late-season fires; 78 (46.7%) of all fires dated with seasonal resolution were early season fires, and 89 (53.3%) were late-season fires. This pattern was similar in both the wilderness and nonwilderness portions of SNWR. In the wilderness, we recorded 17 early season fires (42.5%) and 23 late-season fires (57.5%). In the nonwilderness, we recorded 61 early season fires (48.0%) and 66 late-season fires (52.0%). A χ^2 test revealed no significant differences in seasonal distribution of fires between wilderness and nonwilderness for each of the three time periods (Yates corrected $\chi^2 = 0.000–0.020$, corresponding p values = 0.988–0.877).

We did not detect any significant differences between any two time periods within both the wilderness ($\chi^2 = 0.00–0.020$ and corresponding p values 0.892–0.901) or nonwilderness ($\chi^2 = 0.530–2.230$, $p = 0.135–0.467$). Large fires (>100 ha in the wilderness and >1000 ha in the nonwilderness) were almost exclusively late-season fires (Fig. 9). Finally, a total of 115 samples with direction of first scar identified were analyzed to deduce direction of fire movement. Overall, fires tended to move from southwest to northeast direction (Fig. 10). This pattern also held for single large fire years.

Discussion

General pattern of fire activity over 300 years

The historical (1707–2006) fire regime of SNWR was

© 2008 NRC Canada

Fig. 6. Distribution of fire-return intervals of SNWR between the wilderness and nonwilderness (A) and outwash channel versus sand ridge landforms (B). Note the different scales on the *y* axes. Solid and open bars are uncensored and censored observations, respectively (see text for explanation).

Table 2. Statistical descriptors of fire-return interval distributions of SNWR.

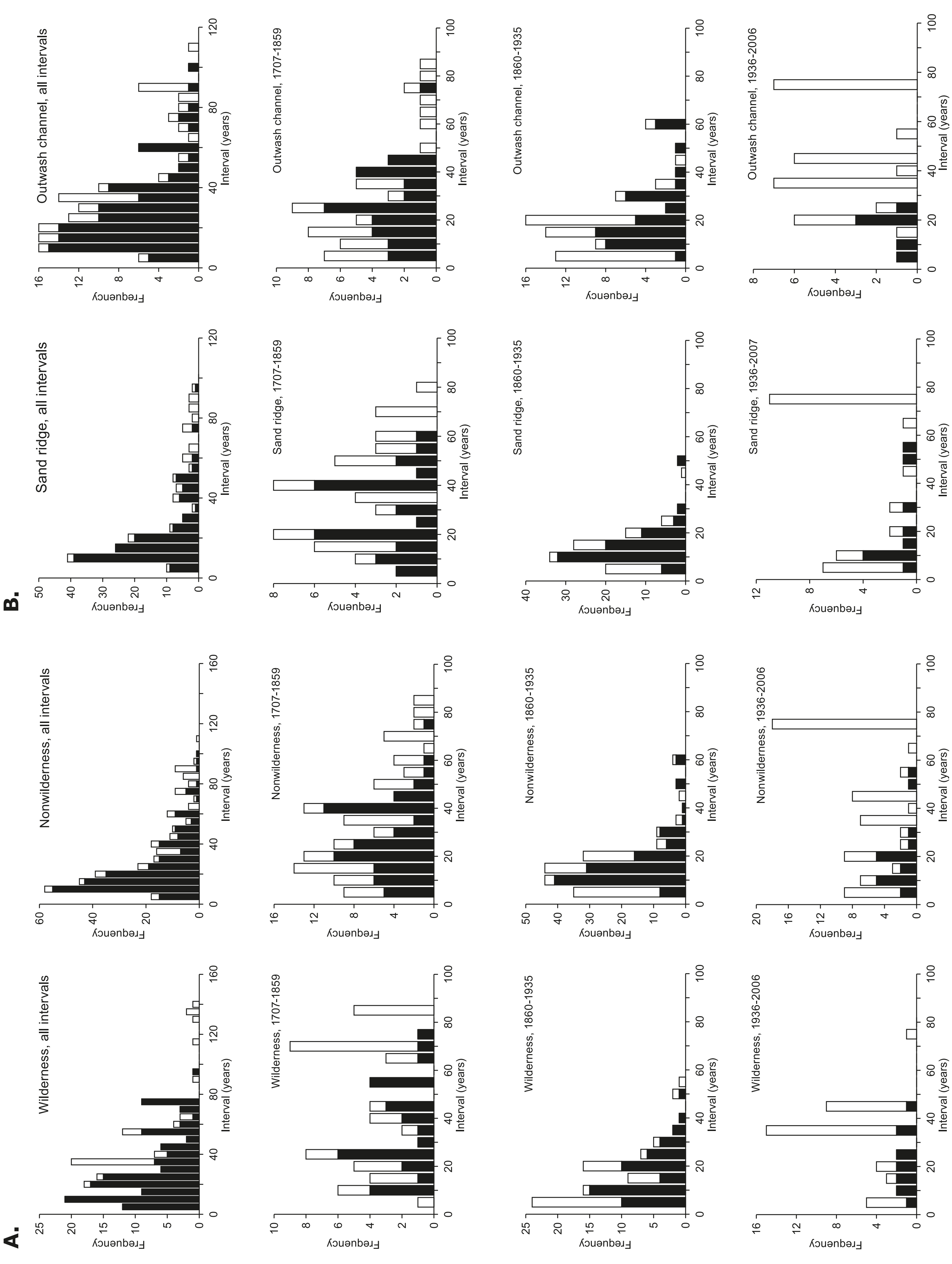
				Weibull distribution			
	Mean ± SD	Median	Range	Shape	Scale	Goodness-of-fit *p*	*n*
Wilderness							
1707–1859	32.7±19.2	29.0	5–72 (2–84)	1.36	74.17	0.223	57 (30)
1860–1935	14.5±10.6	10.0	3–46 (1–50)	1.46	20.82	0.980	83 (30)
1936–2006	18.5±11.5	16.5	3–40 (1–71)	1.24	72.77	0.601	41 (29)
All periods	27.9±21.6	22.0	2–90 (2–138)	1.12	40.89	0.793	155 (29)
Nonwilderness							
1707–1859	24.3±15.1	22.0	3–70 (2–82)	1.28	50.64	0.577	113 (52)
1860–1935	14.1±11.2	11.0	1–59 (1–59)	1.45	20.62	0.610	186 (68)
1936–2006	16.1±13.5	13.5	3–52 (1–71)	0.78	182.30	<0.001	70 (52)
All periods	21.7±18.4	15.0	1–96 (1–106)	1.06	35.63	0.423	310 (67)
Sand ridges							
1707–1859	25.4±15.6	22.0	3–56 (3–75)	1.34	54.47	0.578	52 (25)
1860–1935	11.5±8.1	9.5	1–48 (1–48)	1.59	15.75	0.545	108 (32)
1935–2006	18.0±17.5	10.5	3–52 (1–71)	0.74	151.46	0.002	33 (23)
All periods	18.6±16.5	12.0	1–91 (1–93)	0.99	31.27	0.421	164 (31)
Outwash channels							
1707–1859	23.4±14.7	22.0	3–70 (2–82)	1.31	42.87	0.746	59 (25)
1860–1935	19.7±14.9	15.0	4–59 (3–59)	1.56	30.04	0.795	71 (34)
1936–2006	13.5±7.1	17.0	3–20 (3–71)	0.77	319.24	<0.001	33 (27)
All periods	25.7±19.7	21.0	3–96 (2–106)	1.18	41.29	0.706	135 (34)

