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FIRE AND FOREST HISTORY AT MOUNT RUSHMORE

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Abstract. Mount Rushmore National Memorial in the Black Hills of South Dakota is known worldwide for its massive sculpture of four of the United States' most respected presidents. The Memorial landscape also is covered by extensive ponderosa pine (*Pinus ponderosa*) forest that has not burned in over a century. We compiled dendroecological and forest structural data from 29 plots across the 517-ha Memorial and used fire behavior modeling to reconstruct the historical fire regime and forest structure and compare them to current conditions. The historical fire regime is best characterized as one of low-severity surface fires with occasional (>100 years) patches (<100 ha) of passive crown fire. We estimate that only ~3.3% of the landscape burned as crown fire during 22 landscape fire years (recorded at ≥25% of plots) between 1529 and 1893. The last landscape fire was in 1893. Mean fire intervals before 1893 varied depending on spatial scale, from 34 years based on scar-to-scar intervals on individual trees to 16 years between landscape fire years. Modal fire intervals were 11–15 years and did not vary with scale. Fire rotation (the time to burn an area the size of the study area) was estimated to be 30 years for surface fire and 800+ years for crown fire. The current forest is denser and contains more small trees, fewer large trees, lower canopy base heights, and greater canopy bulk density than a reconstructed historical (1870) forest. Fire behavior modeling using the NEXUS program suggests that surface fires would have dominated fire behavior in the 1870 forest during both moderate and severe weather conditions, while crown fire would dominate in the current forest especially under severe weather. Changes in the fire regime and forest structure at Mount Rushmore parallel those seen in ponderosa pine forests from the southwestern United States. Shifts from historical to current forest structure and the increased likelihood of crown fire justify the need for forest restoration before a catastrophic wildfire occurs and adversely impacts the ecological and aesthetic setting of the Mount Rushmore sculpture.

Key words: dendroecology; fire behavior; fire frequency; fire history; fire severity; forest structure; ponderosa pine; reference dynamics; restoration ecology.

Introduction

Evidence of past ecosystem conditions provides answers for the "what" and "why" of ecological restoration efforts: what do we restore to, and why is it important to do so? Knowledge of historical conditions provides not only guidance but, perhaps more importantly, justification for restoration efforts designed to return an altered or degraded ecosystem to some semblance of its longer-term ecological trajectory (Falk 1990, White and Walker 1997, Landres et al. 1999, Allen et al. 2002, Egan and Howell 2005). Justification of the need for restoration is especially critical in areas such as National Parks or other natural areas where a scientific foundation for restoration is often crucial for public acceptance (Johnson and Campbell 1999, Brown et al. 2001, Brunson and Shindler 2004).

Mount Rushmore National Memorial in the Black Hills of South Dakota is one place where public scrutiny of natural resource decisions can be intense owing to both its large numbers of annual visitors (~2.5 million in 2005) and its symbolism as a national icon. The Memorial is known worldwide for its massive sculpture of four of the United States' most respected presidents. However, the 517-ha landscape encompassed by the National Park Service unit surrounding the sculpture is covered by continuous and often dense ponderosa pine (*Pinus ponderosa*) forest. Exclusion of episodic surface fires in ponderosa pine forests across western North America has resulted in changes in forest structure that have increased the likelihood for widespread, catastrophic crown fires (e.g., Covington and Moore 1994, Brown et al. 2001, Allen et al. 2002). An extensive crown fire at Mount Rushmore would severely impact the ecological and aesthetic setting of the sculpture, and park managers have begun the process of planning and implementing fuel reduction and fire mitigation treatments (NPS 2003).

Mount Rushmore also is relatively unique in the Black Hills in that it contains some of the largest and last contiguous stands of ponderosa pine forest that

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have not had any timber harvest (Symstad and Bynum 2007). Intensive timber harvest began across the Black Hills with Euro-American settlement in 1874 (Graves 1899, Shepperd and Battaglia 2002). By the end of the 1800s large portions of the presettlement forest had already been affected by harvest (Graves 1899), and by the end of the 20th century almost all stands have seen some level of cutting (Shepperd and Battaglia 2002, Brown and Cook 2006). However, about one-third of the forest at Mount Rushmore has not seen any harvest, and approximately two-thirds of the forest is considered as old growth (Symstad and Bynum 2007). The Mount Rushmore forest is similar to other ponderosa pine forests in the Black Hills and could serve as a valuable reference landscape for old-growth ponderosa pine ecosystems throughout the region.

Our purposes with this study were to reconstruct fire and forest history at Mount Rushmore and to estimate changes in forest structure and fire behavior from a historical forest in 1870 to the present (sensu Fulé et al. 1997, 2004). We used a systematic dendroecological sampling approach to reconstruct tree demography (recruitment and mortality) and the fire regime (fire frequency, seasonality, severities, sizes, and spatial patterning) over the past ca. five centuries. We then used reconstructed and current forest structures as inputs for simulation modeling of landscape-scale potential for crown fire behavior using the NEXUS Fire Behavior and Hazard Assessment System (Scott and Reinhardt 2001). Two major questions we addressed with the reconstructions and modeling are the scale of crown fire relative to surface fire in the historical forest and how this scale compares to that in the current forest. Several studies have concluded that variable-severity (also called mixed-severity) fire regimes and denser forest stands occurred historically in many ponderosa pine forests in the northern part of its range, including the Black Hills (Shinneman and Baker 1997, Baker et al. 2007). Variable-severity fire regimes included a mix of crown and surface fire behavior that occurred both during individual fires and in individual stands over time. In this study, we used explicit criteria to examine spatial and temporal scales of fire behavior and stand structure across the historical landscape and compared these to conditions in the current forest. We found that although there is evidence of both crown fires and denser forest structure in the presettlement forest, there have been significant increases in coverage of dense stands and small-diameter trees in the current forest that greatly increase the likelihood of landscape-scale crown fire relative to surface fire. These changes support the need for restoration of the ponderosa pine forest to a more sustainable condition before a catastrophic wildfire occurs and adversely impacts the setting of the Mount Rushmore sculpture.

Methods

Study area

Mount Rushmore National Memorial encompasses 517 ha of forested ridges and swales and granite knobs in the central granitic core area of the Black Hills. Elevations range from ~1340 m to the high point on Mount Rushmore at 1745 m. Vegetation consists mainly of continuous stands of pure ponderosa pine with scattered small patches (<0.1 ha) and individual trees of aspen (*Populus tremuloides*), burr oak (*Quercus macrocarpa*), white spruce (*Picea glauca*), and paper birch (*Betula papyrifera*). A few small meadows (<0.1 ha) also are present in drainage bottoms. Average temperatures at the Memorial headquarters from 1948 to 2004 ranged from 25°C in July to −2°C in January. Precipitation during this period averaged 53 cm/yr, most of which fell as rain during summer convective thunderstorms.

The Black Hills, including the Mount Rushmore area, were initially settled by Euro-Americans in 1874 (Graves 1899, Shepperd and Battaglia 2002). Before this, the Lakota Sioux occupied this area since at least the mid- to late-1700s. Native American history of the Black Hills before the 1700s is less certain. Intensive timber harvest, mining, and widespread livestock grazing accompanied Euro-American settlement, although these impacts were less in the Mount Rushmore area because of the roughness of the terrain and unsuitability for mineral extraction. Timber harvest occurred primarily in the northeastern portions of the Memorial (Symstad and Bynum 2007). The Mount Rushmore sculpture was started in 1927 and completed in 1937, followed by establishment of the National Park Service unit in 1938.

Reconstruction of fire regimes and forest history

We used dendrochronological analyses of static (i.e., present-day) tree ages and fire-scar records to reconstruct the fire regime and forest history at Mount Rushmore. Current age structure reflects tree survivorship resulting from the combination of tree natality and mortality through time. An assumption is often made that large stands of even-aged forest structure generally result from past stand-opening events, such as crown fires, while multi-aged structure generally is indicative of more temporally continuous and spatially patchier mortality and recruitment events. However, evidence of past events is eliminated by subsequent events, and current age structure is the most conservative evidence for past fire severity. Fire scars result from lower-severity surface burning that injures but does not kill a tree. In both cases, dendrochronological cross-dating is crucial to provide absolute pith and fire scar dates for comparison of dated events across landscapes or with climate reconstructions or other annually resolved data (Swetnam and Brown 2008).

