Mixed-Severity Fire Regime in a High-Elevation Forest: Grand Canyon, Arizona

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Executive Summary

Fire regime characteristics of high-elevation forest on the North Rim of the Grand Canyon, Arizona, were reconstructed from fire scar analysis, remote sensing, tree age, and forest structure measurements, a first attempt at detailed reconstruction of the transition from surface to stand-replacing fire patterns in the Southwest. Tree densities and fire-/non-fire-initiated groups were highly mixed over the landscape, so distinct fire-created stands could not be delineated from satellite imagery or the oldest available aerial photos. Surface fires were common from 1700 to 1879 in the 4,400 ha site, especially on S and W aspects. Fire dates frequently coincided with fire dates measured at study sites at lower elevation, suggesting that pre-1880 fire sizes may have exceeded 50,000 ha. Large fires, those scarring 25% or more of the sample trees, were relatively infrequent, averaging 31 years between burns. Four of the five major regional fire years occurred in the 1700's, followed by a 94-year gap until 1879. Fires occurred in significantly dry years and reconstructed Palmer Drought Stress Index values indicated severe drought in major regional fire years. Currently the forest is predominantly spruce-fir, mixed conifer, and aspen. In contrast, dendroecological reconstruction of past forest structure showed that the forest in 1880 was very open, corresponding closely with historical (1910) accounts of severe fires leaving partially denuded landscapes. Age structure and species composition were used to classify sampling points into fire-initiated and non-fire-initiated groups. Tree groups on nearly 60% of the plots were fire-initiated; the oldest such groups appeared to have originated after severe fires in 1782 or 1785. In 1880, all fire-initiated groups were less than 100 years old and nearly 25% of the groups were less than 20 years old. Non-fire-initiated groups were significantly older (oldest 262 years in 1880), dominated by ponderosa pine, Douglas-fir, or white fir, and occurred preferentially on S and W slopes. The mixed-severity fire regime, transitioning from lower-elevation surface fires to mixed surface and stand-replacing fire at higher elevations, appeared not to have been stable over the temporal and spatial scales of this study. Severe fires were not unprecedented in the pre-1880 forest, but the exclusion of frequent surface fires in the mixed conifer and ponderosa pine forests has led to a more homogenous landscape where the extent and severity of contemporary fires may exceed pre-1880 patterns. Information about historic fire regime and forest structure is valuable for managers but the information is probably less specific and stable for high-elevation forests than for low-elevation ponderosa pine forests.

Keywords

Kaibab Plateau, fire scars, age structure, *Picea*, *Pinus*, *Abies*, *Pseudotsuga*, *Populus*

Introduction

Southwestern forests primarily consist of species adapted to relatively xeric conditions in which surface fires recur frequently. Mesic forests are restricted to narrow strips and islands on high mountains and plateaus (Gosz 1992), where typically moist conditions keep most fires small in size (Kipfmueller and Baker 2000). However, living and dead fuels can accumulate to high levels in mesic forests (White and Vankat 1993), eventually supporting intense fire behavior during a dry period. Aspen and Engelmann spruce/subalpine fir stands are commonly considered to have originated following sporadic severe fires. Intervals between severe fires may have been as short as 50 years in aspen stands (Jones and DeByle 1985), while the oldest stands in a spruce-fir chronosequence in Colorado were fire-free for up to 600 years (Aplet et al. 1988). Aspen also provide suitable regeneration habitat for shade-tolerant conifers (Shepperd and Jones 1985), so the long-term replacement of aspen by conifers during the twentieth century has been typically attributed to fire exclusion (Dahms and Geils 1997).

Where fire regimes include both surface and canopy burning, Johnson and Gutsell (1994) proposed a combination of research techniques including fire-scar dates, age structure, and species composition of fire-initiated stands. These methods are invaluable for assessing long-term fire patterns but uncertainties in interpretation are associated with fire scars (precisely dated but representing points on the landscape) as well as with age/species data (imprecision in stand boundaries, fire survivors within stands, imprecision of age data, immediacy of post-fire colonization, and interpretation of successional pathways) (Johnson and Gutsell 1994, Swetnam and Baisan 1996, Swetnam et al. 1999, Baker and Ehle 2001). Recent studies in forests with mixed burning patterns have emphasized different aspects of methodological certainty. For instance, Kipfmueller and Baker (2000) calculated fire rotation (FR, the time required to burn an area equivalent to 100% of the study site), MFI, and mean point fire intervals (a per-tree measure of average intervals between fire scars) in a Wyoming lodgepole pine forest, suggesting that the composite MFI overestimated fire occurrence. Minnich et al. (2000) reconstructed twentieth century fire history in a Baja California mixed conifer forest from aerial photography, arguing that infrequent, severe fires regulated forest structure, rather than the more-frequent surface fires detected from fire scars. Taylor (2000) suggested a closer integration between MFI and FR data along a gradient from low-elevation Jeffrey pine to high-elevation mesic forests by showing that the mean point fire interval corresponded closely with FR values.

The fire ecology of mesic forest types has received extensive study in the north (e.g., Johnson and Gutsell 1994) and California. But the role of fire disturbance in high-elevation forests of the Southwest has received relatively little attention compared to ponderosa pine and lower mixed conifer forests (Swetnam and Baisan 1996a), perhaps because of the relative scarcity of spruce, fir, and aspen forests in this region. Only two studies out of 63 compiled by Swetnam and Baisan (1996a) reported possible stand-replacing fires in high-elevation forests. Swetnam and Baisan (1996b) and Swetnam et al. (2001) found that frequent but isolated fires in the Animas Mountains of SW New Mexico were replaced at intervals by synchronized burning across several study sites, suggesting "high-intensity" burns that may have been analogous to large modern wildfires. Grissino-Mayer et al. (1995) found relative frequent fire recurrence (mean fire interval [MFI] of fires scarring $\geq$ 10% of the sample trees = 6.2 years, period 1575-1880) in mixed conifer forests above 2,750 m elevation in the Pinaleño Mountains, immediately below an Engelmann spruce/corkbark fir stand with several trees > 300 years old, interpreted as regeneration from a stand-replacing fire in 1685. In a similar example of abrupt differences in fire regime over a short distance, Stephens (2001) reported a (MFI) of 9 years in a Jeffrey pine stand and nearly 25 years in a fir-dominated stand located only 100 m away.

Grand Canyon National Park, in northern Arizona, is a useful site to assess the transition from surface to stand-replacing fires in a southwestern ecosystem. Frequent surface fires recurred throughout ponderosa pine and lower mixed conifer forests of the North Rim on the Kaibab Plateau until *circa* 1880 (Wolf and Mast 1998, Fulé et al. 2000). At higher elevation, White and Vankat (1993) used tree age data to show increased spruce-fir forest density since 1880. The forests of the North Rim have never been harvested and livestock grazing was excluded around 1938, affording a regionally unique opportunity to assess landscape-scale forest attributes over an elevational gradient in a near-natural setting (Fulé et al. 2002, Moore and Huffman in review).

Building upon previous work in lower elevation Grand Canyon forests (Fulé et al. 2000, 2002), we selected a large landscape crossing the highest elevation of the park to meet several objectives: (1) reconstruct fire regime characteristics across the transition from surface to stand-replacing fires, using multiple methods (fire scars, tree age and species, spatial patterns of forest stands); (2) determine the relationship between high-elevation fire occurrence and fire patterns in surrounding low-elevation sites, especially in terms of climatic influence on fire; and (3) assess changes in fire regime and forest structure since 1880, the date of initiation of fire exclusion across the Kaibab Plateau.

Methods

Study Area

The study site was the upper mixed conifer and spruce-fir forests covering ~ 4,400 ha at Little Park, comprising the highest elevations on the Kaibab Plateau (to 2,794 m). This high-elevation site completed a transect of study sites described by Fulé et al. (2002). The lower transect ranged from Powell Plateau (PP), Fire Point (FP), and Rainbow Plateau (RP), each ~2,300 m elevation, through Swamp Ridge (~2,500 m). While the complete transect ran west-east along the northern border of Grand Canyon National Park, the prevailing fire season winds are from the southwest. Therefore we added a new study site, Galahad Point, at the base of Kanabownits Canyon (~2,350 m elevation). The Galahad site lay directly downcanyon and upwind from Little Park, allowing us to test whether fire dates differed among low-elevation sites southwest vs. due west of the high-elevation site. Figure 1 shows the topography and location of all the sites.

Average annual precipitation at the North Rim ranger station (elevation 2,542 m) is 58.4 cm, with an average annual snowfall of 328 cm. Temperatures range from an average July maximum of 26° C to an average January minimum of -2° C (White and Vankat 1993). Soil information was derived from an ongoing soil survey (A. Dewall, National Resource Conservation Service, personal communication 2002). Soils at the Galahad site were tentatively classified as the Elledge Family (fine, mixed, superactive, mesic Typic Paleustalfs). Approximately 50% of the BSLP study site has been mapped. Soils on 15-40% slopes were tentatively classified as Kaiparowits gravelly fine sandy loam (very-fine, smectitic, frigid Oxyaquic Paleustalfs). The upland areas on 2-15% slopes above the drainages were tentatively classified as the Kanabownits-Kippers-Kaiparowits complex (loamy-skeletal, mixed, semiactive, nonacid,frigid Oxyaquic Ustorthents). Valley soils along narrow drainages were tentatively classified as the Plite-Canburn Families complex (coarse-loamy, mixed, superactive, frigid Cumulic Haplustolls).

Forests in the Galahad study site were dominated by ponderosa pine with Gambel oak and New Mexican locust (scientific names and species codes are given in Table 1). At Little Park, tree species included ponderosa pine, aspen, white fir, and subalpine fir. Two spruce species were encountered: Engelmann and blue spruce. Both species were combined as "spruce" in this study because of difficulties in distinguishing young trees (Moore and Huffman in review) and our observation of trees at the study site that had characteristics intermediate between the two species.

Fire scar sampling and analysis

Sampling of fire-scarred trees was carried out in 2000 at the Galahad and Little Park sites and 2001 at Little Park. The intent of sampling was to obtain as complete as possible an inventory of fire dates and point locations (i.e., scarred tree locations) where these fires occurred (Swetnam and Baisan 1996). We reconstructed comprehensive fire histories over large study areas. The Galahad study site was completely surveyed along parallel transects to observe all fire-scarred trees. At the much larger Little Park site, fire-scarred trees were encountered predominantly on ridgetops and SW-S-SE aspects. We sampled nearly 100% of these areas, as well as representative transects across northern aspects on each ridge. Trees with the longest and most complete fire records were selected. Partial crosssections were cut from scarred 'catfaces' on trees, logs, and stumps of conifers. Samples were mapped when collected and were well-distributed throughout the study areas (Figure 1).

Laboratory Methods

Samples were mounted on plywood backing and surfaced with sandpaper to 400 grit. Tree rings were crossdated (Stokes and Smiley 1968) with marker years listed in Fulé et al. (2002). Dates were independently confirmed by another dendrochronologist or ring widths were measured and dating was checked with the Cofecha software (Holmes 1983). Following the procedure of Baisan and Swetnam (1990), the season of fire was estimated based on the relative position of fire injury within the annual ring according to the following categories: EE (early earlywood), ME (middle earlywood), LE (late earlywood), L (latewood), and D (dormant). Dormant season scars were assigned to the year of the following earlywood (i.e., spring fires), as conventionally done in the Southwest (Baisan and Swetnam 1990). The final fire on most of the samples occurred after the 1879 latewood was complete but before the 1880 earlywood formed. This fire would conventionally be assigned to the calendar year 1880, but a few samples were scarred in the 1879 late earlywood or latewood. Therefore we interpreted all these scars as occurring in 1879 around the end of the growing season.