Note: Goodness-of-fit for two-parameter Weibull distributions were tested with the Hollander–Proschan test. Mean, standard deviation (SD), and median are given for uncensored observations. Range and *n* values are given for both uncensored and censored (in parentheses) observations. Distribution of fire-return intervals for sand ridges and for outwash channels are shown for nonwilderness only; there were no areas of outwash channel sampled in the wilderness.

characterized by large variation across three different time periods and two different landform types. Frequent small fires and less frequent, regular large fires were the main components of the pre-European settlement fire regime. The fact that most of the earlier large fire years were recorded simultaneously across most of SNWR indicated that total amount of area burned during those years might exceed the total area of SNWR. Therefore, these fires could be described as functioning at the landscape scale rather than the stand scale. Mean FRIs across all periods and parts of the Seney landscape varied between 11.5 and 32.7 years, with corresponding estimates of FC revealing even larger variation in the past fire activity (20–1090 years). Both very short (<5 years) and relatively long (>100 years) FRIs were features of the fire regime, with the shortest intervals as little as 1 year. Besides a human-mediated increase in ignition frequencies, a very short FRI could result from the immediate or delayed mortality of canopy and understory trees, which might result in an increase of fuel loading immediately or few years after a fire. Similarly, very short FRIs (3–5 years) were reconstructed in a red pine forest of northern Vermont during the late 1800s (Engstrom and Mann 1991). However, not all FRIs were short: almost 3% of the FRIs recorded in the wilderness were >100 years. This result suggests that these stands may function as fire-free refugia, which may have an important role in shaping forest composition (Asselin et al. 2001) by providing seed sources to recently burned areas.

Our results suggest that the majority of fires reconstructed at SWNR were nonstand-replacing events. A large percentage of studied sites had several overstory cohorts of red pine (75% in our study), multiple scars commonly found on red pine trees, and mean age of red pine cohorts exceeding the mean FRI for any part of SNWR landscape; all these factors support this conclusion. Therefore, estimation of stand-replacing fires carries an additional uncertainty associated with selecting the dominant cohort within a stand. Aggregated data on cohort ages over the wilderness indicate that it experienced two main cohort-initiation periods around 1870–1890 and 1920–1930. These time periods generally coincided with large fire years of 1864 and 1919. Assuming no human impact on stands in the wilderness, it gives an interval of approximately 60 years between the two events. It is worth noting that only a few cohort-initiation events were recorded during 1700s and the first half of 1800s, which suggests considerable variation in FRI for such events. Nevertheless, our estimate corresponds with a return interval of 68 years for stand-replacing fires reported for red pine forests of northwestern Quebec (Bergeron and Brisson 1990). Compared with less severe surface fires, the return interval of stand-replacing fires is most likely driven by large-scale anomalies in atmospheric circulation patterns (Le Goff et al. 2007).

Although the SNWR landscape of red pine dominated stands within a matrix of patterned fens may not be typical

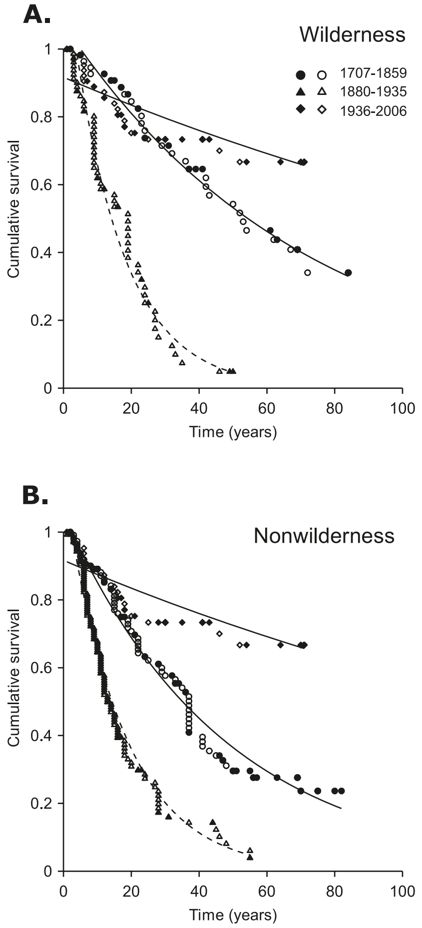


© 2008 NRC Canada

Table 3. Tests for differences in estimated survivorship functions between the two areas (wilderness vs. nonwilderness) of SNWR during three time periods, between different time periods (<1860, 1860–1935, and ≥1936) within areas, between two landform types (sand ridges and outwash channels) within the nonwilderness, and between time periods within landforms.