We established a 500-m grid over the Memorial for sample plot locations (Fig. 1, Table 1). We decreased

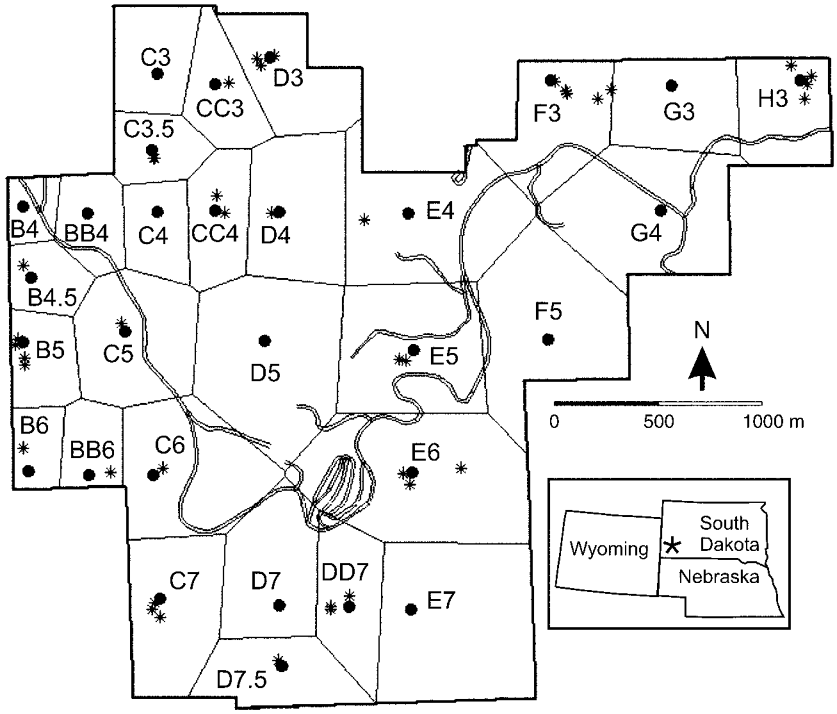

FIG. 1. Locations of plots (circles) and fire-scarred trees (asterisks) sampled at Mount Rushmore, South Dakota, USA. The heavy line is the Mount Rushmore National Memorial boundary. Double light lines are roads. Single light lines are polygon boundaries drawn around plots.

TABLE 1. Plots sampled to reconstruct the fire regime and forest history at Mount Rushmore National Memorial, South Dakota, USA.

Plot	Plot area (m^2)	Polygon area (ha)	Aspect (deg)	Slope (%)	Elevation (m)	No. plot trees	No. polygon trees
B4	774	4.77	20	30	1614	15	0
B4.5	814	7.39	190	15	1655	33	1
BB4	1385	8.97	24	45	1599	30	0
B5	515	9.08	50	45	1645	35	4
B6	1195	5.91	14	40	1663	32	2
BB6	1735	7.5	50	40	1618	34	0
C3	1320	12.37	226	29	1642	38	0
C3.5	394	8.76	150	27	1590	17	3
CC3	1110	10.95	150	35	1636	35	2
C4	1029	9.01	240	20	1614	39	0
CC4	452	11.52	45	35	1604	35	2
C5	1576	21.46	92	52	1622	37	1
C6	1018	25.14	0	0	1547	21	1
C7	1075	20.36	32	40	1546	39	3
D3	1195	16.29	77	3	1642	38	3
D4	804	20.62	65	25	1573	42	0
D5	1735	32.95	168	37	1587	35	1
D7	707	18.82	218	15	1574	34	0
D7.5	855	14.27	142	5	1587	39	2
DD7	755	13.95	180	5	1585	34	3
E4	616	29.53	50	56	1507	34	0
E5	531	27.22	2	12	1522	28	3
E6	380	39.28	12	35	1541	27	3
E7	468	35.66	105	21	1588	38	0
F3	633	18.03	18	21	1429	31	5
F5	266	30.97	7	23	1476	19	0
G3	2075	16.97	124	25	1447	44	0
G4	1765	30.66	125	35	1409	36	0
H3	1257	14.64	24	24	1403	38	4

grid spacing to 250 m in the west and south to increase sampling of older living trees from unharvested stands to better estimate canopy base heights for use in fire behavior modeling. Plot centers were located in the field using a handheld global positioning system (GPS) unit. Plot D3 was moved 50 m west from its original GPS coordinates to avoid a rock outcrop. Plots B4, C5, and G4 were moved 50, 100, and 50 m (respectively) west from their original GPS coordinates to avoid roads. Plots D6 and F4 were not sampled because of their locations in developed areas. The Mount Rushmore sculpture is located ~300 m west from plot D5.

Within each plot, an *n*–tree density-adapted sampling approach was used to select the ~30 presettlement trees (range 27–34) closest to plot center to characterize stand structure and tree demography. This sampling design has been used in numerous recent studies in multiple forest types across the western United States (Brown and Wu 2005, Brown 2006, Heyerdahl et al. 2006, Brown and Schoettle 2008, Brown et al. 2008). Presettlement trees were defined as living trees $\geq$20 cm dbh (diameter at breast height, 1.3 m) and all remnant trees (snags, logs, and stumps). Remnant trees were classified by decay status, whether bark, sapwood, or only heartwood was present. Increment cores were collected from ~10 cm height above ground level on living trees and cross sections were cut with a chainsaw from remnant trees such that one surface was at an estimated 10 cm height above root crown. Cores had to be no more than a field-estimated 10 years from pith to minimize pith offset when determining recruitment dates. Recruitment dates are defined as the 10 cm height pith dates. Diameters at 10 cm height (diameter at sample height, dsh) were measured on all trees. We also measured dbh on living trees for later conversion of dsh to dbh on remnant trees; these values were used for determining tree and stand basal areas. We also collected the same data from ~10 living trees (range 0–12) or recent stumps (some stands on the eastern edge of the Memorial had been thinned 2–4 years prior to sampling) <20 cm dbh closest to each plot center to assess structure and recruitment dates of smaller diameter classes. Increment cores and cross sections from the smaller diameter trees also were collected at ~10 cm heights.

To determine surface fire history and fire timing, we collected cross sections from all dead fire-scarred trees within plots and up to 5 (range 0–5) additional dead fire-scarred trees found within ~80 m of plot centers (Table 1). The additional fire-scarred trees were selected based on presence of multiple fire-scar records with the goal of supplementing fire dates found on plot trees. No cross sections were sampled from living trees because of concerns about possible damage to old-growth trees in the Memorial. Fire-scarred trees sampled outside plots were located with a GPS unit, measured for dsh, and sample height above an estimated root–shoot boundary recorded. We also opportunistically sampled an additional five fire-scarred trees with multiple fire-scar records while walking between plots. Cross sections were collected from these trees as being excellent representations of fire-scar sequences for possible interpretative displays in the Memorial visitor center.

Increment cores and cross sections from a total of exactly 1000 trees were sampled from the Mount Rushmore landscape (Table 1). Twelve trees were species other than ponderosa pine (three each of burr oak, aspen, paper birch, and white spruce). All tree-ring samples were prepared and cross-dated using standard dendrochronological procedures. We used a locally developed master chronology for visual cross-dating of samples. On increment cores and cross sections that did not include pith but inside ring curvature was visible, pith dates were estimated using overlaid concentric circles of varying circumferences that take into account both average inside ring widths and estimated distance to pith. Only after cross-dating of ring series was completed for fire-scarred samples did we assign dates for fire scars. We also assigned seasonal positions for fire scars based on locations within annual ring boundaries (Dieterich and Swetnam 1984). Dormant season scars (recorded between two rings) were dated to the prior year (i.e., to have been fall fires) because of the abundance of late-earlywood or latewood scars when recorded within rings. Any tree-ring samples that could not be cross-dated were not used in subsequent analyses.

We used presence and dates of fire scars and patterns of tree recruitment and mortality to estimate historical fire frequency, fire severity, and fire sizes. Time spans of trees and their fire-scar dates were compiled into fire-demography diagrams (FDDs) using program FHX2 (Grissino-Mayer 2001; Fig. 2). FDDs were compiled both for individual plots and for polygons drawn around plots using a minimum distance tessellation (Fig. 1). Plot trees are those sampled systematically within plots while polygon trees are the additional trees targeted for their fire-scar records (those out to ~80 m from plot centers plus the five additional trees sampled while walking between plots).