Data were analyzed with FHX2 software (Grissino-Mayer 2001). Analysis at each site began with the first year with an adequate sample depth, defined as the first fire year recorded by 10% or more of the total sample size of recording trees at each site (Grissino-Mayer et al. 1995). "Recording" trees are those with open fire scars or other injuries (e.g., lightning scars), leaving them susceptible to repeated scarring by fire. At Galahad the first year of analysis was 1744; at Little Park the first year was 1700. The relationship between climatic fluctuations and fire occurrence was compared with superposed epoch analysis (SEA), using software developed by Grissino-Mayer (1995). A locally developed tree-ring chronology served as a proxy for climate (19 ponderosa pine trees from Powell Plateau, Rainbow Plateau, and Fire Point, master series 1559-1997, series intercorrelation = 0.70, average mean sensitivity = 0.34). The chronology was significantly correlated with reconstructed Palmer Drought Stress Index ($r = 0.67$) for grid point 31 in northern Arizona, A.D. 1694-1978 (Cook et al. 1996). The SEA superimposes fire years and summarizes the climate variable for fire years as well as preceding and succeeding years. Confidence intervals were developed using bootstrapping methods with 1000 simulations based on random windows with the actual fire events.

Forest Structure Sampling and Analysis

At first we attempted to identify fire-initiated stands across the study area (Johnson and Gutsell 1994), using current and the oldest available aerial photographs (1956), vegetation maps (Warren et al. 1982), data from adjacent stands (White and Vankat 1993), and field surveys. All these sources of information showed that forest structure was highly diverse, suggesting a mixture of surface fire together with fire-initiated groups or patches that may have contained many fire survivors. Rather than attempting to force-fit the diverse landscape into a dichotomous scheme of fire-initiated versus non-fire-initiated stands, we sampled on a systematic grid to capture forest age structure and species composition proportional to occurrence.

Sampling plot centers were located from a systematic grid placed over each sampling site. When a gridpoint fell in an unsuitable location (e.g., archeological site), the points 50 m N, E, W, and S were checked for suitability. If none were acceptable, the gridpoint was discarded. Sixty plots at Little Park were located on a grid 600 m E-W and 1,200 m N-S in order to measure with greater sampling density along the prevailing elevational gradient, roughly E-W. The grid spacing was designed to measure fire and forest structure at a large scale, consistent with the expectation of large fires under a mixed-severity fire regime.

Permanent plots based on the National Park Service's Fire Monitoring protocol (NPS 1992, Reeberg 1995) were used to measure current conditions of vegetation and fuels, and to collect dendroecological data for reconstruction of past forest structure. Sampling plots were 0.1ha (20 x 50 m) in size, oriented with the 50 m sides uphill-downhill to maximize sampling of variability along the elevational gradient and to permit correction of the plot area for slope. Plot corners and centers were marked with 30-cm iron stakes sunk flush to the forest floor. The distance and bearing from a tagged reference tree to the center was recorded. Photopoints were established at the corners and quarter-corners of each plot.

Trees larger than 15 cm diameter at breast height (dbh) were measured on the entire plot (1000 m^2) and trees between 2.5-15 cm dbh were measured on one quarter-plot (250 m^2); all trees were tagged. Tree attributes measured were: species, dbh, height, crown base height, dwarf mistletoe rating (0-6, Hawksworth and Geils [1990]), and tree condition (1. live; 2. declining; 3. recent snag; 4. loose bark snag; 5. clean snag; 6. snag broken above breast height; 7. snag broken below breast height; 8. downed dead tree; and 9. cut stump; see Fulé et al. 1997 and 2002 for discussion of snag condition classes). Previous research in northern Arizona showed that ponderosa pines with dbh > 37.5 cm or ponderosa of any size with yellowed bark could be conservatively identified in the field as being of pre-1880 origin (White 1985, Mast et al. 1999). We used the same diameter breakpoint for other conifers (Fulé et al. 2002) and cored all aspen trees > 20 cm. "Conservative" estimation meant that these criteria included all pre-1880 trees as well as numerous post-1880 trees. Tree status was later corrected in the laboratory using age data. All living trees meeting the field criteria above were considered potentially pre-1880 trees and were cored. Ten percent of all post-1880 live trees were also cored. Coring height was 40 cm above ground level, chosen to meet two objectives: first, to measure tree age, and second, to measure growth between the fire regime disruption date and the present (needed for the forest reconstruction). The two objectives conflict because the best coring height for age is ground level, but the butt swell and irregular growth around the root collar make this an inappropriate height for growth measurement. The 40-cm height is the lowest position on the bole for consistency of tree form, permitting a good measurement of growth. Seedling trees, those below 2.5 cm dbh, were tallied by species, condition, and height class in a 50 m^2 subplot.

Understory vegetation was sampled along point line-intercept transects (the two 50-m sidelines of each plot). Vegetation and substrate type (litter, wood, soil, etc.) were recorded every 30 cm along each line for a total of 332 points per plot. Overlaid on each line was a 50 X 10-m belt transect, within which all understory species present were recorded. Canopy cover measured by vertical projection (Ganey and Block 1994) was recorded at 30 cm intervals along the same 50-m sidelines used for the point line-intercept transects. Forest floor and woody debris were measured along four 15.24 m planar intersect transects (Brown 1974) placed in random directions every 10 m along the plot centerline. Woody debris was recorded by size/timelag classes (Anderson 1982): 1, 10, 100, and 1000-hour (sound and rotten) fuels. Litter (L layer, undecomposed material) and duff (F and H layers, decomposing material or humus) were measured every 1.52 m along each transect.

Laboratory Methods

Plot areas were corrected for slope. Tree increment cores were surfaced and visually crossdated (Stokes and Smiley 1968) with tree-ring chronologies we developed. Rings were counted on cores that could not be crossdated, especially younger trees. Additional years to the center were estimated with a pith locator (concentric circles matched to the curvature and density of the inner rings) for cores that missed the pith (Applequist 1958). Past forest structure was reconstructed at the time of disruption of the frequent fire regime, 1887 at the South Rim sites and 1879 at the North Rim sites, following dendroecological methods described in detail by Fulé et al. (1997). Briefly, size at the time of fire exclusion was reconstructed for all living trees by subtracting the radial growth measured on increment cores since fire exclusion. For dead trees, the date of death was estimated based on tree condition class using diameter-dependent snag decomposition rates (Thomas et al. 1979) or historical harvesting records for stumps. To estimate growth between the fire exclusion date and death date, we developed local species-specific relationships between tree diameter and basal area increment (r^2 = .45 to .90). An analogous process of growth estimation was used to estimate the past diameter of the small proportion of living pre-1880 trees for which an intact increment core could not be extracted due to rot. Fuel loadings were calculated from the planar transect data (Brown 1974, Sackett 1980).

Age and species composition data were used to classify the 60 plots into fire-initiated and non-fire-initiated groups, following methods similar to Murray et al. (1998). All age cohorts within each plot were taken into consideration but the greatest weight was given to the oldest trees. When the oldest tree or trees were the fire-resistant species PIPO and PSME, the plot was classified as non-fire-initiated. When the oldest trees were the fire-susceptible species POTR, PIEN, or ABLA, the plot was classified as fire-initiated. ABCO was considered intermediate in fire resistance and old-ABCO plots were classified as non-fire-initiated when accompanied by uneven-aged PIPO or PSME, and as fire-initiated when approximately accompanied by approximately equal-aged POTR. In general, the non-fire-initiated classification corresponded to "uneven-aged" structure, but the fire-initiated classification was not interpreted as "even-aged"—plots with fire-susceptible old trees often had one or more cohorts of younger trees, a category called "mixed severity" by Murray et al. (1998).

Vegetation at the Little Park study area was classified from a Landsat Enhanced Thematic Mapper (ETM) image, acquired 6 June 2000. The classification scheme was designed under the National Vegetation Classification Standards (NVCS) framework, developed through the USGS-NPS Vegetation Mapping Program in association with USGS/BRD, EPA, National Park Service, The Nature Conservancy, Ecological Society of America, and others (USGS 2000). The goal of the NVCS is to provide a consistent national vegetation classification system. The plot data were used as training sites for image classification. The species label for each training site was based on 'importance value' (Taylor 2000) calculated as the sum of the relative frequency (percent stems) and relative abundance (percent basal area) for each species. This approach was similar to the classification method used by White and Vankat (1993) on an adjacent study area, but differed in that we chose to include aspen as a distinct forest type, because of its prevalence in the study area and importance in assessing post-fire succession, and we did not differentiate the two spruce species, for reasons noted below. Four forest types were used in this study: aspen, mixed conifer, ponderosa pine, and spruce-fir. A non-forested grass type was also identified. Vegetation and permanent plot locations are shown in Figure 2.

Understory plant community diversity was assessed with Simpson's and Shannon's indices (Magurran 1988). Simpson's original index (sometimes noted as SI), is inversely related to diversity, so it is more often expressed as Simpson's D (1-SI) or Simpson's D' (1/SI) to generate an index which is directly proportional to community diversity. Simpson's Index is heavily weighted towards the most common species, though, so Shannon's H' was used to distinguish subtle community changes based upon less common species. Plant species were classified into various functional groups which are ecologically significant components of the total plant community. First, legumes (family Fabaceae) were identified due to their importance in many plant communities for their nitrogen-fixing properties. Second, plants were classified based on growth habit (i.e. herb, graminoid, shrub, tree). Third, plant species were classified based upon their life history: annuals, biennials, or perennials.

Statistical analysis of forest structural data included t-tests, multivariate analysis of variance and repeated-measures analysis of variance. Data were transformed where necessary to meet the assumptions of parametric tests. Alpha level was .05.

Results

Fire Scar Data

Galahad Point: Fire history was reconstructed for the period 1744-2000 from 31 fire-scarred trees (Figure 3). Fires recurred frequently until 1879; only one scar occurred after 1879 (1954). The following statistics apply to the 1744-1879 period. The mean fire interval (MFI) calculated from all fire scars was 4.0 years (Table 2). The Weibull median probability interval (WMPI) for the same data was 3.6 years. For fires scarring 25% of more of the samples, average fire intervals increased by approximately 70% to MFI = 6.8 years, WMPI = 6.1 years. The minimum fire-free interval was 1 year, maximum 12 years, rising to 2 and 18 years, respectively, under the 25%-scarred filter. The mean point (or per-sample) fire interval was 11.3 years (minimum 6.5 years, maximum 20.3 years).

The season of fire occurrence was determined on approximately 75% of the fire scars; over 85% of the scars occurred in the middle earlywood to latewood (Table 3). Fire years in which 25% or more of the samples were scarred matched 11 of the 17 major fire years in which fires burned at 3 or all 4 of the sites previously studied on the North Rim transect (Table 4, see Figure 1 for study sites). Superposed epoch analysis showed that fire years were significantly dry (Figure 4). The year preceding fire tended to be relatively wet but the proxy climate variable did not exceed the 90% confidence interval (Figure 4).