Contrasts	Cox–Mantel statistic	*p*
Between-area comparisons		
1707–1859 (period 1)	–2.00	0.046
1860–1935 (period 2)	–0.08	0.940
1936–2006 (period 3)	0.58	0.563
All periods combined	–0.97	0.334
Between-period comparisons (areas)		
Wilderness: period 1 vs. 2	6.38	<0.0001
Wilderness: period 2 vs. 3	–7.42	<0.0001
Wilderness: period 1 vs. 3	0.18	0.854
Nonwilderness: period 1 vs. 2	6.49	<0.0001
Nonwilderness: period 2 vs. 3	–7.40	<0.0001
Nonwilderness: period 1 vs. 3	–3.60	<0.001
Between-landform comparison		
1707–1859 (period 1)	1.23	0.217
1860–1935 (period 2)	–4.89	<0.0001
1936–2006 (period 3)	–1.44	0.149
All periods combined	–2.16	0.031
Between-period comparisons (landforms)		
Ridges: period 1 vs. 2	6.80	<0.0001
Ridges: period 2 vs. 3	–5.50	<0.0001
Ridges: period 1 vs. 3	–1.63	0.108
Channel: period 1 vs. 2	1.83	0.067
Channel: period 2 vs. 3	–4.79	<0.0001
Channel: period 1 vs. 3	–3.77	<0.001

Fig. 7. Estimated survivorship curves for the wilderness (A) and nonwilderness (B) portions of SNWR. The curves show the proportion of an area predicted to remain unburned over time and are exponentially fit to the plotted survivorship times. Solid and open symbols are complete and censored observations, respectively. Statistical details of the analyses are given in Table 3.



of the northern Lake States, there are physiographic analogies to the SNWR mixed-pine forest, including larch (*Larix laricina* (Du Roi) K. Koch) woodlands occupying string fens in northern Quebec (Busque and Arseneault 2005). Although low-intensity fires were frequent in these communities, no stand-replacing fire has occurred based on fire reconstruction studies over the last 350 years. Populations of both larch and red pine have been shown to decline where severe stand-replacing fires dominate the fire regime (Bergeron and Brisson 1990), suggesting that their life strategies may be adapted to habitats with low-intensity fire regimes.

Biological legacies and fire regimes

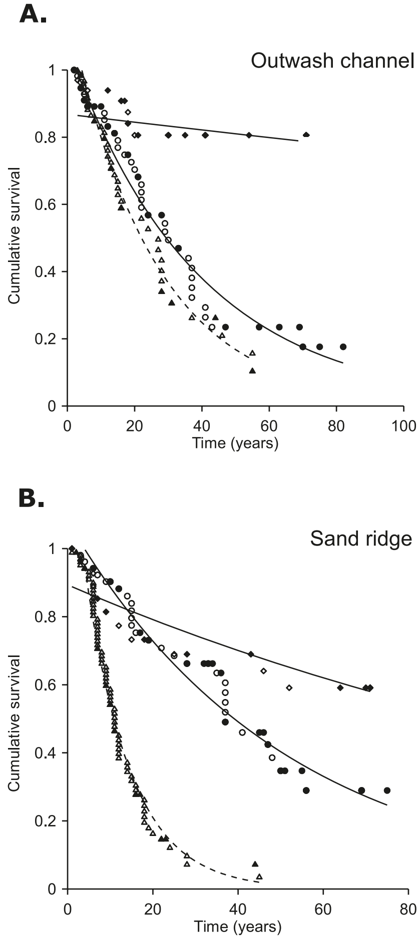
For the most part, the fire history over the last 300 years at SNWR was characterized by moderate fire activity in the pre-European settlement time period (likely fueled by the exploitive timber harvesting), increased fire activity during the European settlement time period, and finally diverging patterns in wilderness and nonwilderness areas after the establishment of the SNWR. Below, we review variation in FRI and FC of these three periods.

For the pre-European settlement period, FRIs varied between 24.3 and 32.7 years. Our estimates were close to the mean FRI interval for surface fires at the northern limit of red pine distribution in Quebec, reported to be approximately 30 years (Bergeron and Brisson 1990). The mean FC on sand ridge landforms both in the wilderness and nonwilderness portions of SNWR was 91–144 years, which was longer than the FC of the outwash channel landforms (53 years) or, in the larger nonwilderness, encompassing both outwash channel and ridge landforms (47 years). We suggest that the longer FCs on sand ridge landforms is associated with the fact that these areas occur within a matrix of fen and bog vegetation, which is not conductive of fire during most of the fire season. Our estimates of FC appear shorter than the 122–250 years reported for the Boundary Waters Canoe Area from 1542 to 1868 (Heinselman 1973). This difference may be due to better spatial resolution in our study, primarily because of a smaller study area (total area 7600 ha vs. 215 000 ha) and, therefore, increased detection rate for fires with the size range of 10–100 ha.

It is reasonable to assume that human impact on forest fire regimes was minimal in the pre-European settlement time period, as compared with more recent periods. We base this conclusion on the written history of the area, which indicates the middle of the 19th century as the time period when large-scale logging started in eastern Upper Michigan (Karamanski 1989; Maybee 1960; Zhang et al. 2000). Native Americans also played a role in shaping the histori-

© 2008 NRC Canada

Fig. 8. Estimated survivorship curves for sand ridges (A) and outwash channels (B) within the nonwilderness portion of SNWR over time. See Fig. 7 for further details.



cal fire regimes; however, these impacts might have been relatively localized (Loope and Anderton 1998; Silbernagel et al. 1997*b*). The prevalence of wetlands and the relative remoteness of SNWR may have influenced the degree of human impact over the settlement periods. At a smaller scale, the following observation supports this assumption. The presence of 350-year-old red pines in the wilderness, located on sand ridges in the matrix of patterned fens, and lack of similar stands in more easily accessible nonwilderness, indicates that cumulative human impact on an area may depend on its connectivity with the rest of the landscape.

More frequent fires in the postsettlement period (1860–1935) resulted in shorter FRIs and FCs across the whole SNWR landscape. We attribute this trend largely to the human impact on fire regimes through increased ignition frequency and higher fuel loads on logging sites and possibly greater fuel connectivity. Active logging began in this part of Upper Michigan in the second half of the 19th century, although historical data show that small-scale timber harvesting had started in the area already in the beginning of 19th century (Karamanski 1989; Taylor 1991). Increased human exploitation of the forests since the Great Cutover (Dickmann and Leefers 2003) inevitably resulted in the more frequent and sometimes deliberate ignitions (Karamanski 1989; Sodders 1997), which were likely reasons for both shorter FRIs and FCs during this period. Additionally, timber harvesting resulted in considerable slash and cutting residues, which might have increased the fire hazard by providing a better fuel continuity (Haines and Sando 1969; Karamanski 1989).

Another possible reason for synchronous decline in FC in wilderness and nonwilderness portions of SNWR was a number of large fires reconstructed during that period (in 1864, 1891, and 1919), all of which occurred across most of the SNWR landscape. Climatic forcing of these fire events is likely. However, among these 3 years, only 1864 is known for its precipitation anomaly, being one of the strongest droughts in North America (Cook et al. 2007; Stahle et al. 2007). Therefore, subregional climatic reconstruction should be helpful (e.g., Girardin et al. 2004) in separating role of humans and climate in increase in fire activity during the postsettlement period.