We estimated fire frequency using both point- and area-based approaches (Li 2002). We first used fire intervals to estimate five point-fire frequencies at four spatial scales: (1) scar-to-scar intervals found on individual trees (areas of <1 m^2); (2) plot composite intervals based on fire-scar dates from plot trees (areas of 266–2075 m^2; Table 1); (3) polygon composite intervals based on fire-scar dates from both plot and polygon trees (areas of 4.77–39.28 ha; Table 1); (4) landscape composite fire intervals based on fire-scar dates recorded in at least two plots for the period from 1529 to 1893 (an area of 517 ha); and (5) landscape composite fire dates that were recorded in at least 25% of the plots from 1529 to 1893 (517 ha). We also compiled intervals from pith dates to dates of first scars on individual trees to compare to scar-to-scar and composited data. Baker and Ehle (2001) argue that origin-to-

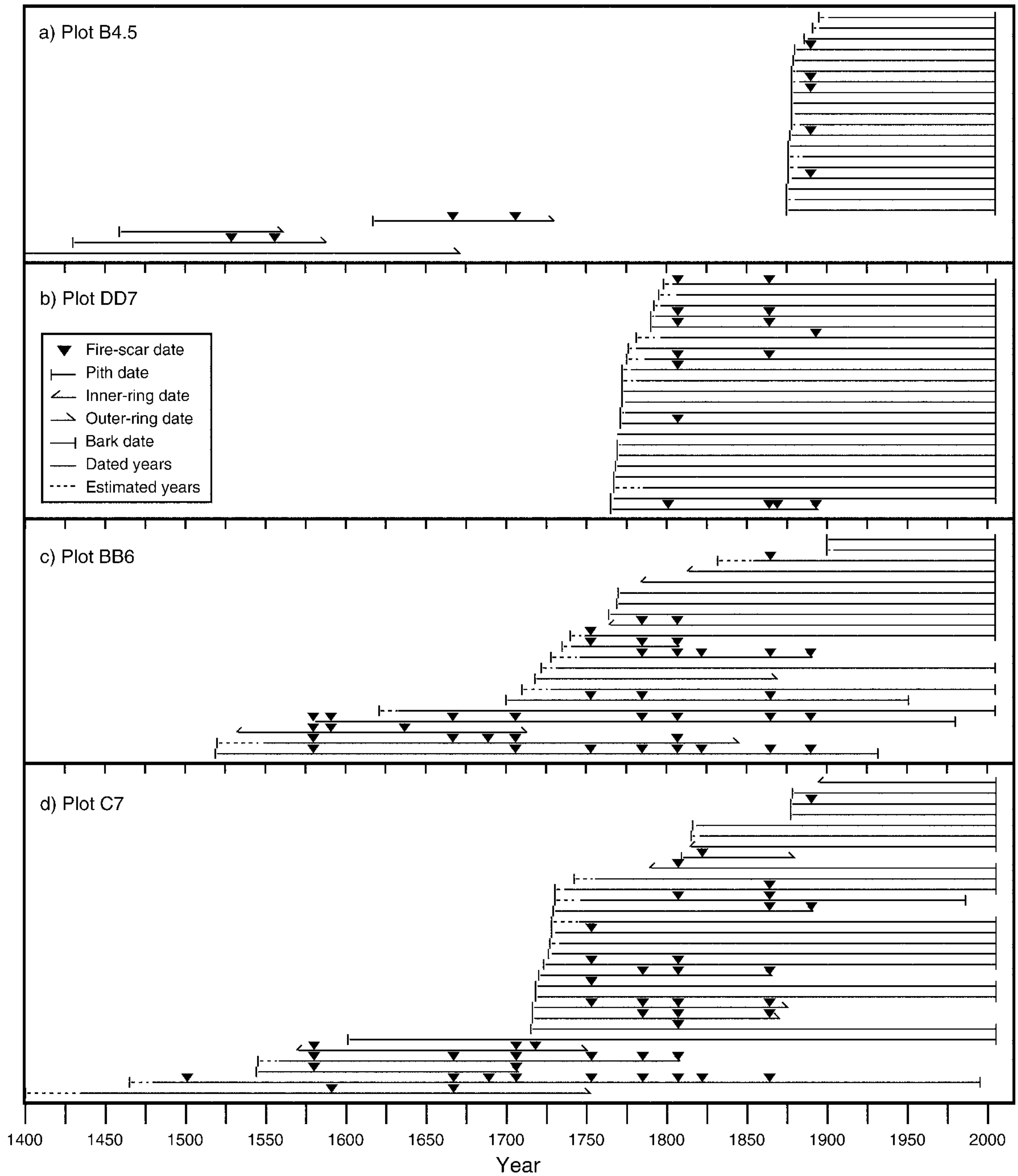

FIG. 2. Representative fire-demography diagrams for four plots. Time spans of individual trees are represented by horizontal lines with dates of fire scars represented by inverted triangles. Dated years (solid lines) were those actually present on the core or cross section, while estimated years (dashed lines) are those estimated to pith based on inner ring curvature. Pith dates are those either present on the core or cross section or that could be estimated with confidence based on inner-ring curvature. Inner-ring dates have an unknown number of rings to pith. Bark dates are those from living trees or from remnant trees for which we are confident of the death date. Outer-ring dates have an unknown number of rings to death date.

first scar intervals should be considered as valid fire-free intervals for assessing fire frequency in ponderosa pine forests. Here we refer to these as pith-to-first scar intervals because we did not make any corrections for sample height (10 cm) pith dates to probable germination (i.e., origin) dates. Finally, we use the 25% filtered landscape fire dates to determine fire rotations of surface fires across the landscape. Fire rotation is an area-based approach that is defined as the time it takes to burn an area the size of the study area (517 ha; Li 2002). We assume that landscape composite fire-scar dates recorded within a polygon are representative of the area of the entire polygon to estimate mean and range of surface fire rotations from 1539 to 1893.

Fire severity was estimated from recruitment and mortality patterns in plot FDDs (Fig. 2). Evidence of past crown fires was based on even-aged recruitment pulses of at least three trees that recruited within 20 years of each other and that either postdated death or outside dates of all previously existing trees (Fig. 2a) or were clumped at the start of an FDD chronology (Fig. 2b). Recruitment pulses were assumed to have originated after canopy mortality caused by crown fire, although other disturbances such as severe mountain pine beetle (*Dendroctonus ponderosae*) attacks, drought-caused tree mortality (e.g., Brown and Wu 2005, Brown 2006), or severe surface fires (that cause complete stem girdling and tree death) also could have been the cause of stand opening before a recruitment pulse. Patterns of even-aged recruitment seen in the contemporary age distributions are, of course, the most conservative evidence for past crown disturbances because older evidence is continually removed from the stand. Crown fires identified from the FDDs were dated to the 25% filter landscape composite fire date that occurred before the plot recruitment pulse. Multi-aged stands with continuous recruitment of individual trees or pulses that temporally overlapped existing trees (Fig. 2c, d) were considered to have no evidence of crown fire. Assignment of crown fires is equivocal in some plots that contain one to a few older trees that predate an even-aged pulse (Fig. 2d; see also Fig. 2 in Brown 2006 for similar FDDs from additional sites across the Black Hills). We assume that most of these pulses are not evidence of crown fire at the scales of plots in which we sampled (less than ~0.2 ha; Table 1), but instead resulted from a pulse of recruitment into what was an open canopy stand (i.e., that crown mortality was not necessarily a prerequisite for an even-aged pulse to have occurred; Brown and Wu 2005, Brown 2006). An exception was made if only one tree in the plot predated a recruitment pulse. In this case, the recruitment pulse was considered to have resulted from crown fire in an effort to be as conservative of the evidence as possible.

Fire sizes and relative spatial scales of crown and surface burning during individual fire years were estimated based on presence of fire scars or even-aged recruitment pulses within polygons. Only polygons where trees were alive or where a recruitment pulse was detected and dated to a fire year were included in fire size and crown-to-surface fire ratio estimates. We also calculated a separate fire rotation period for crown fires to compare to that for surface fires based on fire scars; i.e., how long did it take for crown fire to burn an area equal to that of the study area.

Stand structure

Tree densities (stems per hectare) and basal areas (square meters per hectare) were determined for each plot for both a reconstructed historical forest in 1870 and the current forest at the time of sampling in 2005. We used field measurements to determine current stand structure in 2005. We used both field and tree radial measurements and empirically derived allometric relationships following methods outlined by Brown and Cook (2006) to reconstruct stand structure in 1870 before initial Euro-American settlement.

Distances and azimuths of all sampled trees were measured from plot centers for assessment of stand structure. Radius of each plot was first calculated as the distance from plot center to center of the farthest tree sampled. Separate radii were calculated for the presettlement and smaller diameter trees collected in each plot. Plot areas were then determined as circular plots of the calculated radii (Table 1; Moore 1954, Lessard et al. 2002, Lynch and Wittwer 2003). Basal areas and numbers of living or recently dead (with bark or sapwood present) trees within plots were then scaled to determine the 2005 stand density and tree basal area on a per-hectare basis. Stem basal areas of living trees were determined from dbh measurements, while we converted dsh on recently dead remnants with bark using a linear regression equation derived from the dbh/dsh measurements on living trees: $\text{dbh} = \text{dsh} \times 0.8385$ ($n = 650$, $R^2 = 0.98$, $P < 0.001$). Recently dead remnant trees with only sapwood present were corrected to bark diameters using a separate regression from measurements on living trees: $\text{sapwood} + \text{bark diameter} = \text{sapwood diameter} \times 1.0937$ ($n = 480$, $R^2 = 0.59$, $P < 0.001$).