Contemporary fire records for the Galahad Point area (buffered by approximately 1 km) contained 40 lightning-caused fires between 1936 and 1990. A 3.3-ha wildfire, the Bedivere fire, burned in 1954 and was detected in the fire history. However, the largest fire that burned within the sampling area was the 12-ha Bedivere fire (1990), was not detected in the fire history. All other wildfires were 3.7 ha or less, most commonly recorded as 0.04 ha, the minimum reporting size. All wildfires were suppressed until 1986, when a lightning ignition was allowed to burn, reaching 1.2 ha in size. Taking recent years as the most reliable historic period (1967-1996), 25 lightning fires or 0.8 lightning fires/year occurred.

Big Spring/Little Park: Fire history was reconstructed for the period 1700-2000 from 132 fire-scarred trees (Figure 5). Scarred trees were predominantly encountered on S aspects (53%) followed by W (23%) and E (19%) aspects. Only 5% of samples were collected from N-facing slopes. A number of additional scarred samples were collected but could not be confidently crossdated, due to highly complacent ring series, so they were excluded from the data set.

In a similar pattern to the other North Rim sites, fires nearly ceased after 1879, with only two subsequent fire dates, 1893 (one scarred sample) and 1921 (two scarred samples). The following statistics apply to the 1700-1879 period. The all-scars MFI and WMPI showed the highest fire frequency of any North Rim site, 2.6 and 2.3 years, respectively (Table 2). In contrast, applying the 25%-scarred filter led to the longest fire-free intervals, MFI = 31.0 years and WMPI = 23.4 years, an increase of approximately 900-1100 % over the all-scar intervals. The change in minimum and maximum fire-free years among the different fire categories was also striking: for all scars, the minimum fire-free interval was 1 year and the maximum was 12 years, identical to the Galahad site. But for the 25%-scarred category, the minimum interval rose to 11 years and the maximum to 94 years

(1785 to 1879). The mean point fire interval was 31.9 years (minimum 8.9 years, maximum 99.0 years), nearly three times as long as the Galahad mean point fire interval.

The data set was subdivided by the forest type, aspect, and elevation of the fire-scarred samples, using the 10%-scarred filter as a standard of comparison (Table 2). For the entire data set, with the 10%-scarred filter, MFI = 8.0 years and WMPI = 6.8 years. Differences were relatively minor among the subdivided groups. By forest type, the MFI values for the 10%-scarred groups ranged from 5.7 to 8.8 years, with samples collected in the aspen type having the shortest average fire-free intervals and samples collected in the spruce-fir type having the longest (WMPI values and min/max intervals for these groups are shown in Table 2). By aspect, MFI values for the 10%-scarred groups ranged from 5.5 to 13.1 years, with samples collected on south aspects having the shortest average fire-free intervals and samples from north aspects having the longest. By elevation, MFI values for the 10%-scarred groups ranged from 6.3 to 15.9 years, with samples collected below 2,650 m having the shortest average fire-free intervals and samples collected above 2,750 m having the longest.

The season of fire occurrence was determined on approximately two-thirds of the fire scars. As at the Galahad site, above, the majority of the scars occurred in the middle earlywood to latewood (Table 3), but the proportion of "summer" scars was only 64.4%, less than the 82-88% range of Galahad and the other 4 North Rim sites.

There were only five fire years in which 25% or more of the samples were scarred (Table 4). All five years—1735, 1748, 1773, 1785, and 1879—matched major fire years in which fires burned at 3 or all 4 of the sites previously studied on the North Rim transect as well as at the Galahad site (Table 4). Fire years were significantly dry, as shown by superposed epoch analysis, but pre-fire years were wet. Four of the five pre-fire years were significantly wet at the 90% confidence interval and two exceeded the 95% confidence interval (Figure 4).

Contemporary fire records for the Little Park area (buffered by approximately 1 km) contained 71 lightning-caused fires between 1933 and 1996. None were detected in the fire history. The largest historic fire was the 5.1-ha Upper Big Spring Canyon fire (1940). All other wildfires were 1.6 ha or less, most commonly recorded as 0.04 ha, the minimum reporting size. All wildfires were suppressed until 1981. A total of 4 lightning ignitions were allowed to burn in 1981, 86, and 88, but none exceeded 0.04 ha in size. For the reliable historic period (1967-1996), 38 lightning fires or 1.3 lightning fires/year occurred.

Overstory Forest Structure

Spruce-fir was the predominant forest type classified from Landsat imagery, representing 2,860 ha or 67% of the forested area (Table 5). The remaining types, aspen, mixed conifer, and ponderosa pine, represented 17%, 11%, and 3%, respectively. Average elevation increased from ponderosa pine through mixed conifer and aspen to spruce-fir, but the differences were slight, ranging only over 68 m (Table 5). Data from the systematic plot grid followed similar trends but tended to underrepresent the dominant types and overrepresent the rare types: spruce-fir, aspen, mixed conifer, and ponderosa pine made up 47%, 22%, 25%, and 7% of the plot data, respectively. Average slopes at sample plots ranged from 14.5-19.0% (Table 5). Aspects were not evenly encountered at the 60 plot locations; the overall distribution was N = 9 plots (15%), E = 12 (20%), S = 21 (35%), and W = 18 (30%). This distribution may be related to the skewed relationship between the proportions of forest types calculated from the imagery versus from the plot data, since the ponderosa pine forest type was encountered only on S and W aspects (Table 5).

Forest structure was highly varied within forest types (Table 6), with all six dominant tree species occurring in all four forest types. Only RONE was limited to a single forest type (mixed conifer). In contrast, ABLA averaged over 100 trees/ha in all four types, although the trees were relatively small, as indicated by the low basal area values (< 3.2 m^2/ha except in the spruce-fir type). POTR trees were actually outnumbered by PIEN trees in the aspen type, although POTR dominated in basal area. Total tree densities ranged from 946 trees/ha in the spruce-fir type to 1382 trees/ha in the aspen type and basal areas ranged from approximately 28 to 39 m^2/ha. Mean canopy cover (Table 7) ranged from 52-61%, with the highest average (61%) and maximum (85%) occurring in the aspen type. The spruce-fir type had the greatest range in cover values, 63%. Tree regeneration was dense, from > 5,000 to > 12,000 seedlings or spouts per ha (Table 8). POTR was the most prolific species, averaging > 2,000 new trees/ha in each forest type. At the other extreme, PIPO had no recorded regeneration in the aspen and spruce-fir types, which make up 78% of the landscape. No PIEN seedlings were found in the ponderosa pine vegetation type, but it only comprised 3% of the study area.

Age distributions (center date at the 40-cm sampling height) by forest type are shown in Figure 6a-d. All forest types had numerous old trees of several species but were numerically dominated by post-1880 regeneration. The ponderosa pine age structure was artificially truncated because through random chance no pole-sized trees were sampled (10% subsampling) on the four ponderosa pine forest type plots. The oldest trees by species were: ABCO 1735, ABLA 1811 (ABLA was never the oldest tree on any plot), PIEN 1788, PIPO 1618, POTR 1770, and PSME 1693.

Fire-initiated groups were identified on 58% of the area (35/60 plots) (Table 9). North and east aspects had higher proportions of fire-initiated groups (71% of the N- and E-facing plots), while south and west aspects were nearly equal in fire-initiated versus non-fire-initiated groups (51% and 49%, respectively, of the S- and W-facing plots). The aspen and spruce-fir types had predominantly fire-initiated groups (73% of the plots in these two forest types), the ponderosa pine type was evenly split (50% each), and the mixed conifer type had predominantly non-fire-initiated groups (80%). The two oldest plots classified as fire-initiated were PIEN-dominated, with oldest trees dating to 1788 and 1791. The oldest POTR group dated to 1792 and ten additional old-POTR groups predated 1860. Given the imprecision of age sampling at 40 cm above ground level and estimating rings to pith on many increment cores, we did not attempt to finely delineate age cohorts within plots. However, clearly distinct cohorts of POTR were evident on 9 plots. Five of these plots were classified as non-fire-initiated; the remaining four were classified as fire-initiated, with POTR as the oldest tree species. The oldest trees on fire-initiated plots were significantly older than on non-fire-initiated plots (mean = 1754 vs. 1851, t-test $P < .001$). Fire-initiated plots also had steeper slopes (mean = 18.8% vs. 12.7%) but the difference was not statistically significant. Mean elevations of fire-initiated and non-fire-initiated plots were nearly equal (2,690 m vs. 2,682 m).

Reconstruction of forest structure in 1880 (Table 10) showed that past forests were significantly less dense and had significantly lower basal area (paired t-test). Total tree densities ranged from 150 to 337 trees/ha, only 16-24% as dense as the contemporary forest. Basal areas in 1880 ranged from approximately 10 to 18 m^2/ha, about 36-46% as dense as the contemporary forest. Basal area values for PIPO, POTR, and PSME were the least changed over the 1880-2000 period, but ABCO, ABLA, and PIEN were sparse in the 1880 forest, in contrast to their current dominance.

Forest Floor and Understory Plant Community

Substrates: Litter frequency decreased steadily from about 72% to 59% following the elevational gradient from the ponderosa pine type through mixed conifer, aspen and spruce-fir forests, (Table 11).

Plant frequency increased over this same gradient, though the highest value (about 27%) was actually found in the aspen forest. Bare soil comprised less than 3% of the ground surface in all forest types. The depth of duff material in the forest floor was relatively consistent (2.3 to 2.9 cm) but litter depth was significantly higher in the ponderosa pine forest type (1.6 cm) as in the other three types (0.8-0.9 cm) (Table 12). Woody debris biomass was very high, ranging from 95 to 142 Mg/ha. Coarse woody debris (1000 H timelag or > 7.62 cm diameter) was relatively balanced between sound and rotten material, except in the ponderosa pine forest type. Variability in woody debris biomass was also high, with standard errors ranging from 17-70% of the means.

Species richness and diversity: Sample sizes varied significantly between forest cover types and species richness tends to covary with sample sizes, so mean species richness per plot was calculated for each forest type (Table 13). Species richness increased along the elevational gradient, peaking in the aspen cover type with an average 30.6 species per plot. The range was small, however, with a minimum average richness of 27.5 species in the ponderosa pine type. Table 13 shows the Simpson's D, Simpson's D', and Shannon's H indices calculated for the four forest types. There was a trend of increasing diversity with increasing elevation, but the Simpson's indices showed the aspen community as slightly more diverse than the spruce-fir community while Shannon's index shows the spruce-fir community as the most diverse. The ponderosa pine was least diverse but also had the fewest plots (Table 5).

Non-native species comprised a very small percentage of the sampled vegetation communities (Table 14), with absolute frequencies of non-natives ranging from about 0.1% in the mixed conifer type to about 0.7% in the ponderosa pine type, and relative frequencies ranging from about 0.4-3.9% for the same communities. The non-native component of the point line-intercept data (Table 15) was made up entirely of two species, *Poa pratensis* (Kentucky bluegrass) and *Taraxacum officinale* (common dandelion), neither of which is listed as noxious by state or federal agencies. To assess the aerial extent of these non-native populations, frequencies were also calculated from the belt transect data based upon the number of plots where non-native plants were found, out of the total number of plots for each cover type. Table 16 shows the results of this analysis, with percentages ranging from about 6.67% for *P. pratensis* in the mixed conifer plots to 25% for the same species in the ponderosa pine plots. Combining the two sources of information above, we can conclude that these two non-native plant species are not uncommon across the landscape, but in low abundance.