The time period after SNWR establishment was characterized by diverging trends in fire activity in different parts of SNWR. In the wilderness, the level of fire activity, expressed as FRI and FC, returned to levels observed in the pre-European settlement period. Beside frequent small fires, the large Seney Fire of 1976 (Anderson 1982) was a major contributor to the fire statistics for this area. Despite its size (260 km^2), our data show that this fire was not a stand-replacing event in the portions of the wilderness we studied. This observation supports our view that fires in mixed-pine forests comprised mostly of red pine at SNWR are mostly nonstand-replacing events, whereas stands comprised of more pyrophytic jack pine have a greater likelihood of experiencing these type of events (Bergeron et al. 1997).

In the nonwilderness portion of SNWR, levels of fire activity decreased dramatically as compared with previous time periods. Effective fire suppression policy that was introduced since the establishment of the SNWR (USFWS 2003), was the evident cause of decrease in fire activity. This trend, considered at a wider geographic perspective, was probably one of the most considerable human-mediated impacts on natural disturbance regimes in North America, leading to changes in stand densities, fuel loadings, and canopy compositions (Parsons and Debenedetti 1979; Baker 1993; Cumming 2005). Our reconstruction showed that fire ignitions did take place during that period; however, these fires tended to be small, probably because of active suppression activities. The most striking difference between the pre-European settlement and the period after SNWR establishment was observed for the outwash channel landforms in the nonwilderness portion of SNWR. It appears possible that high connectivity of this part of the landscape facilitated implementation of recent suppression policies as it did the spread of natural fires in the pre-European settlement time period.

Because our reconstruction is based on the dating of fire scars, it is important to realize that our fire chronologies provide a conservative estimate of the past fire activity at the SNWR. Not all fires result in tree scarring, and both decay of the deadwood and the limited number of samples per site might have caused some fires to be missed in the reconstruction (Swetnam and Baisan 1996). Specifically,

© 2008 NRC Canada

Table 4. Statistical descriptors of fire regime within different parts of SNWR during three time periods.

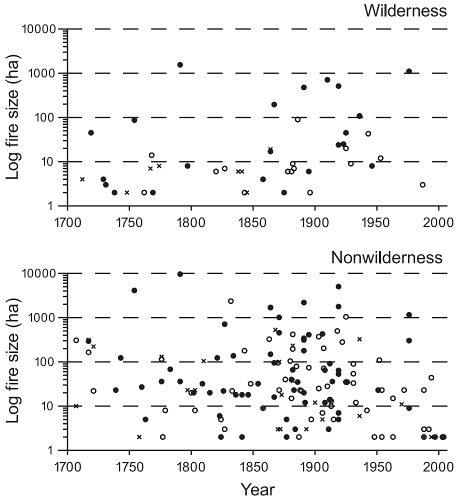
	Time period		
Subarea and variable	1707–1859	1860–1935	1936–2006
Wilderness			
Fire cycle (years)[a]	144 (29–369)	20 (11–30)	148 (20–436)
Fire size (ha)[b]	70 (2–1550)	149 (2–810)	143 (3–1106)
N	27	20	9
Nonwilderness			
Fire cycle (years)	47 (24–78)	19 (11–30)	196 (100–327)
Fire size (ha)	497 (6–9626)	325 (6–5032)	107 (5–1156)
N	48	81	26
Outwash channel			
Fire cycle (years)	53 (39–70)	30 (22–39)	179 (73–323)
Fire size (ha)	397 (26–2231)	298 (9–1534)	204 (5–1156)
N	27	31	11
Sand ridge area			
Fire cycle (years)	91 (36–193)	22 (16–31)	1090 (240–2809)
Fire size (ha)	241 (4–3211)	193 (1–1601)	33 (3–327)
N	26	46	12

Note: Both outwash channels and sand ridges were located in the nonwilderness (see Fig. 1).

[a] Values are means with the 10% and 90% confidence limits obtained through bootstrapping given in parentheses.

[b] Values are means with the range given in parentheses.

Fig. 9. Relationship between fire sizes and fire seasonality for 1707–2006 in the wilderness (A) and nonwilderness (B) areas of SNWR. Open circles show early season fires, with fire scars located within the earlywood, and solid circles show late-season fires, with fire scars located in the latewood. Crosses are the fires with unidentified seasonal dating. Note the logarithmic scale of the *y* axis.



this might lead to estimated FRIs and FCs to be longer than in reality.

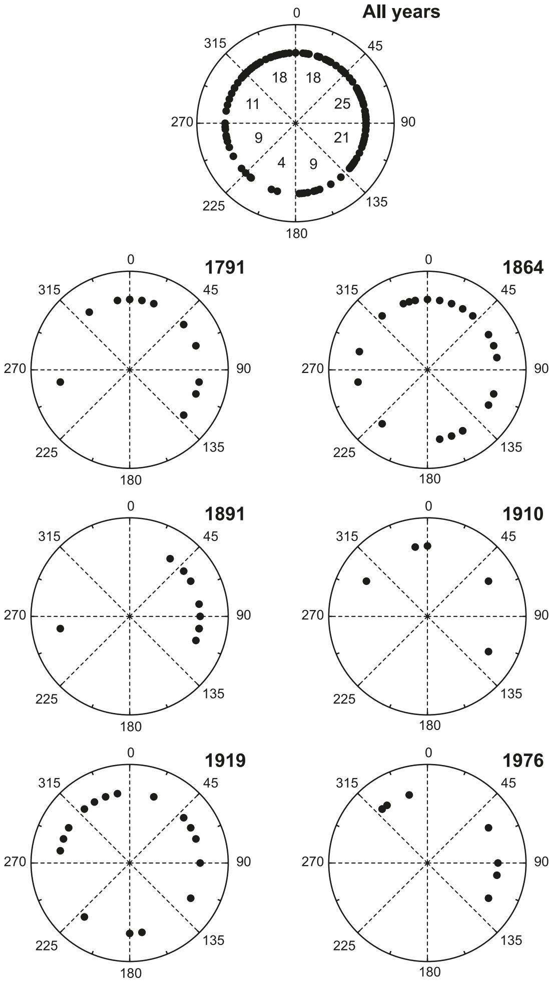
Effects of landform type

Synchronous changes in FRIs and FCs across the entire landscape were of higher magnitude than their variation between landform types. This trend suggests that landform-related factors were not the main drivers behind the variation in the historical fire regime at SNWR. For example, the increase in fires during the European settlement period (1860–1935) occurred simultaneously in all areas of SNWR. With no evidence of logging and associated probabilities of ignitions in the wilderness per se, the spread of fires from the surrounding areas experiencing logging operations may explain this dynamic. Similarly, a decrease in fires in the period after SNWR establishment (>1935), as compared with the previous time period, may be attributed to the decrease in the amount of outside ignitions contributing to the fire activity inside the wilderness. Therefore, we hypothesize that sand ridges in the wilderness, which appear relatively unconnected to the rest of the landscape because of the patterned fen vegetation matrix, have nevertheless been indirectly affected by the pattern of fire activity in neighboring parts of landscape, providing additional sources of ignitions.