Historical basal areas and tree densities were reconstructed based on the number and sizes of trees alive in 1870 in each plot. Tree diameters in 1870 were either measured directly from radii on cores or cross sections that postdated 1870 or estimated based on empirically derived stem diameter conversion equations. Radial distance from pith to 1870 on measured radii was corrected to bark diameters using the formula: $([\text{radius} \times 2 \times 1.0937] \times 0.8385)$ (the first term for bark correction and the second for dsh to dbh conversion). A regression-derived conversion equation was then applied to remnant trees that did not postdate 1870 based on their state of decay and outside dates. Remnant trees with only heartwood present and outside dates (i.e., not death dates; Fig. 2) at or after 1800 were assumed to have been alive in 1870 (Brown and Cook 2006). For these trees, we used a conversion equation to estimate dbh in 1870 based on ratio of heartwood diameter (htd) to total tree diameter measured on living trees: $\text{dbh}_{1870} = \text{htd}^{0.389} \times 15.03$ ($n = 456$, $R^2 = 0.64$, $P < 0.001$; see also Brown and Cook 2006 for a similar regression for tree diameters in 1900). Remnant trees with outside dates before 1800 were assumed not to have been alive in 1870 and therefore not included in 1870 structural estimates. Trees and sizes present in each plot in 1870 were then scaled to determine the historical stand density and basal area on a per-hectare basis.

Fire behavior modeling

We used the NEXUS Fire Behavior and Hazard Assessment System (Scott and Reinhardt 2001) to model fire behavior based on reconstructed 1870 and current 2005 stand structures in each plot. NEXUS combines a surface fire spread model (Rothermel 1972) with crown fire models for transition from surface to crown fire (Van Wagner 1977) and crown fire spread (Rothermel 1991). Fires are broadly categorized as either surface or crown fires based on the dominant fuel stratum involved in burning, with crown fires further classified as either passive or active. Passive crown fires are often called "torching" fires in which surface fuels dominate fire spread but individual trees or patches of trees are killed by crown combustion in areas where tree or fuel density is greater. Active crown fires are those in which fire spread is primarily through crown fuels. Our objectives with this analysis were to assess the relative proportions in the historical and current forests of surface fire, possible transitional behavior between surface and passive crown fire, and active crown fire, and to estimate changes in the crowning index (CI), the wind speed at 6.1-m height above ground required to sustain active canopy burning.

Fire behavior modeled by NEXUS depends on three primary variables that describe fuel type and amount along with variables for topography and weather conditions (Scott and Reinhardt 2001). The fire behavior fuel model characterizes key controls on surface fire behavior, such as surface fuel bulk density and surface-to-volume ratio (Anderson 1982). We used fuel model 2 (timber, grass, and understory) for the historical forest, by assuming that grass and herbaceous understories were the primary fuels under the historical fire regime of frequent surface fires. This assumption is mainly based on historical photographs of Black Hills forests in the vicinity of Mount Rushmore (Grafe and Horsted 2002). For the current forest, we used fuel model 9 (long-needled pine litter), because needle litter is a much greater component than grass and herbaceous fuels today (C. L. Wienk, *unpublished fuels data*). The second fuel variable is canopy base height (cbh), which is a principal control for the transition from surface to canopy burning. We measured cbh as height to first live branch on all living trees in the field using a hypsometer. Heights were averaged by plot to derive mean cbh in the current (2005) forest. We then developed an empirical regression relationship between tree dbh and cbh to estimate cbh in the 1870 forest based on the reconstructed stem densities by size classes for each plot: cbh (m) = dbh (cm) $\times$ 0.193 ($n = 592$, $R^2 = 0.35$, $P < 0.001$). The final variable is canopy bulk density (cbd), the amount of canopy fuel available for burning during crown fires. We used an existing empirically derived allometric equation for ponderosa pine to estimate cbd based on plot basal areas and tree densities (Cruz et al. 2003): $\ln(\text{cbd}) = (0.435 \times \ln[\text{basal area}]) + (0.579 \times \ln[\text{tree density}]) - 6.649$. Basal areas (square meters per hectare) and tree densities (trees per hectare) in the 2005 forest were those measured from plots while we used reconstructed values for the 1870 forest. NEXUS also requires inputs for foliar moisture content, which was set at 100% for all runs, and slope, for which we used the plot field measurements (Table 1). Finally, we used FireFamily Plus (Bradshaw and Brittain 1999) to construct moderate and severe fire weather scenarios. Weather data from 1966 to 2004 from Custer, South Dakota, were used to determine the 80th (moderate) and 97th (severe) percentile weather conditions for 1 July through 30 September, the historical fire season (Table 2; Battaglia 2007). Custer is located ~18 km southwest of the Mount Rushmore study area.

TABLE 2. The 80th (moderate) and 97th (severe) percentile weather conditions from Custer, South Dakota (1964–2004), used for NEXUS fire behavior modeling.

Weather parameter	80th percentile	97th percentile
1-h fuel moisture (%)	6.6	4.4
10-h fuel moisture (%)	8.3	5.5
100-h fuel moisture (%)	11.4	8.2
Temperature (°C)	30.0	33.3
Maximum 1-minute wind speed (km/h)	21	30.6

Note: The "1-h" refers to 1-hour time-lag fuel moisture, etc.

RESULTS

Fire regime and forest history

A total of 907 trees (90.7% of all trees collected) were cross-dated. Trees that could not be cross-dated were primarily older remnant cross sections for which dating could not be found, trees with very tight ring series with numerous missing rings, or cross sections from remnants with too few rings (generally <100 rings) to be confident of cross-dating with the master chronology. Recruitment dates of plot trees and fire-scar dates from both plot and polygon trees are summarized in Fig. 3. Only pith dates are shown in Fig. 3 (i.e., excluding inside dates; see Fig. 2). A total of 802 trees (80.2% of all trees collected, 88.4% of cross-dated trees) had pith present on the increment core or cross section or for which a pith date could be estimated based on the inside ring curvature. Furthermore, a total of 728 trees (72.8% of all trees collected, 80.3% of cross-dated trees) either had pith or were within an estimated 10 years from pith. Fire scars were common on trees sampled systematically within plots in addition to polygon trees targeted for their fire-scar records (see also Fig. 2). Only two fire scars were recorded on trees after the last landscape fire (defined as those fire years recorded at ≥25% of plots) in 1893, both in 1912 (Fig. 3). Most fire scars were recorded as late-earlywood, latewood, or dormant season scars, with only one exception in 1652 when most fire scars were recorded as middle-earlywood.

Fire frequency varied strongly by spatial scale (Fig. 4, Table 3). Scar-to-scar intervals recorded on individual