Functional groups: Seven species of legumes were found on the study plots (Table 17), with only *Lotus utahensis* (Utah birdsfoot trefoil) being found in all four forest types. Overall, legumes made up a relatively small part of the vegetation communities, ranging from about 2.3-5.5% of total plants encountered (Table 18). Graminoid and shrub species are listed for each cover type in Table 19. Frequency of graminoids at the study site increased from about 3.9% in the ponderosa pine community to about 9.4% in the spruce-fir community (Table 20). Shrub frequency was generally low across the study site (Table 20), and did not seem to follow a consistent trend, though the frequency peaked in the aspen community at about 4.5%. No annual plant species were encountered on the study plots (Table 21). The vast majority of species found in all four cover types were perennials, ranging from about 98.6% in the spruce-fir forest to 100% in the ponderosa pine forest. The remaining species encountered were either biennials or plants which can live as biennials or perennials.

Discussion

Fire regime

Fire scar data: An overall interpretation of a fire regime with numerous small fires but few larger fires is supported by the fire-scar data at the Little Park site. Seventy fire years were identified between 1700 and 1879 at the Little Park study area, resulting in the lowest mean fire interval (2.6 years) of any Grand Canyon study site. This extraordinarily high fire frequency supports the suggestion that total fire occurrence is likely to increase with increasing size of the study area, simply because more and more small fires are encountered (Minnich et al. 2000, Baker and Ehle 2001). But the majority of the fires scarred a low proportion of the sample trees. With a 10%-scarred filter, the MFI rose to 8.0 years, slightly higher than the 4.0 years at Galahad and 4.5-7.1 years at the 4 other lower-elevation study sites described by Fulé et al (in review). However, it was the 25%-scarred filter that distinguished Little Park from the lower sites: 31.0 years versus 6.4-9.0 years. Mean point fire intervals at Galahad and Little Park, 11.3 and 31.9 years, respectively, were similar to the composite fire frequencies calculated under the 25%-scarred filter.

Lightning fire ignition data from the twentieth century suggests that ample lightning ignition opportunities would have existed to support the fire frequency, even apart from any additional human-caused ignitions (Allen 2002). Compared to other studies in the Southwest, MFI values using the 10%-scarred filter ranged from approximately 10-26 years in 21 fire history studies in mixed conifer forests listed by Swetnam and Baisan (1996a). With the 25%-scarred filter, the MFI values rose only slightly, to approximately 15-26 years (Swetnam and Baisan 1996a).

The fire-scar fire history at Little Park is qualitatively different from lower-elevation fire histories because samples were not evenly distributed by aspect, so surface fire occurrence cannot be assumed to have occurred uniformly over the landscape. The presence of fire-initiated tree groups indicates that the absence of fire-scarring is evidence of severe fire, rather than of the lack of fire. Even where trees were fire-scarred, the relatively long mean point fire intervals compared to Galahad suggest that individual locations on the ground were burned much less frequently at Little Park, similar to the pattern of increasing mean point fire intervals with elevation observed by Taylor (2000) in California.

Forest structure data: Approximately 58% of the plot-scale tree groups appeared to have been fire-initiated. Aspen groups would provide the clearest evidence of past severe fire if the oldest aspens reliably represent a post-fire cohort. This interpretation may be confounded by the observation of multiple aspen cohorts on four of the old-aspen groups classified as fire-initiated (11%). If uneven-aged aspen regeneration were not uncommon, as suggested by Ripple and Larson (2000), old aspens would indicate a minimum fire-free period but not necessarily the date of a stand-replacing fire. One or more aspen cohorts were also encountered on five plots (20%) with old fire-resistant trees, classified as non-fire-initiated, a finding that suggests that fire severity variation even at the sub-plot scale (< 0.1 ha) may have influenced the relationship between fire survivors (PIPO or PSME) versus fire-initiated POTR groups.

Spruce age data are also subject to alternative interpretations depending on possible differences in successional pathways. The oldest spruce groups, dating to *circa* 1790, might represent immediate post-fire regeneration following fires in 1773, 1782 or 1785. The possibility that spruce could colonize post-fire openings immediately is supported by the fact that the old-spruce age distribution is narrow (1788, 1791, 1795, 1796). However, if the spruce succeeded an aspen stand, the date of an original stand-replacing fire might precede 1790 by 100 or more years.

The most reliable interpretation of the age data from fire-susceptible trees is in terms of minimum fire-free periods. By this criterion, the two old-age PIEN groups and the oldest POTR group (1770) show that approximately 5% of the Little Park site has not had a fire severe enough to kill fire-susceptible trees for 210-230 years. The post-1879 fire-free period is an artifact of land management practices introduced by European settlers (Altshul and Fairley 1989, Wolf and Mast 1998, Fulé et al. 2002), so 1880 would serve as a better endpoint for calculating the fire-free period. In 1880, 5% of the landscape had been fire-free for at least 90-110 years. Analogous calculations suggest that 10% of the landscape was fire-free for at least 60-80 years in 1880, 12% was fire-free for at least 40-60 years, 8% was fire-free for at least 20-40 years, and 18% was fire-free for less than 20 years. Finally, an additional 5% of the landscape was unforested in 1880 and regenerated with fire-susceptible species between 1890-1930.

If the sampling proportions and tree ages of fire-initiated plots were translated directly into proportional area, the data listed above could be used to create a time-since-fire map and calculate fire rotation (e.g., Heinselmann 1973, Agee and Krusemark 2001). Several studies have used fire scars in conjunction with stand mapping, species composition, and tree age measurements for developing comprehensive fire regime studies (e.g,, Niklasson and Granstrom 2000, Taylor 2000). However, the assumptions underlying fire rotation analysis are substantial: "distinct boundaries between different aged burns" (Johnson and Gutsell 1994:243) and clear dating of stand origins, or, in the case of fire sizes calculated from fire-scar data, assigning fire areas around point locations of scarred trees (Agee 1993). These assumptions were not well-supported in this study, where distinct stands could not be clearly delineated, fire-initiated groups were intermixed with non-fire-initiated groups, and fire-scarred trees were not encountered evenly across the landscape. Therefore we summarize fire frequency in approximate terms.

The weighted average of the fire-free periods, about 22 years, is reasonably close to the MFI of 31 years for the fire-scar data with the 25%-scarred filter and the mean point fire interval of 31.9 years. Taken together, the fire-scar and forest structural approaches are consistent with a mixed-severity fire regime at the Little Park site. On approximately 40% of the landscape, notably on S and W aspects, surface fires recurred. Even where these fires burned hot enough to initiate POTR groups at the < 0.1 ha scale, old PIPO, PSME, and ABCO trees survived. On approximately 60% of the landscape, severe fires occurred at intervals that averaged 20-30 years in the century preceding 1880. During that period, at least 5% of the landscape had not burned at all and at least 23% had burned within the past 20 years.

Fire-climate relationship and fire extent

Fires at Little Park and Galahad tended to occur in drier years, as seen throughout the Southwest (Swetnam and Baisan 1996a, 2001, Allen 2002) and at lower elevation North Rim sites (Fulé et al. 2000). Swetnam and Baisan (1996a) suggested that a lagging effect of fine fuel production in wet years might be followed by fires; we saw a trend toward wet conditions in the pre-fire year at Galahad (Figure 4). But the observation that several years before fire were significantly wet at Little Park was surprising. Similar trends were seen with the 25%-scarred Little Park fire chronology (data not shown), but there were only five such fire dates between 1700-1879, limiting the reliability of the fire-climate relationship. It will be helpful to compare further studies from high-elevation southwestern forests to assess whether the pre-fire moist conditions are a regional or just a local phenomenon.

Fires were clearly connected from the lower landscapes to the higher-elevation Little Park site. All five major fire years at Little Park (25%-scarred filter) coincided with major regional fire years (Table 4). The Galahad site was actually less connected to Little Park, located upslope and downwind, than were the four western sites in 1773, a major regional fire year in which fire was not detected at Galahad. The dry PDSI values reconstructed in regional fire years (Table 4) support Swetnam and Baisan's (1996a) contention that climate must have been the major factor underlying regionally synchronous burning. The reconstructed PDSI values do not have the accuracy of measured weather data (see Cook et al. [1996] for validation data), but comparisons may be useful in relative terms. The average PDSI value for all the fire years listed in Table 4 was -1.89, compared to an average of -0.37 for the entire 1700-1879 period. Average PDSI in the five major regional fires at both low and high elevation (shaded rows in Table 4) was -3.28, corresponding to severe drought conditions (Pyne et al. 1996).

Considering only the actual sampled areas, fires in major regional fires such as 1785 (Figure 1) appear to have covered at least 5,000 ha. Continuous fuels existed *between* all the sites and Little Park shared major fire years with the sites located directly west, so fire in years such as 1785 would have been likely to have burned at least 24,000 ha, the between-site area. But fuels extend to the north and east well beyond the extent of all the sampled areas, suggesting that fires may easily have exceeded 50,000 ha in size.

Changes in forest structure

Dendroecological reconstruction showed that forest tree structure had changed substantially since 1880. The accuracy of the forest reconstruction procedure used here was discussed by Fulé et al. (1997, 2002). In general, reconstructions are reliable if the site has not been disturbed, if the period of reconstruction is not excessive relative to the lifespan of the trees, and if dead tree evidence (stemwood, bark, and/or rootball) is likely to persist (e.g., Habeck 1990, Foster et al. 1996). In the present case, the Little Park study site has never been harvested and was not burned over the reconstruction period. Individuals from all six dominant tree species had lifespans 200-300% of the reconstruction period (Figure 6). The persistence of dead tree evidence is well-established for ponderosa pine in northern Arizona (Mast et al. 1999, Huffman et al. 2001). The persistence of other species is less documented, especially for small trees. However, the striking changes in forest structure would still be evident even if dense small tree groups had been present in 1880 and subsequently disappeared. As an example of the upper boundary of error, assume that all the trees < 15 cm dbh present today were to be added to the 1880 forest. The tree density would rise by an average 526 trees/ha but the basal area would increase by only 1.3 m^2/ha, accounting for only about 6.5% of the rise in basal area from 1880 to 2000.

The oldest forest survey of the Kaibab Plateau, by Lang and Stewart (1910), was in remarkably close agreement with the forest reconstruction results (Table 22). Lang and Stewart's (1910) "spruce-balsam" stand average tree densities were within 5-9% of reconstructed tree density values. The "spruce-balsam" (spruce-fir) was described as occupying "only northern aspects up to 8,800 feet [2,683 m] elevation where it extends over the ridges. Occasionally large veteran yellow pine occur among the balsam and spruce, a strong evidence that the primeval forest was pine. It is thought that the mixed type has succeeded the original yellow pine on account of the cumulative effects of severe fires, and is still advancing upon it" (Lang and Stewart 1910:9). With respect to fires, they commented on "vast denuded areas, charred stubs and fallen trunks ... The old fires extended over large areas at higher altitudes, amounting to several square miles on either side of Big Park [now called DeMotte Park, immediately N of the present Little Park study site] and to numerous smaller irregular areas over the remainder of the forest" (Lang and Stewart 1910:18-19). Aspen was described as "exceedingly active in restocking burns" and they photographed "Spruce-Balsam coming in under aspen on northern exposure of old burns" (Lang and Stewart 1910:9, 18-19). In contrast, at lower elevation, "evidence indicates light ground fires over practically the whole forest..." (Lang and Stewart 1910:19).