Although differences in fire regimes across landform types appeared minor compared with the temporal variation in fire regimes, our study showed a consistent pattern of variation in FC across SNWR with longer FCs in portions of the landscape less conductive of fire (sand ridge landform type both in the wilderness and nonwilderness portions of SNWR) and shorter cycles in the area with high connectivity between red pine stands (outwash channel landform type). Whether these minor differences associated with the fire regimes between outwash channels and sand ridges significantly influenced stand composition and structure (including fuel loadings) remains to be determined.

During most fire seasons, the patterned fen matrix presents an effective fire break that limits eventual fire activity on sand ridges. On several occasions (i.e., 1754, 1791, 1864, 1891, 1919, and 1976) fires burned across SNWR, covering both wilderness and nonwilderness and

Fig. 10. Reconstructed direction of fires. Fire direction was calculated as an inverse of the direction of first fire scar on sampled trees for (A) the whole data set (central uppermost graph) and (B) separately for six major fire years recorded at SNWR. North–south directions on circular plots are represented by 0 and 180°, respectively. In Fig. 10A, the values in the sectors are the numbers of samples indicating the respective fire directions.



indicating that, during these years, the wetland vegetation was conductive to fire. The capacity of the poorly drained lacustrine plain to serve as a fire break was higher in the beginning of the fire season, because snowmelt transformed such areas into waterlogged marshland. Over the course of the summer, water levels generally decline, increasing fire conductive capacity of the vegetation. In line with this reasoning, almost all of the major fire years at SNWR were late-season fires and tended to move in a northeasterly direction. A similar temporal pattern of major fires, occurring typically at the end of the fire season, has been observed in many parts of the boreal zone, including forests of Alaska (Kasischke et al. 2002) and eastern European pine forests (Drobyshev et al. 2004). Therefore, because of its impact on tree regeneration and mortality patterns, landform-specific fire regimes may contribute to the variation in successional pathways within a landscape, as was suggested for upland forests of Lower Michigan (Host et al. 1987; Whitney 1987).

Characteristics of the terrain have been previously shown to influence fire behavior and result in variation in site-specific FRIs as a function of landscape connectivity and structure of vegetation cover acting as natural fire breaks (Engelmark 1987; Hellberg et al. 2004), fuel properties (Li et al. 2000; Rollins et al. 2002; Krawchuk et al. 2006), and environmental conditions, e.g., wind climate (Bessie and Johnson 1995). However, during at least 7 years over the last three centuries, fires have extended across the whole SNWR landscape; this suggests that landform-controlled differences in fire regimes tend to disappear with increasingly severe fire weather (Johnson et al. 1998).

We have more confidence in the estimates of fire regime in the wilderness than for the nonwilderness. This is due to better detection probability in this part of the SNWR and the higher possibility of interactions between natural factors (e.g., climate variation, ignition frequency, and fuel composition) and human factors (e.g., affecting primarily ignition frequency and fuel composition) on landform types in the nonwilderness. We believe that higher uncertainty in fire regime estimates in the nonwilderness acts towards lowering the estimates of actual fire activity. The reasons for this result include our conservative method of delineating the fire areas and our conservative adjustment for the lower detection probability of smaller fires. Our study clearly shows that the historical fire regime on outwash channels was more intense compared with sand ridges, but the difference between the two landform types should be interpreted as a minimum difference; the actual variation in fire activity across the SNWR landscape may be possibly even higher.

The estimated FC in the wilderness area (148 years) was found to be consistent over the pre-European settlement time period (1707–1860). This observation suggests that, currently, this part of SNWR has an FC very close to the long-term mean documented before extensive timber harvesting commenced in eastern Upper Michigan. Therefore, this area represents a valuable baseline for other studies of natural red pine forests and a benchmark for restoration efforts. An issue to be explored in future studies is how the legacies of time periods with different fire regimes translate into the current forest structure and projected successional pathways.

Management implications

Frequent (once in 25–35 years), low-intensity fires are most likely the main feature of the historic fire regimes in the red pine forest ecosystems of SNWR as they are elsewhere across the northern Lake States (Frelich and Lorimer 1991). Therefore, a management plan aimed at conserving and restoring red pine dominated forest ecosystems should include such fire events, which promote red pine regeneration and maintain low fuel loadings (Drobyshev et al. 2008). However, where red pine has been replaced by other species, other management tools (including mechanical treatments) should be considered before initiating a prescribed fire program. Prior to European settlement, site FRIs ranged between 23 and 33 years, which we consider as reference values for the SNWR. Seasonal reconstructions indicate that most of the study area burned during late-season fires, a pattern presenting a potential challenge for the forest managers. Emulating such a seasonal pattern would mean conducting prescribed fires under conditions

© 2008 NRC Canada

of higher fire hazard, generally observed towards the end of the fire season (Nelson 2001; Stocks et al. 2002).

An important question is whether a conservation- and restoration-oriented management plan should target certain levels of FRIs or be designed with estimates of historical FC in mind. For SNWR, these were about 150 years for the sand ridge landform type embedded within the patterned fens and about 50 years in the outwash channel landform type. We believe that it is the FC estimates that should ultimately serve as long-term guidelines for the conservation burns. The biological rationale for this decision comes from the fact that amount of area burned appears to be an important factor controlling the long-term dynamics of the amount of different habitat types, for example, abundance of early successional stages (Brooks 2003; Litvaitis 2003), landscape structural diversity, and dynamics of deadwood (Johnstone 2006), all of which have been shown to be closely connected to the dynamics of many species groups (Granström 2001; Muona and Rutanen 1994). Also, FC data incorporate our knowledge on spatial extent of fires, a feature of fire regime not represented in the FRI data.