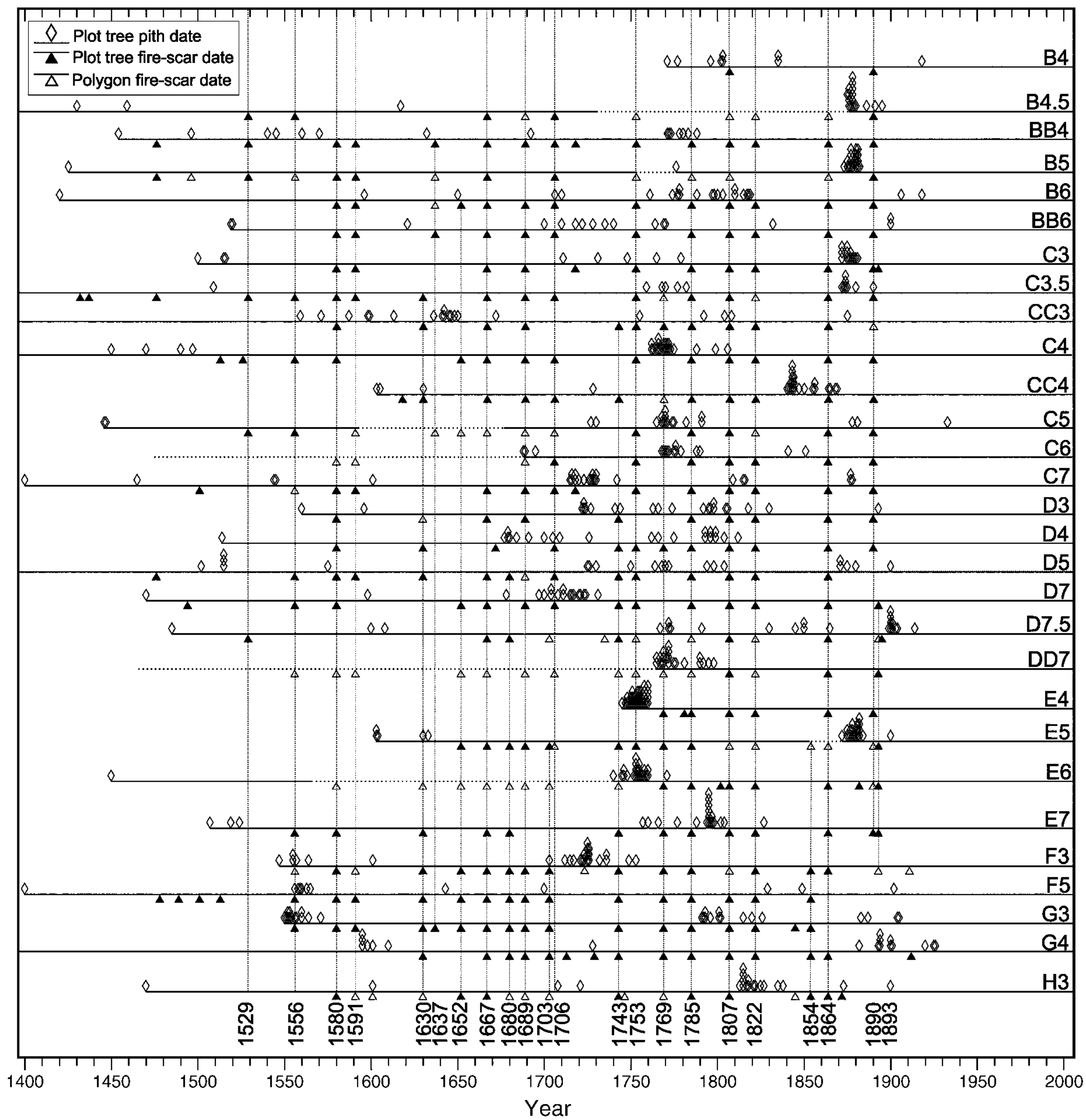

FIG. 3. Summaries of presettlement plot tree pith (recruitment) dates and fire scars for each plot and polygon. Solid lines are time spans encompassed by plot trees; dashed lines are additional time spans encompassed by polygon trees. Dates at the bottom of the panel are those recorded as fire scars in at least 25% of the plots or in which crown fire was reconstructed as having occurred before the date.

trees were heavily skewed to longer intervals, with a mean fire interval (MFI) of 34 yr and 20.5% of intervals >50 yr. Plot intervals composited on systematically collected trees within plots had an MFI of 27 yr with 12.7% of intervals >50 yr. Additional trees that were collected specifically for their fire-scarred records in polygons around plots reduced composited MFI slightly to 24 yr with 8.7% of intervals >50 yr. Compositing fire intervals across the landscape (fire dates recorded in at least two plots) further reduced the skew caused by longer intervals found in both plot and polygon composites and reduced MFI to 16 yr with a range of intervals from 3 to 39 yr. Filtering to 22 fire years that were recorded in at least 25% of the plots between 1529 and 1893 (bottom dates in Fig. 3) resulted in a MFI of 17 yr with a range of intervals again from 3 to 39 yr. Pith-to-first-scar intervals were the longest of all, with often extremely long intervals between the pith and when the first fire was recorded on a tree. The mean pith-to-first scar interval was 51 yr with 40.9% of intervals >50 yr and 14.1% of intervals >100 yr. Modal intervals based on 5-yr bins in Fig. 4 did not vary by spatial scale,

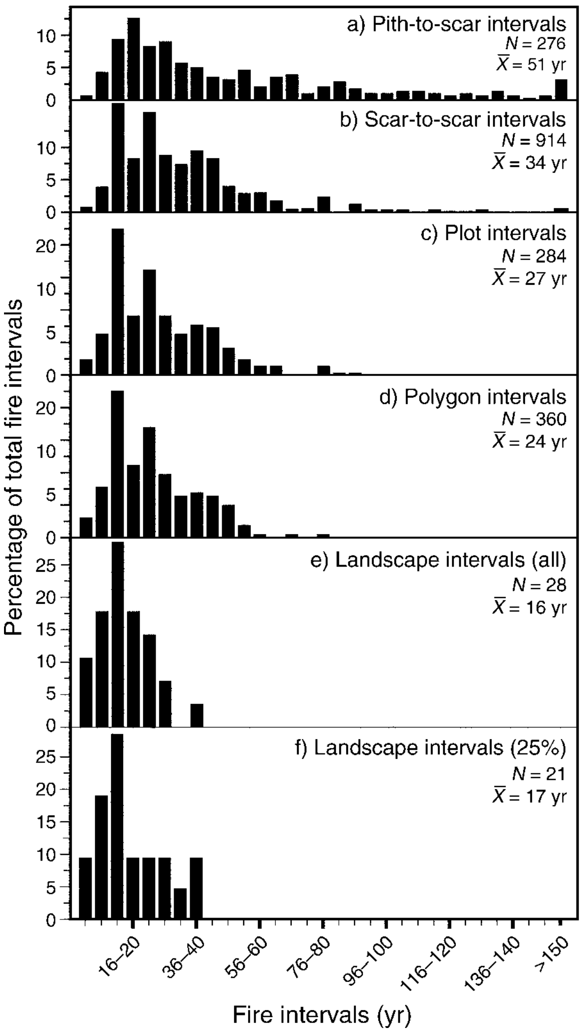

FIG. 4. Percentages of fire intervals by spatial scales. N is total number of fire intervals used in each panel (see also Table 3).

with the most common intervals ranging from 11 to 15 years (Table 3).

Even-aged recruitment pulses that fit criteria as evidence of crown fire were found in 10 plots during five fire years. Only 223 ha were estimated to have burned in crown fires out of a total of 6529 ha burned in 22 landscape fires between 1529 and 1893, or ~3.3% of the total area burned (Fig. 5). Crown fire occurred over relatively small portions of the total area estimated to have burned during most fire years, although crown fire in 1529 was estimated to have burned at least 66 ha while surface fire burned an additional 80 ha. Area burned in surface fires during the landscape fire years resulted in an estimate of surface fire rotation of 30 yr with a range from 15 to 51 yr from 1529 to 1893. Area burned in crown fires resulted in a crown fire rotation of 846 years (223 ha burned during the 365-yr period between 1529 and 1893 across the 517-ha landscape).

Stand structure

Tree densities and basal areas (BA) within plots varied considerably in the historical forest but were on average significantly lower ($P < 0.01$ in Student's t test) than in the current forest (Fig. 6a, Table 4). Average BA in the current forest was measured to be 30.7 m^2/ha and estimated in the historical forest to be 23.6 m^2/ha. Canopy base height (cbh) also was significantly higher in the historical than current forest (Fig. 6b, Table 4). Mean cbh for all plots was measured to be 4.2 m in the current forest and estimated at 6.1 m in the historical forest. Differences in structure were primarily due to increases in smaller diameter trees in the current forest, although there also has been a loss of larger trees relative to the historical forest (Fig. 7). Tree density changed from an average of 280 trees/ha in the historical forest to 1309 trees/ha currently, mainly due to a large increase in the numbers of trees <20 cm dbh.

Fire behavior modeling

Surface and canopy fuel structures in the 1870 forest would have supported mainly surface fires under both moderate and severe weather conditions according to NEXUS model output (Fig. 8). Only one plot in the historical forest contained crown structure that would have been able to support crown burning during moderate weather conditions. The possibility of crown fire in the historical forest rises to five plots under severe weather conditions. In contrast, almost one-half of the

TABLE 3. Measures of fire frequency at Mount Rushmore.

Measure	Pith-to-scar	Scar-to-scar	Plot composite	Polygon composite	Landscape composite (all)	Landscape composite (25%)
No. intervals	276	914	284	360	28	21
Mean (yr)	51	34	27	24	16	17
SD (yr)	46	25	16	14	8	10
Median (yr)	35	26	22	22	15	15
Mode (yr)	16–20	11–15	11–15	11–15	11–15	11–15
Skewness	2.05	2.59	1.18	0.99	0.75	0.66
Range (yr)	4–298	1–251	3–87	1–76	3–39	3–39

Notes: Landscape composite intervals (all) are intervals between any fire that was recorded in at least two plots. Landscape composite intervals (25%) are those recorded in at least 25% of plots between 1529 and 1893 (dates at the bottom of Fig. 3). The mode is based on 5-yr bins shown in Fig. 4.