The oldest PIEN tree on the Little Park site had a center date of 1788 at the 40-cm coring height. Much older spruce stands were reported by Grissino-Mayer et al. (1995) in southern Arizona (300+ years) and Aplet et al. (1988) in Colorado, where four out of five study sites had trees in the 251-275 year category and the oldest trees were in the 601-625 year category. The oldest ABLA at Little Park had a center date of only 1811, well below the 200-500 year age range of ABLA in Colorado (Aplet et al. 1988). The oldest POTR tree was about 230 years (1770 center date) but the majority of POTR were also young. A notable gap in POTR regeneration occurred around the 1920-1940 age class (Figure 6). POTR cohorts from this period were probably killed by heavy deer browsing during the 1920's and 30's (Rasmussen 1941, Merkle 1954, 1962, Mitchell and Freeman 1993, Fulé et al. 2002). Apart from this gap, POTR regeneration was consistent in all forest types since the early 1800's and heavy POTR regeneration also occurred through the mid-1900's (Figure 6), with several plots supporting multiple age cohorts of POTR. These findings tend to support the hypothesis of Ripple and Larsen (2000) that aspen is capable of uneven-aged regeneration in the absence of heavy herbivore pressure, notably elk. Unlike most regions with aspen in the West, the Kaibab Plateau has been free of elk until recently and the current population is small (S. Germaine, Arizona Game and Fish Department, personal communication 2001).

The intersecting evidence of intermittent large-scale fires recorded on fire scars, relatively young age structure of all fire-susceptible species, low forest density, and historical accounts reinforce a picture of a very open forest of many young trees in 1880, with more than half of the Little Park site in early post-fire successional stages. These findings stand in contrast to long-lived PIEN and ABLA trees in other sites and with pollen and plant macrofossils in local sediment cores (Bear Lake, approximately 3 km N of Little Park) that show *Picea* spp. and *Abiea lasiocarpa* presence at the highest elevations of the Kaibab Plateau for nearly 13,000 years, predating the arrival of PIPO and PSME by about 5,000 years (Weng and Jackson 1999).

We hypothesize that the mixed-severity fires may have exhibited an unstable or non-equilibrium pattern dominated by drought influences. Meko et al. (1995) identified the periods 1879-1883 and 1773-1782 as the driest 5- and 10-year periods in Arizona since 1600. Salzer's (2000) bristlecone pine chronology from northern Arizona did not identify exactly the same periods as Meko et al. (1995), but recorded several warm/dry episodes in the 1700's and none in the 1800's. At Little Park, relatively frequent large fires in the 1700's (4 fire dates: 1735, 1748, 1773, and 1785) contrast with a single large fire in the 1800's (1879, although fires in 1806, 1847, and 1873 were also widespread, scarring ~ 23% of the samples, nearly meeting the 25%-scarring criterion). One interpretation of the fire and climate data is that wetter conditions in the 1800's made possible the 94-year period without a major fire from 1785 to 1879. Fuel accumulation in this period could have supported more severe fire behavior at the landscape scale during the 1879 fire event than the shorter fire-free periods prior to 1785, leading to the open post-fire forest conditions reconstructed in 1880 and observed by Lang and Stewart (1910). However, if 1806, 1847, and 1873 are also considered as major fire years, the apparent difference between the burning patterns in the 1700's and 1800's disappears. Under this alternative interpretation, the open forest conditions and young stand age may have been typical of both centuries. Given the relatively short temporal period of these data, it is not possible to say which pattern was characteristic of the long-term fire regime at high elevations.

Currently a very dense spruce-fir forest predominates at Little Park. All forest types—even those that predominantly contain non-fire-initiated groups—exhibit high densities (782-1,382 trees/ha), basal areas (28-39 m^2/ha), canopy cover (52-61%), and woody debris (99-142 Mg/ha). These characteristics can support high-intensity, severe, stand-replacing fires. The Outlet fire, ignited in a prescribed burning operation on May 9, 2000, burned over 5,260 ha of Grand Canyon National and Kaibab National Forest lands SE of Little Park (Bertolette and Spotskey 2001). Within the park, approximately 30% of the fire area burned with low severity (tree scorching but no overstory mortality), 34% with moderate severity, 35% with high severity (complete overstory mortality), and less than 2% unburned (Bertolette and Spotskey 2001, and D. Bertolette, personal communication, 2002). The post-fire distribution of burn severities appears similar to the distribution of fire-initiated/non-fire-initiated groups at Little Park in 1879, suggesting that fires similar to the Outlet fire are not unprecedented in the high-elevation forest. However, the high severity burning in the Outlet fire was concentrated in the center of the wind-driven burn area (Bertolette and Spotskey 2001), in contrast to the highly mixed pattern of fire-initiated/non-fire-initiated groups at Little Park.

Park managers should recognize the near certainty that fire behavior similar to that observed on the Outlet fire will also occur at Little Park and other high-elevation forests under windy drought conditions. Severe burning is historically precedented in many of these forests, as shown in this study, but the vegetation types dominated by PIPO, PSME, and ABCO appear to have become unusually dense with young fir and spruce trees. Instead of mixed-severity fire behavior in a patchwork of stand densities, the modern forest forms a highly homogenous fuel complex (White and Vankat 1993) that appears likely to burn with high severity over a greater fraction of the landscape than in 1785, 1879, or other past fire years. Managers may wish to consider prescribed burning and fuel reduction treatments on S and W aspects, as well as extensive use of wildland fires managed for resource benefits, if they choose to try to restore more historically balanced fuel conditions. We were able to reconstruct pre-1880 forest structure with high precision, as supported by Lang and Stewart's (1910) measurements. However, our understanding of "historic conditions" in a broader sense is likely to always remain imprecise for two reasons. First, while the combination of fire scar analysis, remote sensing, tree age, and forest structure measurements applied here served to provide a relatively detailed and multifaceted assessment of past fire regimes, there are inherent limitations to all these methods that circumscribe inferences about the severity and exact geographical extent of past fire events. The transition zone studied here, changing from surface to stand-replacing fires, may be the most complex case for fire regime reconstruction. Second, even if we were fully able to reconstruct the details of every fire from 1700 to 1879, the pattern of severe burning did not appear to be stable over the spatial and temporal scale of the study. The end of the free-burning fire regime, 1879, coincided with the greatest extent of young, fire-initiated groups of any time in the past 300 years. Quite possibly, no "stable" distribution of severe burning events ever existed. These considerations imply that managers may be best advised to view the historic condition in high-elevation southwestern forests as a relatively general guide to reference conditions, in contrast to the more specific and temporally stable reference data available for lower-elevation ponderosa pine forests.

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Table 1. Tree species found on sampling plots at Grand Canyon study sites.

Species	Common Name	Code
Abies lasiocarpa (Hook.) Nutt.	Subalpine fir	ABLA
Abies concolor (Gordon & Glendinning) Hoopes.	White fir	ABCO
Picea engelmannii Parry ex Engelm.	Engelmann spruce	PIEN
Picea pungens Engelm.	Blue spruce	Combined with PIEN
Pinus ponderosa var. *scopulorum* P. & C. Lawson	Ponderosa pine	PIPO
Populus tremuloides Michx.	Quaking aspen	POTR
Pseudotsuga menziesii (Mirb.) Franco var. *glauca* (Beissn.) Franco	Rocky Mountain Douglas-fir	PSME
Quercus gambelii Nutt.	Gambel oak	QUGA
Robinia neomexicana Gray	New Mexican locust	RONE

Table 2. Statistical summaries of fire occurrence data from fire-scarred trees. Weibull median probability values are not reported (n/a) where the Weibull model failed to fit the fire interval data adequately (Kolmogorov-Smirnov test, alpha = .05).

Site / Analysis Period / Number of fire-scarred sample trees	No. of Intervals	Mean (MFI)	Median	Minimum	Maximum	Standard Deviation	WMPI (Weibull median probability interval)
Galahad / 1744-1879 / N = 31							
All scars	34	4.0	3	1	12	2.6	3.6
10% scarred	29	4.0	4	1	12	2.7	4.3
25% scarred	20	6.8	6	2	18	4.6	6.1
Little Park / 1700-1879 / N = 132							
All scars	69	2.6	2	1	12	1.8	n/a
10% scarred	22	8.0	6.5	2	25	6.6	6.8
25% scarred	5	31.0	13	11	94	35.7	23.4
Little Park by Forest Type (1700-1879, 10% scarred)							
Aspen (N = 38)	31	5.7	4	1	16	4.2	4.9
Mixed Conifer (N = 15)	15	8.7	5	1	28	8.3	6.6
Ponderosa Pine (N = 9)	21	7.4	6	2	25	5.2	6.6
Spruce-fir (N = 53)	20	8.8	7	2	32	7.7	7.2
Little Park by Aspect (1700-1879, 10% scarred)							
North (N = 6)	10	13.1	11	1	34	11.4	9.8
East (N = 23)	23	7.7	6	1	24	6.9	6.0
South (N = 61)	32	5.5	5.5	1	13	3.4	5.0
West (N = 27)	18	9.8	5.5	1	28	8.7	7.7
Little Park by Elevation (1700-1879, 10% scarred)							
2550-2650 m (N = 31)	28	6.3	6	2	19	4.2	5.7
2650-2750 m (N = 66)	24	6.7	5	1	21	5.1	5.7
> 2750 m (N = 18)	11	15.9	11	2	36	12.9	12.7

Table 3. Estimated season of fire occurrence based on the location of the fire injury within the annual ring. Seasonality was more difficult to determine on samples with narrow rings, deteriorated wood, or insect damage in the scarred area.

Site / Season Determined	Season Undetermined	Dormant	Early Earlywood	Middle Earlywood	Late Earlywood	Latewood	D+EE (spring)	M+LE+L (summer)
Galahad								
174 (74.7%)	59 (25.3%)	8 (4.6%)	18 (10.3%)	42 (24.1%)	68 (39.1%)	38 (21.8%)	26 (14.9%)	148 (85.1%)
Little Park								
365 (66.7%)	182 (33.3%)	77 (21.1%)	53 (14.5%)	145 (39.7%)	44 (12.1%)	46 (12.6%)	130 (35.6%)	235 (64.4%)

Table 4. Major fire years based on the percentage of fire occurrence (all fires) at four North Rim study sites between 1721 and 1879. The four sites are Powell Plateau, Fire Point, Rainbow Plateau, and Swamp Ridge (Fulé et al. 2002). Fire years recorded by 25% or more of fire-scarred samples at Galahad and Little Park are indicated by the date. Shaded rows are major regional years at low and high elevation. Reconstructed Palmer Drought Stress Index (PDSI) values are shown for northern Arizona (grid point 31, Cook et al. [1996]). Negative PDSI values indicate dry years.