Using FC reference values as a benchmark would imply introducing fire activity at the scale of 100–1000 ha, which may not be currently feasible because of high fuel loadings in many formerly red pine dominated forests and the high risk associated with escaping prescribed fires. Therefore, we advocate for the use of FRI data for the development of interim management targets. Subsequent versions of conservation plans may want to address targets based on FC data, which will inevitably expand the plan's scope and include the management of larger fire events (≥100 ha). High costs associated with prescribed fires of such size may render them currently infeasible as a tool to achieve the required amount of fire-impacted areas. In this context, a combination of small prescribed fires and a less restrictive fire suppression policies towards larger fires may prove a viable alternative even given clearly a negative perception of large fires in the general public (Arvai et al. 2006). Given that approach, smaller and easier to manage fires can maintain historical fire regimes at the scale of single stands, addressing the dominance of small fire events in the total number of historical fire record. Smaller fires may also help reduce continuity of fuels and risk of uncontrolled fire spread during larger fire events. Moreover, removal of excessive fuel loads in red pine stands should decrease the risk of high-intensity fires, which were shown to eliminate red pine (Bergeron and Brisson 1990).

At SNWR, areas in the nonwilderness should be the main targets for the application of conservation- and restoration-oriented fire management. Very long FCs on the sand ridge landform type of the nonwilderness coincide with short FRIs and reflect the fact that, during the most recent time period, fires in this part of the area, although relatively frequent, were confined to the sand ridges and did not expand over the poorly drained lacustrine plain. To what degree the conservation- and restoration-oriented management of red pine ecosystems should extend beyond the pine forests themselves is the subject of further analysis and discussion.

Acknowledgements

Salaries and financial support for this research were provided by a grant from the Joint Fire Science Program (Project 05-2-1-86), the Ohio Agricultural Research and Development Center, The Ohio State University, and SNWR. We thank the staff of SNWR for their logistical support. We also thank Stephen Rist, Bridget Deemer, Jim Downs, and Heather Whitman for assistance in the field.

References

Ahlgren, C.E. 1976. Regeneration of red pine and white pine following wildfire and logging in northeastern Minnesota. J. For. **74**: 135–140.

Albert, D.A. 1995. Regional landscape ecosystems of Michigan, Minnesota, and Wisconsin: a working map and classification. USDA For. Serv. Gen. Tech. Rep. NC-178.

Anderson, S.H. 1982. Effects of the 1976 Seney National Wildlife Refuge wildfire on wildlife and wildlife habitat. United States Department of Interior, Fish and Wildlife Service, Washington, D.C. Resour. Publ. 146.

Arvai, J., Gregory, R., Ohlson, D., Blackwell, B., and Gray, R. 2006. Letdowns, wake-up calls, and constructed preferences: people's responses to fuel and wildfire risks. J. For. **104**: 173–181.

Asselin, H., Fortin, M.J., and Bergeron, Y. 2001. Spatial distribution of late-successional coniferous species regeneration following disturbance in southwestern Quebec boreal forest. For. Ecol. Manage. **140**: 29–37. doi:10.1016/S0378-1127(00)00273-5.

Baker, W.L. 1993. Spatially heterogeneous multiscale response of landscapes to fire suppression. Oikos, **66**: 66–71. doi:10.2307/3545196.

Barnett, L. 1982. U.P. surveyors. Mich. Hist. **66**: 24–31.

Bergeron, Y., and Brisson, J. 1990. Fire regime in red pine stands at the northern limit of the species range. Ecology, **71**: 1352–1364. doi:10.2307/1938272.

Bergeron, Y., and Gagnon, D. 1987. Age structure of red pine (*Pinus resinosa* Ait.) at its northern limit in Quebec. Can. J. For. Res. **17**: 129–137. doi:10.1139/x87-024.

Bergeron, Y., Leduc, A., and Li, T.X. 1997. Explaining the distribution of *Pinus* spp. in a Canadian boreal insular landscape. J. Veg. Sci. **8**: 37–44. doi:10.2307/3237240.

Bergeron, Y., Flannigan, M., Gauthier, S., Leduc, A., and Lefort, P. 2004. Past, current and future fire frequency in the Canadian boreal forest: implications for sustainable forest management. Ambio, **33**: 356–360. doi:10.1639/0044-7447(2004)033[0356:PCAFFF]2.0.CO;2. PMID:15387074.

Bessie, W.C., and Johnson, E.A. 1995. The relative importance of fuels and weather on fire behavior in sub-alpine forests. Ecology, **76**: 747–762. doi:10.2307/1939341.

Brooks, R.T. 2003. Abundance, distribution, trends, and ownership patterns of early-successional forests in the northeastern United States. For. Ecol. Manage. **185**: 65–74. doi:10.1016/S0378-1127(03)00246-9.

Brown, P.M. 2006. Climate effects on fire regimes and tree recruitment in Black Hills ponderosa pine forests. Ecology, **87**: 2500–2510. doi:10.1890/0012-9658(2006)87[2500:CEOFRA]2.0.CO;2. PMID:17089659.

Busque, D., and Arseneault, D. 2005. Fire disturbance of larch woodlands in string fens in northern Quebec. Can. J. Bot. **83**: 599–609. doi:10.1139/b05-028.

Caprio, A.C., and Swetnam, T.W. 1995. Historic fire regimes along an elevational gradient on the west slope of the Sierra Nevada, California. *In* Proceedings of a Symposium on Fire in Wilderness and Park Management: Past Lessons and Future Opportunities, 30 March – 1 April 1993, Missoula, Mont. *Technical* *coordinators* J.K. Brown, R.W. Mutch, C.W. Spoon, and R.H. Wakimoto. USDA For. Serv. Gen. Tech. Rep. INT-GTR-320.

Cleland, D.T., Crow, T.R., Saunders, S.C., Dickmann, D.I., Maclean, A.L., Jordan, J.K., Watson, R.L., Sloan, A.M., and Brosofske, K.D. 2004. Characterizing historical and modern fire regimes in Michigan (USA): a landscape ecosystem approach. Landsc. Ecol. **19**: 311–325. doi:10.1023/B:LAND.0000030437.29258.3c.

Comer, P.J., Albert, D.A., Wells, H.A., Hart, B.L., Raab, J.B., Price, D.L., Kashian, D.M., Corner, R.A., and Schuen, D.W. 1995. Michigan's native landscape, as interpreted from the General Land Surveys 1816–1856. Michigan Natural Features Inventory, Lansing, Mich.

Cook, E.R., Seager, R., Cane, M.A., and Stahle, D.W. 2007. North American drought: reconstructions, causes, and consequences. Earth Sci. Rev. **81**: 93–134. doi:10.1016/j.earscirev.2006.12.002.

Corace, R.G., III, Goebel, P.C., and Seefelt, N.E. 2008. Approaches to forest management at National Wildlife Refuges in the Lake States: Seney National Wildlife Refuge and Kirtland's Warbler Wildlife Management Area. Forest Wisdom, The Forest Guild, Santa Fe, N.M.

Cumming, S.G. 2005. Effective fire suppression in boreal forests. Can. J. For. Res. **35**: 772–786. doi:10.1139/x04-174.