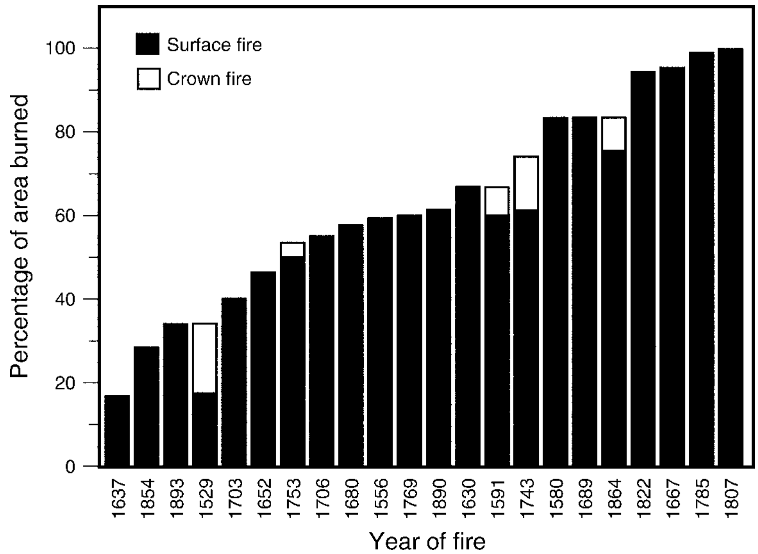

FIG. 5. Relative percentages of the Mount Rushmore landscape (517 ha) estimated to have burned in 22 landscape fire years (recorded at ≥25% of plots; bottom dates in Fig. 3) by both crown and surface fire. Fire years are arranged in order of estimated total size.

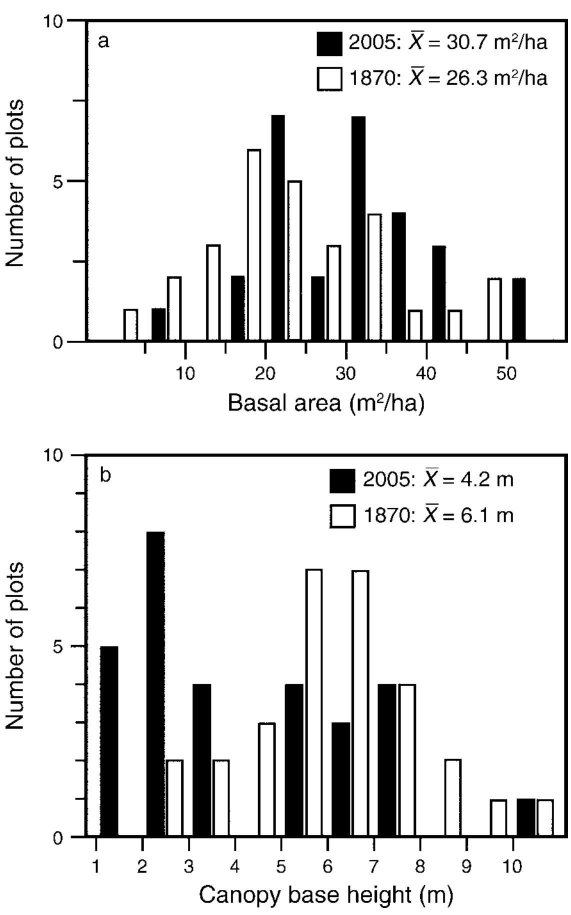

FIG. 6. (a) Plot basal areas and (b) canopy base heights reconstructed in 1870 and measured in 2005.

plots in the current forest have fuel structures that could support crown fire under moderate weather conditions, and most could burn by active crown fire under severe weather conditions. The shift in susceptibility to crown fire is reflected in a downward shift in the crowning index (CI) from historical to current conditions in almost all plots (Fig. 9). Only one plot that had been both intensively logged in the early settlement period and recently thinned was estimated to have had a large upward shift in CI from historical to current conditions.

DISCUSSION

Did Mount Rushmore experience a variable-severity fire regime?

Baker et al. (2007) outline a variable-severity fire regime model for northern ponderosa pine forests, including the Black Hills, in which "most of the landscape historically experienced or [was] capable of supporting high-severity fire and most stands (i.e., 1–100 ha areas of forest) have evidence of mixed- or high-

TABLE 4. Forest structural elements reconstructed for 1870 and measured in 2005.

Year	Trees (no./ha)	BA (m^2/ha)	CBH (m)	CBD (kg/m^3)
1870	280 (36)**	23.6 (2.2)**	6.1 (0.3)**	0.12 (0.01)**
	37 to 711	3.6 to 47.6	2.6 to 10.1	0.03 to 0.29
2005	1309 (288)**	30.7 (2.7)**	4.2 (0.5)**	0.30 (0.04)**
	79–7434	10.4–53.9	1.4–10.0	0.04–0.89

Notes: BA is basal area, CBH is canopy base height, and CBD is canopy bulk density. All measurements are for ponderosa pine. Standard errors of the mean are in parentheses, and ranges are underneath. Asterisks mark significant differences in *t* tests between column means ($P < 0.01$).

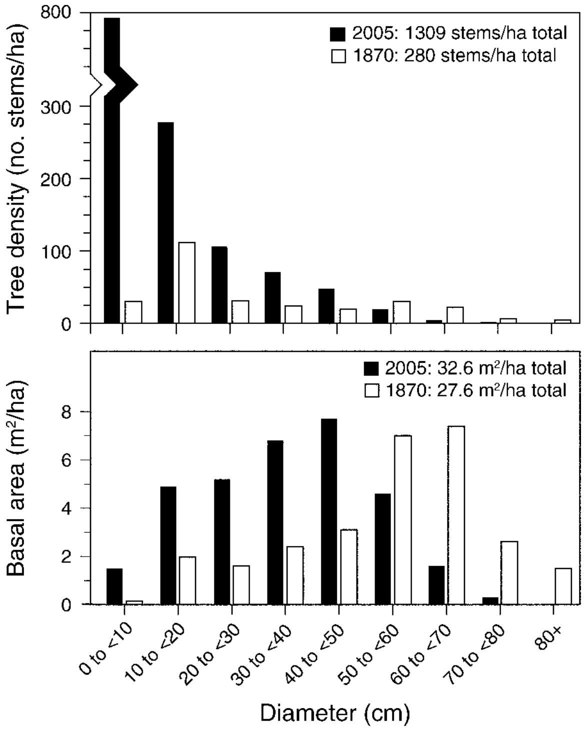

FIG. 7. (a) Total tree densities and (b) basal areas by dbh size classes reconstructed in 1870 and measured in 2005.

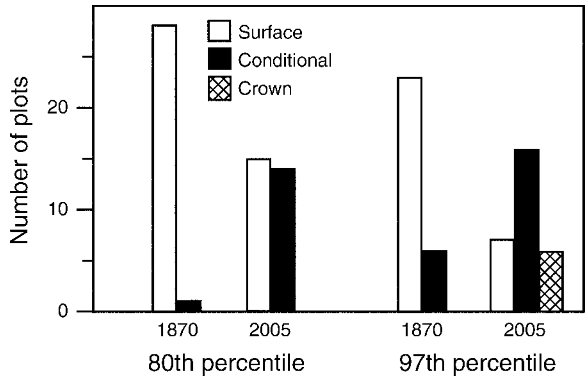

FIG. 8. Fire behavior estimated from NEXUS output by plot based on reconstructed 1870 and measured 2005 stand structures for moderate and severe weather scenarios (Table 2). "Conditional" is when passive crown fire conditions are present in stand structure but crown fire initiation depends on more intense surface burning or by crown fire burning from an adjacent stand (Scott and Reinhardt 2001). "Crown" is when stand conditions would support active canopy fire.

severity fire over the last few centuries." Although we found evidence of crown fires in the historical forest at Mount Rushmore, we found no evidence to support Baker et al.'s definition for a variable-severity fire regime. Most stands do not have evidence of high-severity fire (Figs. 3 and 5) and most, at least in the forest present in 1870, were not capable of supporting high-severity fire (Fig. 8).

The historical fire regime from 1529 to 1893 at Mount Rushmore is best characterized as one of low-severity surface fires with occasional (>100 years) patches (<100 ha) of passive crown fire. The ubiquity of fire-scarred trees documents that surface fires were a prevalent disturbance agent in the historical forest for at least several centuries before 1893 (Figs. 2 and 3). We estimate that only ~3.3% of the landscape burned in crown fires between 1529 and 1893 (Fig. 5). NEXUS modeling also suggests that surface fires would have dominated fire behavior in the 1870 forest during both moderate and severe weather conditions (Fig. 8). Based on the fire rotations we reconstructed, between 1529 and 1893 surface fire would be expected to burn an area the size of the Mount Rushmore landscape once every 30 years, while it would take over 800 years to burn the same amount of area by crown fire.