Major fire years at 4 North Rim study sites	No. of sites with fire	Total no. of sites	Percent	Fire at Galahad Point (≥25% scarred)	Fire at Little Park (≥ 25% scarred)	Palmer Drought Stress Index (PDSI)
1733	3	4	75%			-0.10
1735[A]	4	4	100%		1735	-4.83
1739[B]	3	4	75%			-1.87
1744	3	4	75%	1744		0.27
1748[A,B]	4	4	100%	1748	1748	-2.38
				1754		-1.78
1755[A,B]	4	4	100%			-3.61
				1760		0.73
1773[A]	4	4	100%		1773	-3.42
				1777		-3.31
				1780		-3.40
				1782		-4.53
1785[A]	4	4	100%	1785	1785	-1.09
				1797		0.95
1800	3	4	75%	1800		-2.83
1806[A]	3	4	75%			-1.52
1810	3	4	75%			-0.61
				1818		-0.46
1822	3	4	75%	1822		-2.48
1829[A,B]	4	4	100%	1829		-0.79
1834[A]	4	4	100%	1834		-1.06
1840	3	4	75%			3.42
1841	3	4	75%	1841		-1.51
1845[A]	3	4	75%			-3.46
1851	3	4	75%	1851		-0.83
				1857		-5.44
				1863		-2.32
1873[A]	4	4	100%	1873		-2.11
1879[A]	4	4	100%	1879	1879	-4.66

[A] Also a major fire in the 25%-scarred category.
[B] Also a fire year at Grandview on the South Rim (Fulé et al. in review).

Table 5. Vegetation types, area, and average elevations for each type were calculated from Landsat imagery and digitial elevation models. Plot data includes the number of plots classified as each forest type, average slope, and the number of plots in each with north, east, south, and west aspects. No plots were located in the grass vegetation type.

Forest Type	Area (ha)	Average Elevation	Number of Sample Plots	Average Slope (%) on plots	N Aspect	E Aspect	S Aspect	W Aspect
Aspen	712	2,692	13	16.6	5	1	5	2
Mixed Conifer	453	2,666	15	14.5	2	3	5	5
Ponderosa Pine	248	2,629	4	19.0	0	0	3	1
Spruce-fir	2,860	2,697	28	16.6	2	8	8	10
Subtotal	**4,273**				**9**	**12**	**21**	**18**
Grass	108	2,690	n/a	n/a	n/a	n/a	n/a	n/a
Total	**4,381**		**60**					

Table 6. Current forest structure by vegetation type. Species codes are **GE**nus + **SP**ecies (e.g., *Abies concolor* = ABCO). Two species of spruce, *Picea engelmannii* and *Picea pungens*, are grouped together under the code PIEN.

Forest Type	ABCO	ABLA	PIEN	PIPO	POTR	PSME	RONE	Total
				Tree Density (trees/hectare)				
Aspen	196.4 (61.6)	306.1 (114.7)	419.1 (122.3)	68.8 (23.4)	352.5 (29.6)	39.3 (8.9)	0	1382.2 (170.6)
Mixed Conifer	216.1 (57.1)	159.9 (66.9)	190.8 (51.4)	46.6 (9.9)	170.1 (52.3)	76.2 (15.5)	13.7 (13.7)	873.4 (88.9)
Ponderosa Pine	199.4 (103.8)	108.7 (108.7)	2.8 (2.8)	228.2 (46.2)	222.6 (128.6)	20.6 (7.1)	0	782.3 (117.2)
Spruce-Fir	17.8 (7.2)	241.3 (43.2)	440.3 (71.7)	22.1 (7.5)	207.4 (38.7)	17.0 (5.4)	0	946.0 (99.1)
				Basal Area (m^2/hectare)				
Aspen	3.6 (1.1)	3.1 (0.9)	7.6 (1.2)	3.8 (0.8)	11.3 (1.2)	2.2 (0.5)	0	31.6 (1.7)
Mixed Conifer	14.3 (3.5)	2.1 (0.6)	5.4 (1.2)	4.6 (0.7)	3.2 (0.7)	9.2 (2.5)	0.02 (0.02)	38.8 (2.7)
Ponderosa Pine	8.2 (3.9)	1.5 (1.5)	0.06 (0.06)	18.9 (4.3)	6.0 (2.4)	0.6 (0.2)	0	35.3 (4.6)
Spruce-Fir	0.8 (0.4)	5.7 (1.0)	13.9 (1.3)	2.2 (0.8)	3.6 (0.5)	1.7 (0.5)	0	27.8 (1.8)

Table 7. Canopy cover by forest type.

Forest Type	Mean	Standard Error	Minimum	Maximum
Aspen	60.9%	3.6%	37.0%	84.6%
Mixed Conifer	55.3%	3.4%	22.9%	75.0%
Ponderosa Pine	53.0%	10.8%	31.6%	80.4%
Spruce-Fir	51.6%	2.3%	18.7%	81.2%

Table 8. Tree regeneration.

Forest Type	ABCO	ABLA	PIEN	PIPO	POTR	PSME	RONE	Total
Aspen	1,025 (458)	3,176 (1,169)	2,038 (756)	47 (25)	5,846 (1,497)	139 (100)	0	12,271
Mixed Conifer	1,933 (975)	1,166 (505)	270 (122)	0	3,828 (839)	270 (148)	550 (550)	8,017
Ponderosa Pine	1,662 (1,589)	50 (50)	0	1,062 (1,062)	2,304 (1,320)	50 (50)	0	5,128
Spruce-Fir	211 (112)	1,847 (475)	1,199 (420)	0	4,187 (819)	129 (101)	0	7,573

Table 9. Data from 60 field plots classified as fire-initiated or non-fire-initiated based on tree age and species composition.

	Fire-Initiated	Non-Fire-Initiated
All plots (N = 60)	35	25
Aspect		
N	7	2
E	8	4
S	9	9
W	11	10
Forest Type		
Aspen	10	3
Mixed Conifer	3	12
Ponderosa Pine	2	2
Spruce-Fir	20	8
Species of Oldest Tree		
ABCO	2	4
PIEN	12	0
PIPO	0	17
POTR	21	0
PSME	0	4

Table 10. Reconstruction of forest structure in 1880.

Forest Type	ABCO	ABLA	PIEN	PIPO	POTR	PSME	RONE	Total
				Tree Density (trees/hectare)				
Aspen	12.0 (5.3)	5.4 (2.7)	12.4 (4.0)	43.9 (11.4)	167.7 (29.5)	10.0 (4.1)	n/a	251.4 (37.9)
Mixed Conifer	59.3 (15.1)	2.7 (1.6)	10.9 (3.6)	64.9 (7.3)	59.4 (11.9)	45.8 (13.2)	n/a	242.8 (14.4)
Ponderosa Pine	30.3 (20.3)	0	0	159.1 (48.5)	142.1 (50.0)	5.0 (5.0)	n/a	336.5 (49.6)
Spruce-Fir	10.1 (5.6)	11.9 (2.9)	39.1 (6.4)	14.3 (5.3)	60.3 (10.3)	14.2 (5.0)	n/a	149.8 (14.8)
				Basal Area (m^2/hectare)				
Aspen	1.0 (0.5)	0.3 (0.2)	1.0 (0.4)	4.9 (1.1)	2.5 (0.7)	1.2 (0.7)	n/a	10.8 (1.8)
Mixed Conifer	5.5 (1.3)	0.1 (0.1)	0.6 (0.3)	5.4 (1.0)	0.7 (0.2)	5.4 (1.8)	n/a	17.6 (2.1)
Ponderosa Pine	1.9 (1.3)	0	0	9.6 (4.1)	1.4 (0.5)	0.4 (0.4)	n/a	13.3 (5.3)
Spruce-Fir	1.1 (0.6)	0.9 (0.2)	3.9 (0.8)	0.9 (0.3)	0.9 (0.2)	2.0 (0.8)	n/a	9.7 (1.2)

Table 11. Substrate frequencies by cover type (%).

Cover type	Plant	Litter	Soil	Rock	Wood	Bole	Stump	Scat
Aspen	27.1	58.7	1.2	1.0	11.2	0.9	0	0
Mixed Conifer	21.6	63.1	0.8	1.7	11.6	1.0	0.1	0.1
Ponderosa Pine	16.7	72.1	1.8	1.1	7.5	0.8	0	0
Spruce-Fir	25.3	58.5	2.5	1.5	11.5	0.7	0.02	0.01

Table 12. Forest floor and woody debris Woody debris is listed by moisture timelag class: 1H = wood 0-0.64 cm diameter, 10H = 0.64-2.54 cm, 100H = 2.54-7.62 cm, 1000H > 7.62 cm. The 1000H categories is divided into sound and rotten categories.

Forest Type	Litter depth (cm)	Duff depth (cm)	1H (Mg/ha)	10H (Mg/ha)	100H (Mg/ha)	1000H Sound (Mg/ha)	1000H Rotten (Mg/ha)	Total Woody Debris (Mg/ha)
Aspen	0.9 (0.1)	2.7 (0.4)	1.5 (0.2)	3.1 (0.5)	8.7 (1.0)	44.2 (7.2)	41.6 (10.6)	99.1 (16.8)
Mixed Conifer	0.9 (0.1)	2.4 (0.2)	1.5 (0.2)	3.8 (0.4)	8.0 (1.1)	47.9 (10.1)	57.8 (15.2)	119.1 (21.5)
Ponderosa Pine	1.6 (0.2)	2.9 (1.0)	0.5 (0.1)	3.4 (0.1)	4.1 (1.1)	76.2 (60.7)	10.4 (4.8)	94.6 (65.3)
Spruce-Fir	0.8 (0.1)	2.3 (0.3)	1.6 (0.2)	3.0 (0.3)	6.7 (0.9)	65.0 (11.6)	65.6 (14.9)	141.9 (24.9)

Table 13. Species richness and diversity by cover type.

Cover type	Ave. Richness	Simpson's D'	Simpson's D	Shannon's H'
Aspen	30.6	14.62	0.93	3.15
Mixed Conifer	27.5	13.32	0.92	2.99
Ponderosa Pine	27.5	10.51	0.90	2.62
Spruce-Fir	29.6	14.54	0.93	3.27

Table 14. Species nativity frequencies, from point-line intercept transects, by forest type.

Forest type	Nativity	Absolute frequency (%)	Relative frequency (%)
Aspen	Introduced	0.3	0.9
	Native	27.5	98.8
	Unknown	0.1	0.2
Mixed Conifer	Introduced	0.1	0.4
	Native	21.1	99.2
	Unknown	0.1	0.4
Ponderosa Pine	Introduced	0.7	3.9
	Native	16.5	96.1
Spruce-Fir	Introduced	0.2	0.6
	Native	25.1	97.0
	Unknown	0.6	2.4

Table 15. Introduced species frequencies from point-line intercept transects, by cover type.

Cover type	Species	Absolute frequency (%)	Relative frequency (%)
Aspen	*Poa pratensis*	0.3	0.9
Mixed Conifer	*Poa pratensis*	0.1	0.4
Ponderosa Pine	*Poa pratensis*	0.7	3.9
Spruce-Fir	*Poa pratensis*	0.2	0.6
	Taraxacum officinale	0.01	0.04

Table 16. Introduced species aerial frequencies from belt transects, by cover type.

Cover type	Species	Frequency (%)
Aspen	*Poa pratensis*	23.1
	Taraxacum officinale	7.7
Mixed Conifer	*Poa pratensis*	6.7
Ponderosa Pine	*Poa pratensis*	25
Spruce-Fir	*Poa pratensis*	17.9
	Taraxacum officinale	14.3

Table 17. Legumes found on point-line intercept transects, and their frequencies, by cover type.