Dickinson, M.B., and Johnson, E.A. 2001. Fire effects on trees. *In* Forest fires: behavior and ecological effects. *Edited by* E.A. Johnson and K. Miyanishi. Academic Press, London. pp. 477–526.

Dickmann, D.I., and Leefers, L.A. 2003. The forests of Michigan. University of Michigan Press, Ann Arbor, Mich.

Dodson, B. 1994. Weibull analysis. American Society for Quality Control, Milwaukee, Wis.

Drobyshev, I., Niklasson, M., Angelstam, P., and Majewski, P. 2004. Testing for anthropogenic influence on fire regime for a 600-year period in Jaksha area, Komi Republic, East European Russia. Can. J. For. Res. **34**: 2027–2036. doi:10.1139/x04-081.

Drobyshev, I., Goebel, P.C., Hix, D.M., Corace, R.G., III, and Semko-Duncan, M.E. 2008. Interactions between forest composition, structure, fuel loadings and fire history: a case study of red pine-dominated forests of Seney National Wildlife Refuge, Upper Michigan. For. Ecol. Manage. In press.

Efron, B., and Tibshirani, R.J. 1994. An introduction to the bootstrap. Chapman & Hall, New York.

Engelmark, O. 1987. Fire history correlations to forest type and topography in northern Sweden. Ann. Bot. Fenn. **24**: 317–324.

Engstrom, F.B., and Mann, D.H. 1991. Fire ecology of red pine (*Pinus resinosa*) in northern Vermont, USA. Can. J. For. Res. **21**: 882–889. doi:10.1139/x91-123.

Flannigan, M., Campbell, I., Wotton, M., Carcaillet, C., Richard, P., and Bergeron, Y. 2001. Future fire in Canada's boreal forest: paleoecology results and general circulation model – regional climate model simulations. Can. J. For. Res. **31**: 854–864. doi:10.1139/cjfr-31-5-854.

Flannigan, M.D., and Bergeron, Y. 1998. Possible role of disturbance in shaping the northern distribution of *Pinus resinosa*. J. Veg. Sci. **9**: 477–482. doi:10.2307/3237262.

Frelich, L.E., and Lorimer, C.G. 1991. Natural disturbance regimes in hemlock–hardwood forests of the Upper Great Lakes region. Ecol. Monogr. **61**: 145–164. doi:10.2307/1943005.

Friedman, S.K., and Reich, P.B. 2005. Regional legacies of logging: departure from presettlement forest conditions in northern Minnesota. Ecol. Appl. **15**: 726–744. doi:10.1890/04-0748.

Girardin, M.P., Tardif, J., Flannigan, M.D., Wotton, B.M., and Bergeron, Y. 2004. Trends and periodicities in the Canadian Drought Code and their relationships with atmospheric circulation for the southern Canadian boreal forest. Can. J. For. Res. **34**: 103–119. doi:10.1139/x03-195.

Granström, A. 2001. Fire management for biodiversity in the European boreal forest. Scand. J. For. Res. **16**(Suppl. 3): 62–69. doi:10.1080/028275801300090627.

Grissino-Mayer, H.D. 1999. Modeling fire interval data from the American Southwest with the Weibull distribution. Int. J. Wildland Fire, **9**: 37–50. doi:10.1071/WF99004.

Haines, D.A., and Sando, R.W. 1969. Climatic conditions preceding historically great fires in the North Central Region. USDA For. Serv. North Cent. For. Range Exp. Stn. Res. Rep. NC-34.

Heinselman, M.L. 1965. String bogs and other patterned organic terrain near Seney, Upper Michigan. Ecology, **46**: 185–188. doi:10.2307/1935270.

Heinselman, M.L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quat. Res. **3**: 329–382. doi:10.1016/0033-5894(73)90003-3.

Hellberg, E., Niklasson, M., and Granström, A. 2004. Influence of landscape structure on patterns of forest fires in boreal forest landscapes in Sweden. Can. J. For. Res. **34**: 332–338. doi:10.1139/x03-175.

Horton, K.W., and Bedell, G.H.D. 1960. White and red pine ecology, silviculture and management. Can. Dep. North. Aff. Natl. Res. For. Branch Bull. 124.

Host, G.E., Pregitzer, K.S., Ramm, C.W., Hart, J.B., and Cleland, D.T. 1987. Landform-mediated differences in successional pathways among upland forest ecosystems in northwestern Lower Michigan. For. Sci. **33**: 445–457.

Johnson, E.A., Miyanishi, K., and Weir, J.M.H. 1998. Wildfires in the western Canadian boreal forest: landscape patterns and ecosystem management. J. Veg. Sci. **9**: 603–610. doi:10.2307/3237276.

Johnstone, J.F. 2006. Response of boreal plant communities to variations in previous fire-free interval. Int. J. Wildland Fire, **15**: 497–508. doi:10.1071/WF06012.

Kaplan, E.L., and Meier, P. 1958. Nonparametric estimation from incomplete observations. J. Am. Stat. Assoc. **53**: 457–481. doi:10.2307/2281868.

Karamanski, T. 1989. Deep woods frontier: a history of logging in northern Michigan. Wayne State University Press, Detroit, Mich.

Kasischke, E.S., Williams, D., and Barry, D. 2002. Analysis of the patterns of large fires in the boreal forest region of Alaska. Int. J. Wildland Fire, **11**: 131–144. doi:10.1071/WF02023.

Krawchuk, M.A., Cumming, S.G., Flannigan, M.D., and Wein, R.W. 2006. Biotic and abiotic regulation of lightning fire initiation in the mixedwood boreal forest. Ecology, **87**: 458–468. doi:10.1890/05-1021. PMID:16637370.

Lee, E.T., Desu, M.M., and Gehan, E.A. 1975. A Monte-Carlo study of the power of some two-sample tests. Biometrika, **62**: 425–432. doi:10.1093/biomet/62.2.425.

Le Goff, H., Flannigan, M.D., Bergeron, Y., and Girardin, M.P. 2007. Historical fire regime shifts related to climate teleconnections in the Waswanipi area, central Quebec, Canada. Int. J. Wildland Fire, **16**: 607–618. doi:10.1071/WF06151.

Li, C., Flannigan, M.D., and Corns, I.G.W. 2000. Influence of potential climate change on forest landscape dynamics of west-central Alberta. Can. J. For. Res. **30**: 1905–1912. doi:10.1139/cjfr-30-12-1905.

Litvaitis, J.A. 2003. Are pre-Columbian conditions relevant baselines for managed forests in the northeastern United States? For. Ecol. Manage. **185**: 113–126. doi:10.1016/S0378-1127(03)00250-0.