Defining a variable-severity fire regime in ponderosa pine and related forests must focus primarily on questions of scale: how much area and how often did crown fire burn relative to surface fire? It is likely that most (if not all) individual fires were variable severity at some spatial scale, ranging from single trees or small patches of trees killed by passive crown fire to larger patches possibly killed by active crown fire. Furthermore, even in stands that may have originated after crown fires, surface fires still dominated over longer time periods (e.g., Fig. 3). Fire ecologists often assign variable- or mixed-severity fire regimes to specific forest types or landscapes but typically without any accompanying specification of the spatiotemporal scales of crown-to-surface burning (e.g., Agee 1993, Morgan et al. 1996, Brown 2000, Schoennagel et al. 2004, Baker et al. 2007). However, simply to describe a historical fire regime as variable severity is by itself not useful either for characterizing fire as an ecological process or for fire

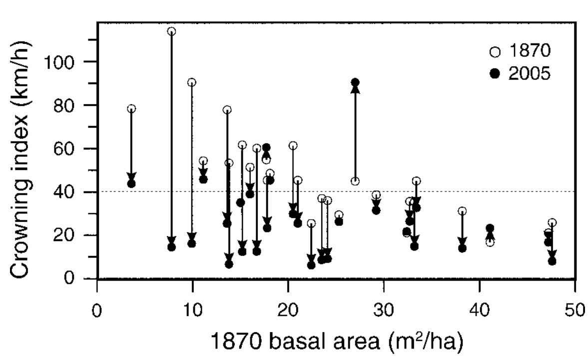

FIG. 9. Changes in the crowning index (CI; the 6.1-m height wind speed required to sustain active canopy burning) from reconstructed stand structure in 1870 to current structure in 2005 for each plot. Plots are arranged in order of their reconstructed 1870 basal areas. Arrows indicate direction of change in CI from 1870 to 2005. The dotted line is at 40.2 km/h, which is considered to be the CI below which stands are at high risk of crown fire (Fiedler et al. 2004).

management or ecological restoration purposes. For example, without reference to scale it is possible to conclude that recent variable-severity fires in ponderosa pine forests (i.e., that have included both surface burning as well as large areas of crown mortality) are within a historical range of variability even though areas of crown mortality are orders of magnitude larger than any area that occurred historically (e.g., Romme et al. 2003). We propose that future definitions of variable-severity fire regimes in ponderosa pine and related forests must be accompanied by descriptions of the maximum spatial extent and how often crown fire occurred over a defined period of time. Furthermore, fire ecologists must define whether a variable-severity fire regime included active crown fire or only passive crown fire in which fire spread was primarily through surface fuels. This would provide additional information about the dominant fuel stratum involved in fire spread, information that is critical to understanding fire as an ecological process affecting forest structure and ecosystem function (e.g., Agee 1993).

How often did fires occur at Mount Rushmore?

Surface fires ceased after 1893 probably initially as a result of domestic livestock grazing that reduced fine fuel mass and continuity in and around the Mount Rushmore area (Fig. 3; e.g., Swetnam and Betancourt 1998, Brown and Sieg 1999). Mean fire intervals before 1893 varied by spatial scale, with the longest recorded as scar-to-scar intervals on individual trees (Fig. 4b, Table 3). Long scar-to-scar intervals resulted from three possible factors: (1) spatial heterogeneity in burning at the scale of individual trees; (2) burning that was not recorded as a fire scar; and (3) fire scars burned off or eroded after initial formation. This latter factor is likely especially prevalent when relying on often heavily decayed logs, snags, and stumps for fire-scar records such as we did in this study. Scar-to-scar intervals thus result both from actual fire-free periods as well as "missing" scars either from unrecorded fires or those lost from fire-scar sequences before or during sampling.

Because scar-to-scar intervals on individual trees are equivocal as to whether they represent true fire-free periods, fire-scar studies have generally relied on composited fire dates from multiple trees in a stand or landscape to assess fire frequency (e.g., Dieterich and Swetnam 1984, Swetnam and Baisan 1996, Falk 2004). At Mount Rushmore, compositing by systematically (i.e., randomly) sampled plot trees removed the longest intervals found on individual trees and reduced the mean fire interval by ~21% (34 yr to 27 yr; Fig. 4c, Table 3). A problem with compositing is that as the area sampled or number of samples included in the composite increases, the mean fire interval generally decreases because more small fires (less than the size of the area composited) may be found (Falk 2004). However, compositing by plot is a less biased estimate of point fire frequencies than individual trees because of the likelihood of missing fire scars. Plot sizes varied but all were less than ~0.2 ha (Table 1), which should be considered as the minimum area over which a reliable estimate of a point fire frequency is possible at Mount Rushmore (see also Falk 2004). However, because plot trees were randomly selected, there may still be missing fire dates in the plot composites, i.e., fires that burned in the plot but were not recorded by any of the plot trees. Thus, additional trees targeted specifically for their fire-scar records sampled outside of plots reduced the mean interval slightly to 24 yr (Fig. 4d, Table 3). Most of the polygon trees were sampled within ~80 m of the plot centers (an area of ~2 ha). Moving up to the landscape scale and compositing to fires that occurred in at least two plots removed more of the longest intervals and reduced the mean fire interval to 16 years (Fig. 4e). The largest landscape fires (those that occurred in at least 25% of plots) burned on average every 17 years (Fig. 4f). Recent empirical studies that have looked at spatial scales of burning have found that the 25% filtered landscape fire-scar dates are robust estimates of fire frequency across larger areas because they remove the effect of smaller fires (Falk 2004, Van Horne and Fulé 2006). This number and its range (Table 3) should be considered as the most robust for how often fires should burn somewhere within the Mount Rushmore landscape during any future restoration effort.

Pith-to-first scar intervals have an even longer "tail" in their interval distribution than scar-to-scar intervals (Fig. 4a). Similar to scar-to-scar intervals, these also represent both true fire-free intervals as well as intervals with missing scars, but they also reflect physiological changes in the susceptibility of trees to scarring as they age. Mature ponderosa pine trees are well adapted to surface fires, with thick bark that protects cambial tissues during burning. Unless a tree was scarred early in its life, it may have been several decades later (and multiple fires that did not scar the tree) before an initial scar was formed. Initial scarring on older trees is undoubtedly a highly contingent event in a tree's life, possibly resulting from a buildup of heavier fuels next to the stem (for example, from falling branches or death of a neighboring tree) or when fire otherwise burns hot or long enough to kill the cambium. Once scarred, however, ponderosa pine trees tend to be very susceptible to further scarring during subsequent fires because the bark is thinner on the rapidly growing woundwood formed at the margins of previously killed cambium (Smith and Sutherland 2001). Almost all fire history studies have avoided using pith- (or origin-; Baker and Ehle 2001) to-first scar intervals because of their ambiguity as true fire-free intervals (see also Van Horne and Fulé 2006). Our data document that many pith-to-first scar intervals are longer than even scar-to-scar intervals on individual trees, and support the likelihood that these are not valid fire-free intervals to include in estimates of fire frequency.

In contrast to mean or median fire intervals, other measures of central tendencies in fire interval distributions also should be considered when estimating how often fires burned at a point or across a landscape. We believe this is the first study that has reported the mode in interval distributions (Table 3). Although the mean, median, and range of intervals varied from single trees to the entire landscape, the mode did not. The most common fire intervals regardless of scale were from 11 to 15 years (Fig. 4). This lack of variation with scale suggests this is the most robust range of intervals to target for restoration of surface fires across all spatial scales on the Mount Rushmore landscape.

How has forest structure changed, and what does this mean for fire severity?