Cover type	Species	Absolute frequency (%)	Relative frequency (%)
Aspen	*Lotus utahensis*	0.6	2.0
	Robinia neomexicana	0.3	1.0
	Trifolium pinetorum	0.02	0.1
Mixed Conifer	*Lotus utahensis*	0.4	1.7
	Lotus wrightii	0.02	0.1
	Robinia neomexicana	0.8	3.7
Ponderosa Pine	*Lotus utahensis*	0.7	3.9
Spruce-Fir	*Lotus utahensis*	0.3	1.2
	Lupinus argenteus	0.01	0.04
	Lupinus hillii	0.3	1.0
	Trifolium sp.	0.02	0.1

Table 18. Total legume frequencies from point-line intercept transects, by cover type.

Cover type	Absolute frequency (%)	Relative frequency (%)
Aspen	0.9	3.1
Mixed Conifer	1.2	5.5
Ponderosa Pine	0.7	3.9
Spruce-Fir	0.6	2.3

Table 19. Graminoid and shrub species found on plots in the four cover types.

Cover type	Growth Habit	Species	Absolute Frequency (%)	Relative Frequency (%)
Aspen	Graminoid	*Blepharoneuron tricholepis*	0.3	0.9
	Graminoid	*Bromus anomalus*	0.2	0.7
	Graminoid	*Bromus ciliatus*	1.0	3.7
	Graminoid	*Bromus* sp.	0.02	0.1
	Graminoid	*Carex duriuscula*	1.6	5.9
	Graminoid	*Carex occidentalis*	0.2	0.7
	Graminoid	*Carex rossii*	1.3	4.8
	Graminoid	*Carex* sp.	0.1	0.3
	Graminoid	*Elymus elymoides*	0.3	1.2
	Graminoid	*Koeleria macrantha*	0.02	0.1
	Graminoid	*Muhlenbergia montana*	1.7	6.1
	Graminoid	*Poa fendleriana*	0.7	2.5
	Graminoid	*Poa pratensis*	0.3	0.9
	Shrub	*Ceanothus fendleri*	0.1	0.3
	Shrub	*Juniperus communis*	3.8	13.8
	Shrub	*Rosa woodsii*	0.02	0.1
	Shrub	*Rubus* sp.	0.02	0.1
	Shrub	*Mahonia repens*	0.5	1.7
Mixed Conifer	Graminoid	*Blepharoneuron tricholepis*	0.04	0.2
	Graminoid	*Bromus anomalus*	0.1	0.6

	Graminoid	*Bromus ciliatus*	0.3	1.5
	Graminoid	*Carex duriuscula*	1.7	8.0
	Graminoid	*Carex geophila*	0.1	0.4
	Graminoid	*Carex occidentalis*	0.5	2.6
	Graminoid	*Carex rossii*	1.8	8.7
	Graminoid	*Carex siccata*	0.3	1.2
	Graminoid	*Carex* sp.	0.1	0.6
	Graminoid	*Elymus elymoides*	0.3	1.2
	Graminoid	*Muhlenbergia montana*	0.02	0.1
	Graminoid	*Poa fendleriana*	0.9	4.2
	Graminoid	*Poa pratensis*	0.1	0.4
	Shrub	*Amelanchier* sp.	0.02	0.1
	Shrub	*Juniperus communis*	2.6	12.0
	Shrub	*Rosa woodsii*	0.02	0.1
	Shrub	*Mahonia repens*	0.2	0.9
Ponderosa Pine	Graminoid	*Bromus ciliatus*	0.2	1.3
	Graminoid	*Carex duriuscula*	1.1	6.6
	Graminoid	*Carex occidentalis*	0.5	2.6
	Graminoid	*Carex rossii*	1.1	6.1
	Graminoid	*Elymus elymoides*	0.1	0.4
	Graminoid	*Poa fendleriana*	0.3	1.8
	Graminoid	*Poa pratensis*	0.7	3.9
	Shrub	*Juniperus communis*	1.7	9.6
	Shrub	*Mahonia repens*	0.6	3.5
Spruce-Fir	Graminoid	*Blepharoneuron tricholepis*	0.5	2.1
	Graminoid	*Bromus ciliatus*	0.3	1.0
	Graminoid	*Bromus* sp.	0.1	0.5
	Graminoid	*Carex duriuscula*	1.5	5.8
	Graminoid	*Carex occidentalis*	1.0	3.7
	Graminoid	*Carex rossii*	1.3	5.0
	Graminoid	*Carex* sp.	0.3	1.1
	Graminoid	*Elymus elymoides*	0.3	1.2
	Graminoid	*Festuca arizonica*	0.01	0.04
	Graminoid	*Festuca ovina*	1.0	3.7
	Graminoid	*Festuca sp.*	0.03	0.1
	Graminoid	*Koeleria macrantha*	0.3	1.0
	Graminoid	*Muhlenbergia montana*	2.0	7.8
	Graminoid	*Poa fendleriana*	0.5	2.1
	Graminoid	*Poa pratensis*	0.2	0.6
	Graminoid	*Poa* sp.	0.1	0.3
	Shrub	*Ceanothus fendleri*	0.03	0.1
	Shrub	*Juniperus communis*	2.0	7.9
	Shrub	*Rosa woodsii*	0.01	0.04
	Shrub	*Rubus idaeus*	0.01	0.04
	Shrub	*Rubus* sp.	0.01	0.04
	Shrub	*Sorbus dumosus*	0.02	0.1
	Shrub	*Mahonia repens*	0.1	0.2

Table 20. Growth habit frequencies from point-line intercept transects, by cover type.

Cover type	Growth habit	Absolute frequency (%)	Relative frequency (%)
Aspen	Graminoid	7.8	27.9
	Shrub	4.5	16.1
Mixed Conifer	Graminoid	6.3	29.6
	Shrub	2.8	13.2
Ponderosa Pine	Graminoid	3.9	22.8
	Shrub	2.3	13.2
Spruce-Fir	Graminoid	9.4	36.2
	Shrub	2.2	8.4

Table 21. Life history frequencies from point-line intercept transects, by cover type. B=biennial, PB=biennial or perennial, and P=perennial.

Cover type	Life history	Absolute frequency (%)	Relative frequency (%)
Aspen	B	0.3	1.1
	PB	0.02	0.1
	P	27.5	98.8
Mixed Conifer	PB	0.02	0.1
	P	21.2	99.9
Ponderosa Pine	P	17.2	100
Spruce-Fir	B	0.1	0.2
	PB	0.3	1.1
	P	25.5	98.6

Table 22. Forest reconstruction in 1880 compared with Lang and Stewart's (1910:12) "spruce-balsam" or "mixed type" stand averages, trees/ha. Lang & Stewart used the common name "balsam fir" for both *A. concolor* and *A. lasiocarpa*. Lang & Stewart did not report POTR density.

	PIPO	PSME	Firs	PIEN	Total
Reconstruction: Trees $\geq$ 15.24 cm dbh	34.2	20.8	29.8	22.2	107.0
Lang & Stewart: Trees $\geq$ 15.24 cm dbh	45.7	19.3	30.5	21.9	117.4
Percent Difference	**-25%**	**7.7%**	**-2.3%**	**1.4%**	**-8.9%**
Reconstruction: Trees $\geq$ 30.48 cm dbh	20.2	12.4	12.6	9.6	54.8
Lang & Stewart: Trees $\geq$ 30.48 cm dbh	23.1	10.6	13.4	10.5	57.7
Percent Difference	**-12.6%**	**17%**	**-6%**	**-8.6%**	**-5%**

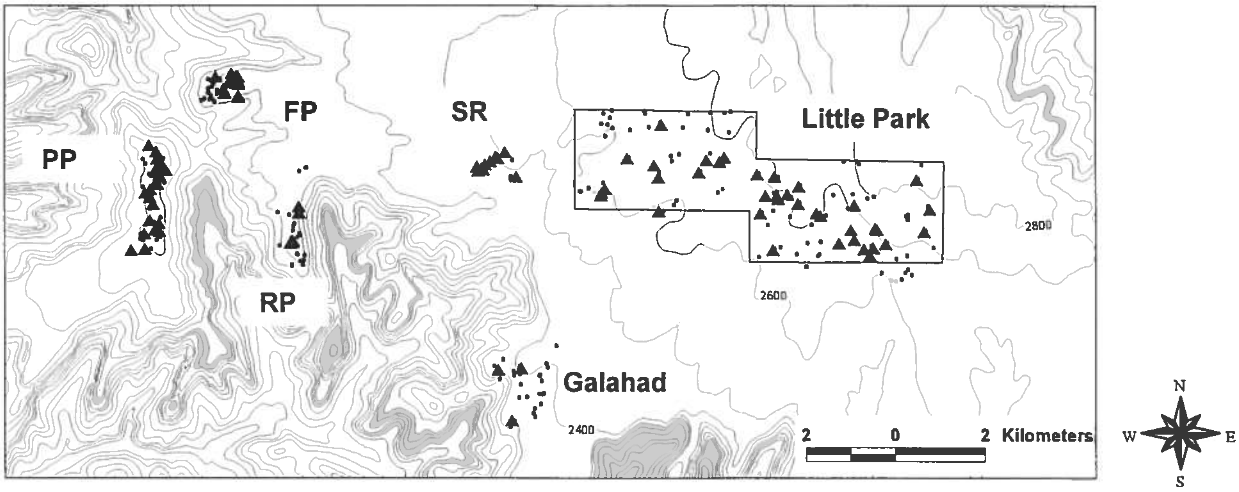


Figure 1. Study sites on the North Rim of Grand Canyon National Park. From west to east, sites are Powell Plateau (PP), Fire Point (FP), Rainbow Plateau (RP), Swamp Ridge (SR), Galahad Point, and Little Park. Forest structure and fire regimes of the first four sites were described by Fulé et al. (2002 and in review). Galahad and Little Park are described in this study. Symbols scattered across the study sites indicate locations of fire-scarred sample trees. Large triangles are trees that were scarred by fire in 1785. Smaller circles are trees not scarred in 1785.

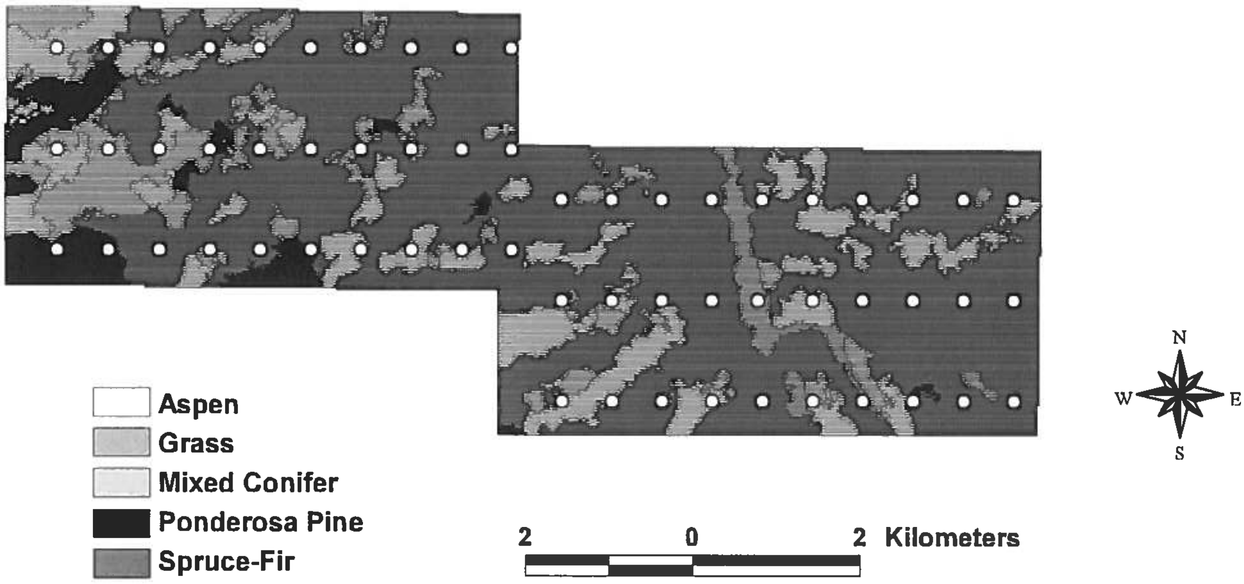


Figure 2. Vegetation types of the Little Park study site. Symbols indicate locations of permanent 20 X 50-m plots on a 600-m X 1,200-m sampling grid.