Loope, W.L., and Anderton, J.B. 1998. Human vs. lightning ignition of presettlement surface fires in coastal pine forests of the upper Great Lakes. Am. Midl. Nat. **140**: 206–218. doi:10.1674/0003-0031(1998)140[0206:HVLIOP]2.0.CO;2.

Losey, E.B. 2003. Seney National Wildlife Refuge: its story. Lake Superior Press, Marquette, Mich.

Maybee, R.H. 1960. Michigan's white pine era 1840–1900. Michigan Historical Commission, Lansing, Mich.

Midwestern Regional Climate Center (MRCC). 2007. The Midwestern Regional Climate Center. NCDC & the Illinois State Water Survey, Champaign, Ill. Available from mcc.sws.uiuc.edu/index.jsp [accessed 14 September 2007].

Muona, J., and Rutanen, I. 1994. The short-term impact of fire on the beetle fauna in boreal coniferous forest. Ann. Zool. Fenn. **31**: 109–121.

Nelson, R.M. 2001. Water relations of forest fuels. *In* Forest fires: behavior and ecological effects. Academic Press, San Francisco, Calif. pp. 79–149.

Niklasson, M., and Granström, A. 2000. Numbers and sizes of fires: long-term spatially explicit fire history in a Swedish boreal landscape. Ecology, **81**: 1484–1499.

Palik, B., and Zasada, J. 2003. An ecological context for regenerating multi-cohort, mixed-species red pine forests. USDA For. Serv. North Cent. Res. Stn. Res. Note NC-382.

Parsons, D.J., and Debenedetti, S.H. 1979. Impact of fire suppression on a mixed-conifer forest. For. Ecol. Manage. **2**: 21–33. doi:10.1016/0378-1127(79)90034-3.

Rollins, M.G., Morgan, P., and Swetnam, T. 2002. Landscape-scale controls over 20th century fire occurrence in two large Rocky Mountain (USA) wilderness areas. Landsc. Ecol. **17**: 539–557. doi:10.1023/A:1021584519109.

Rudolf, P.O. 1990. *Pinus resinosa* Ait. red pine. *In* Silvics of North America. Vol. 1. Conifers. *Technical coordinators* R.M. Burns and B.H. Honkala. USDA Agric. Handb. 654. pp. 442–455.

Schweingruber, F.H., Eckstein, D., Serre-Bachet, F., and Bräker, O.U. 1990. Identification, presentation and interpretation of event years and pointer years in dendrochronology. Dendrochronologia, **8**: 9–38.

Silbernagel, J., Chen, J., Gale, M.R., Pregitzer, K.S., and Probst, J. 1997*a*. An interpretation of landscape structure from historic and present land cover data in the eastern Upper Peninsula of Michigan. USDA For. Serv. Gen. Tech. Rep. NC-192.

Silbernagel, J., Martin, S.R., Gale, M.R., and Chen, J.Q. 1997*b*. Prehistoric, historic, and present settlement patterns related to ecological hierarchy in the eastern Upper Peninsula of Michigan, USA. Landsc. Ecol. **12**: 223–240. doi:10.1023/A:1007946907682.

Sodders, B. 1997. Michigan on fire. Thunder Bay Press, Holt, Mich.

Sokal, R.R., and Rolf, F.J. 1995. Biometry: the principles and practice of statistics in biological research. W.H. Freeman, New York.

Stahle, D.W., Fye, F.K., Cook, E.R., and Griffin, R.D. 2007. Tree-ring reconstructed megadroughts over North America since AD 1300. Clim. Change, **83**: 133–149. doi:10.1007/s10584-006-9171-x.

Starker, T.J. 1934. Fire resistance in the forest. J. For. **32**: 462–467.

StatSoft Inc. 2001. STATISTICA (data analysis software system), version 6 edition. StatSoft Inc., Tulsa, Okla. Available from www.statsoft.com.

Stocks, B.J., Mason, J.A.., Todd, J.B., Bosch, E.M., Wotton, B.M.., Amiro, B.D., Flannigan, M.D., Hirsch, K.G.., Logan, K.A., Martell, D.L., and Skinner, W.R. 1997*b*. Large forest fires in Canada, 1959–1997. J. Geophys. Res. Atmos. **108**(D1): 8149. doi:10.1029/2001JD000484.

Stokes, M.A., and Smiley, T.L. 1968. An introduction to tree-ring dating. University of Chicago Press, Chicago, Ill.

Swetnam, T.W. 1996. Fire and climate history in the Central Yenisey region, Siberia. *In* Fire in ecosystems of boreal Eurasia. *Edited by* J.G. Goldammer and V.V. Furyeav. Kluwer Academic Publishers, Dordrecht, the Netherlands. pp. 90–104.

Swetnam, T.W., and Baisan, C.H. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. USDA For. Serv. Gen. Tech. Rep. RM-286.

Swetnam, T.W., and Baisan, C.H. 2003. Tree-ring reconstructions of fire and climate history in the Sierra Nevada and southwestern United States. *In* Fire and climatic change in temperate ecosystems of the western Americas. *Edited by* T.T. Veblen, W. Baker, G. Montenegro, and T.W. Swetnam. Springer, New York. Ecol. Stud. 160. pp. 158–195.

Taylor, S. 1991. Tahquamenon Country. A look at its past. Historical Society of Michigan, Ann Arbor, Mich.

United States Fish and Wildlife Service (USFWS). 2003. Environmental assessment for wildland fire management plan. Seney National Wildlife Refuge, Seney, Mich.

Van Wagner, C.E. 1978. Age-class distribution and the forest fire cycle. Can. J. For. Res. **8**: 220–227. doi:10.1139/x78-075.

Whitney, G.G. 1986. Relation of Michigan presettlement pine forests to substrate and disturbance history. Ecology, **67**: 1548–1559. doi:10.2307/1939085.

Whitney, G.G. 1987. An ecological history of the Great Lakes forest of Michigan. J. Ecol. **75**: 667–684. doi:10.2307/2260198.

Woodall, C.W., Charney, J.J., Liknes, G.C., and Potter, B.E. 2005. What is the fire danger now? Linking fuel inventories with atmospheric data. J. For. **103**: 293–298.

Zhang, Q.F., Pregitzer, K.S., and Reed, D.D. 2000. Historical changes in the forests of the Luce District of the Upper Peninsula of Michigan. Am. Midl. Nat. **143**: 94–110. doi:10.1674/0003-0031(2000)143[0094:HCITFO]2.0.CO;2.