There was considerable structural heterogeneity in the historical forest, ranging from open to dense stands of trees (Fig. 6a). Similar structural heterogeneity was found by Brown and Cook (2006) at both stand and landscape scales in ponderosa pine forests across the Black Hills. However, despite the presence of dense stands historically, significant structural changes have occurred from the presettlement forest in 1870 to the current forest in 2005 (Table 4). In general, there has been a shift in density from relatively open to relatively closed stands across the range of variation in basal areas (Fig. 6a). Changes in tree density were primarily the result of increases in smaller diameter trees (Fig. 7a). Much of the Memorial currently contains "dog-hair" stands of very slow-growing and crowded younger trees, similar to the situation found in many southwestern ponderosa pine forests (e.g., Allen et al. 2002). However, changes in basal area also were the result of losses of larger diameter trees across the landscape (Fig. 7b). Most of the basal area of the historical forest was in larger trees while that of the current forest is mostly in small- to medium-size trees. Loss of larger trees was due both to timber harvest that took place in parts of the Memorial as well as recent death of older trees. Many of the oldest trees we sampled were killed in recent decades by mountain pine beetles, as evidenced by the presence of blue stain in the sapwood. Although we have no evidence of the magnitude of bark beetle mortality in the historical forest, we suspect that many of the older trees that recently died were more susceptible to successful beetle attacks owing to stress caused by asymmetric competition with dog-hair trees that have established over the past century of fire exclusion (Biondi 1996).

Increased tree density and the presence of many younger trees with lower crown base heights have greatly increased the likelihood of both passive and active crown fires in the current forest (Figs. 8 and 9). Structural conditions in the reconstructed 1870 forest suggest that only one stand was capable of supporting conditional crown burning during moderate weather conditions, with four additional plots capable of supporting crown burning during severe weather events (Fig. 8). The majority of the forest in 1870 was not at high risk for crown fire while the majority of the current forest is (Fig. 9). Spatial and temporal heterogeneity in historical stand conditions was likely the cause of variation in fire severity through time (Fig. 5). Denser patches of trees in a matrix of open- to medium-density forest would have been occasionally able to "torch out" during strong winds or when burning conditions were otherwise conducive to the occurrence of passive crown fires. However, even when such events occurred, fire spread across the landscape still would have primarily been through surface fuels, whereas current forests are capable of supporting active crown fires in which fire spread would be through aerial fuels (Fig. 8).

We attempted to be both objective and thorough in reconstructing the structure of the 1870 forest by collecting both living and remnant trees in systematically sampled plots. However, the results shown in Figs. 8 and 9 should be considered as conservative evidence for potential crown fire incidence in the 1870 forest because it is possible that smaller trees that were present in 1870 have since died and decayed before we were able to sample them. Decay of smaller trees would depend on how much heartwood was present in the stem. Sapwood decays relatively rapidly (within a few decades of tree death) and if the tree was small enough not to have much heartwood we would not have been able to account for it when determining historical stand structure (Fig. 7). Thus, the trajectories in crowning index shown in Fig. 9 should be considered as potential ranges of fire behavior in each plot. The 1870 crowning index can be viewed as the most likely of probable plot-level fire behavior based on all available dendroecological evidence, but there is some decreasing likelihood that the real CI lies on the range of the trajectory to current conditions depending on how much of a "missing" component of smaller trees there may have been in the plot. However, we strongly suspect that the model results for the 1870 forest are correct or nearly so, because the fire history data also document that surface burning dominated fire behavior during historical fires (Figs. 3 and 5) and provide additional support for NEXUS model results. Taken together, the fire history and model results lead to a similar conclusion that dominant fire behavior in the Mount Rushmore forest has shifted from surface to crown fire during the period of fire exclusion from the late 19th to early 21st centuries. However, future efforts using NEXUS modeling may further refine the uncertainties in structural conditions and fire behavior in the historical forest. For example, NEXUS modeling could be used to develop a distribution of crowning index with varying crown base heights and bulk densities instead of a single number based on surviving trees. Such an approach would allow more robust statements about historical landscape fire behavior.

What do the historical data mean for ecological restoration at Mount Rushmore?

Historical data from Mount Rushmore strongly support the need for restoration of both forest structure and surface fire regimes to reduce the scale and likelihood of crown fire during future wildfires and to restore related ecosystem conditions and processes. Brown et al. (2004) outline four factors to increase fire resilience in fire-adapted ecosystems: reduce surface fuels, increase height to live crown, decrease crown density, and keep the larger trees of fire-resistant species. Changes in canopy structure, especially the lowering of canopy base heights, will require that stands be initially thinned to restore structure to the point that fire severity can be better controlled in prescribed fires. Thinning of most patches of trees 20 cm dbh and smaller will begin to restore canopy structure and basal area to a historical configuration (Fig. 7a). Unharvested stands contain many older trees from the presettlement forest, and these should be maintained in the restored forest as much as possible. Much of the existing forest has high ecological integrity because it has not been harvested (Symstad and Bynum 2007), and we expect that this will only be enhanced once a restoration program is in place.

There is increasing recognition of the primary importance of "place" to inform ecological restoration efforts in fire-adapted ecosystems (Allen et al. 2002, Brown et al. 2004, Schoennagel et al. 2004, Noss et al. 2006). Studies of local fire regimes and forest history are critical both to assess need for intensive restoration treatments such as thinning or prescribed fire and for predicting success in restoring an ecosystem to a semblance of its historical condition. However, Falk (2006) also suggests restoration efforts must focus on the ecological processes that sustain and characterize ecosystem function rather than concentrating solely on local ecosystem patterns produced by those processes (see also Stephenson 1999). Viewed from a process-centered perspective, the local fire history may be seen as only one realization of multiple stochastic and deterministic processes that affected a particular configuration we see or reconstruct today (Lertzman et al. 1998, McKenzie et al. 2006). The true focus for restoration ecology therefore should be to characterize underlying ecosystem dynamics that affected both the specific realization that we are able to reconstruct as well as the range of variation possible within a particular ecosystem type.

Restoration efforts in ponderosa pine ecosystems have been largely, albeit not completely, guided and justified by studies from the southwestern United States (e.g., Covington and Moore 1994, Fulé et al. 1997, Moore et al. 1999, Allen et al. 2002, Friederici 2003). Frequent surface fires primarily affected by seasonal droughts promoted and maintained mostly low-density stands of multi-aged trees through density-independent control on seedling recruitment (Brown and Wu 2005). The ponderosa pine forest at Mount Rushmore did not burn as often as many in the Southwest (e.g., Swetnam and Baisan 1996), but the overall effect of repeated, episodic surface fires on forest structure and subsequent fire behavior was similar. Diversely structured stands of multi-aged trees resulted from spatially and temporally episodic recruitment and mortality (Figs. 3 and 6; see also Brown 2006). Scorching of lower branches by surface burning resulted in higher canopy base heights in mature trees and less susceptibility to passive crown burning (Fig. 6b). Furthermore, again similar to southwestern ponderosa pine forests, the cessation of episodic surface fires at Mount Rushmore beginning in 1893 has resulted in unchecked tree recruitment with concurrent increases in stand densities (Fig. 6a), a lowering of stand-level canopy base heights (Fig. 6b), increased numbers of small trees (Fig. 7a), and an overall increase in crown fire potential (Figs. 8 and 9) in current forests. Exclusion of surface fires also has undoubtedly had cascading effects on related ecosystem processes, such as decreased understory species density and diversity because of less light penetration through the canopy, deeper litter and duff layers, and increased competition for soil water and nutrients with trees.

A process-centered restoration approach defines and uses historical dynamics in both biotic and ecosystem processes as central foci for restoration design and implementation (Falk 2006). In ponderosa pine forests throughout their range, the central ecological theme is that of episodic surface fire. Although some ponderosa pine forests apparently experienced passive and perhaps active crown fires in the past, the relative scale of crown-to-surface burning is often based on questionable evidence, difficult to quantify even when adequate evidence is available, and, perhaps more critically, difficult to address in restoration programs. In most cases, land managers and restoration ecologists attempting to restore fire to fire-adapted ponderosa pine ecosystems are simply trying to reintroduce surface fires for the first time in over a century, and are not able to accommodate "prescribed crown fires" at this point in time. Furthermore, crown and variable-severity fires will undoubtedly continue to occur in future wildfires. Funding and management policies for landscape-scale restoration programs in ponderosa pine forests are currently not in place, and large areas of dense, dog-hair stands of trees that are highly susceptible to crown fires will undoubtedly continue to persist in most ponderosa pine landscapes well into the future (Stephens and Ruth 2005, Donovan and Brown 2007). Yet what is currently missing from virtually all ponderosa pine forests throughout their range in western North America are surface fires, especially those that are allowed to burn across large areas. We know with great certainty that such fires occurred with regularity in the past. Thus, what is needed for management are concerted efforts to identify and restore ponderosa pine forests (such as at Mount Rushmore) that have much of their presettlement structure intact, and can serve as reference landscapes for ponderosa pine ecology (Donovan and Brown 2007, Millar et al. 2007). Restoration of surface fires and forest structure at Mount Rushmore will not only restore structure, function, and resilience to this uniquely valuable old-growth ponderosa pine ecosystem but also reduce the adverse effects of a future wildfire on the world-renowned Mount Rushmore sculpture.

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