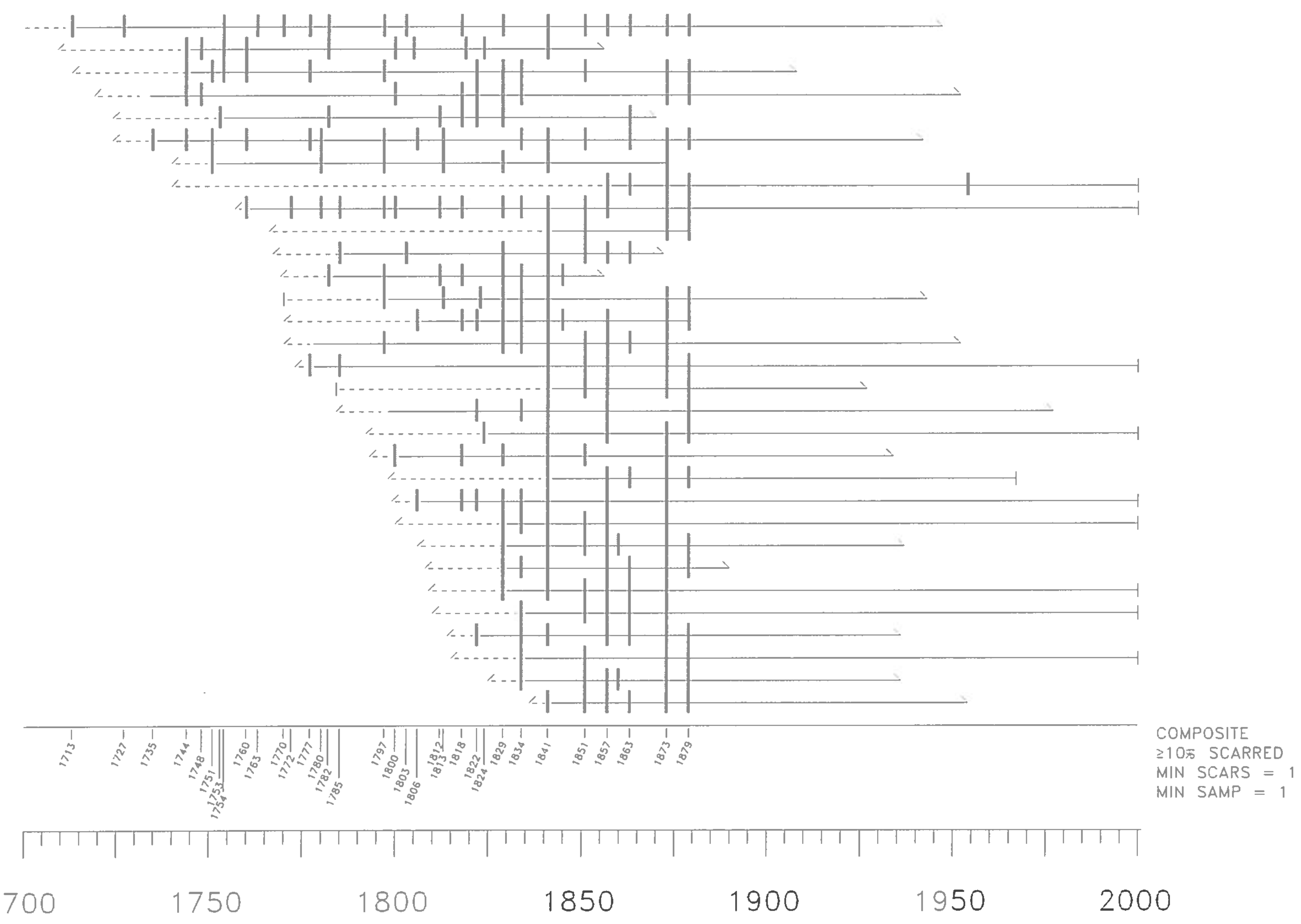
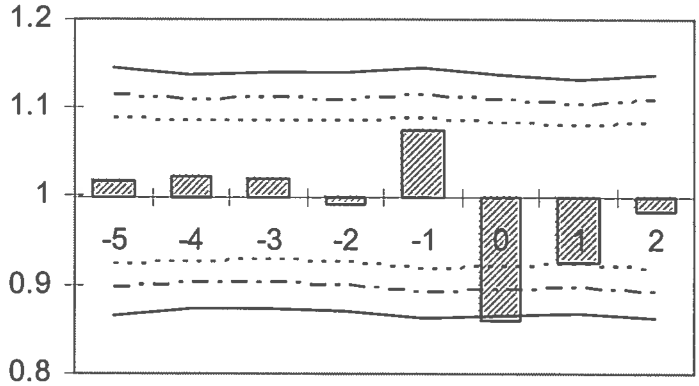


Figure 3. Fire chart for 31 fire-scarred trees from the Galahad study site. Years in which fires scarred 10% or more of the samples are shown on the lower axis.



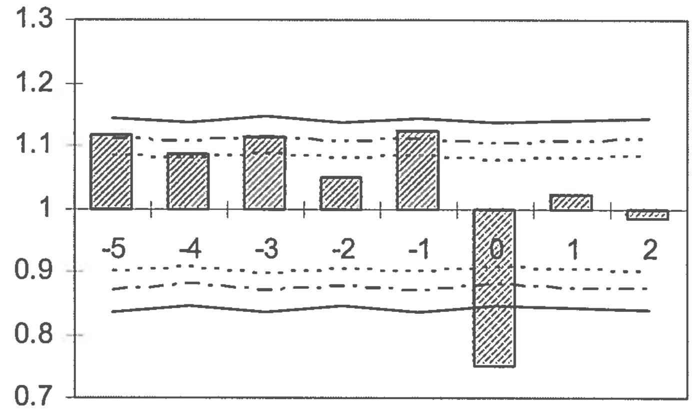


Figure 4. Fire years were significantly dry, as shown by superposed epoch analysis (SEA) showing the relationship between local climate (tree-ring width index) and occurrence of fires scarring 10% or more of the sample trees at the Galahad (top) and Little Park (bottom) study sites. The average climate value is scaled to one. Bootstrapping procedures were used to assess the statistical significance of climate departures above the mean ("wet years") and below the mean ("dry years") in the fire years (year 0), the five years preceding fires (-5 through –1), and the two years after fires (1 and 2). The three lines above and below the x-axis in each graph represent confidence intervals of 90%, 95%, and 99%.

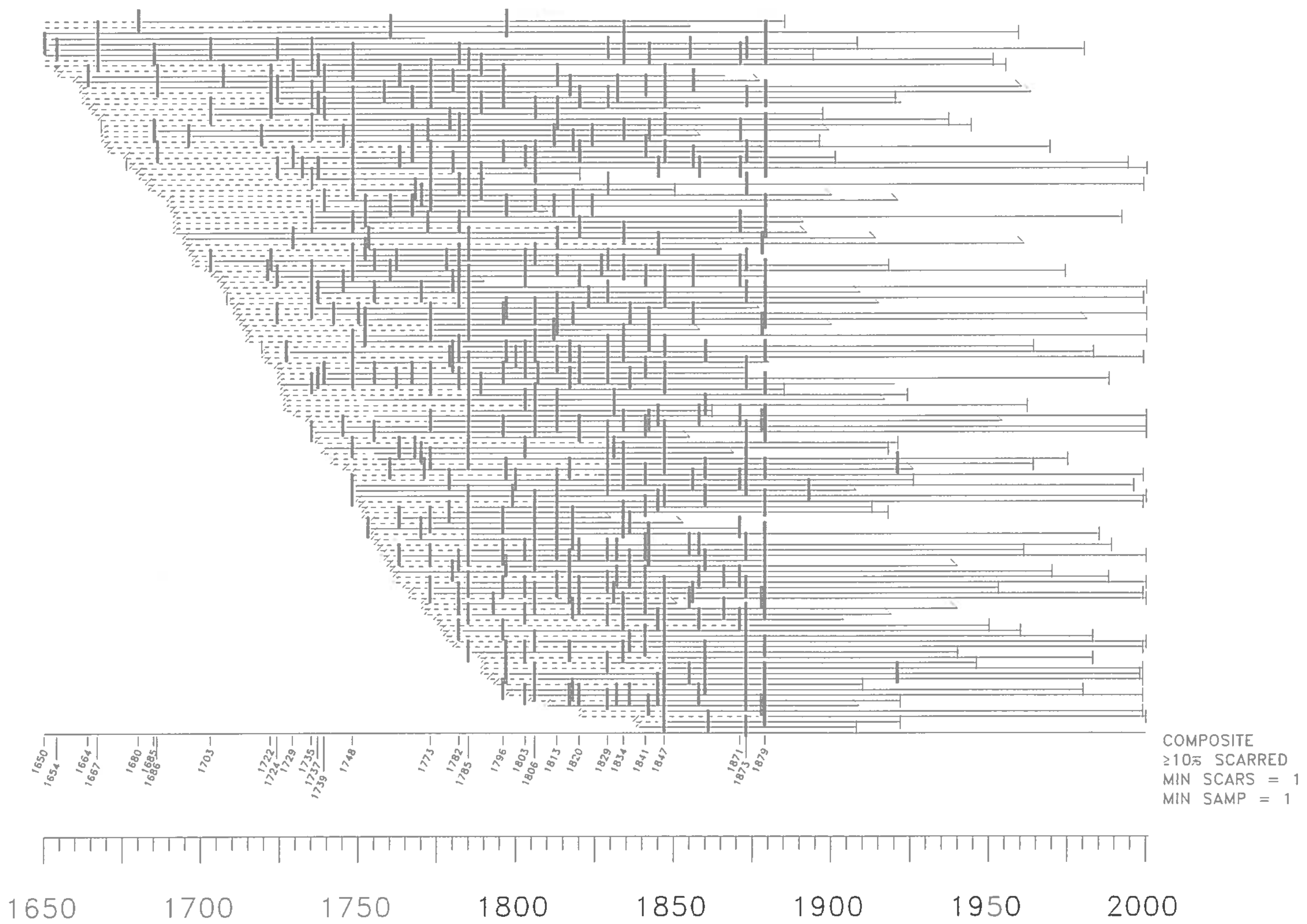


Figure 5. Fire chart for 132 fire-scarred trees from the Little Park study site. Years in which fires scarred 10% or more of the samples are shown on the lower axis.

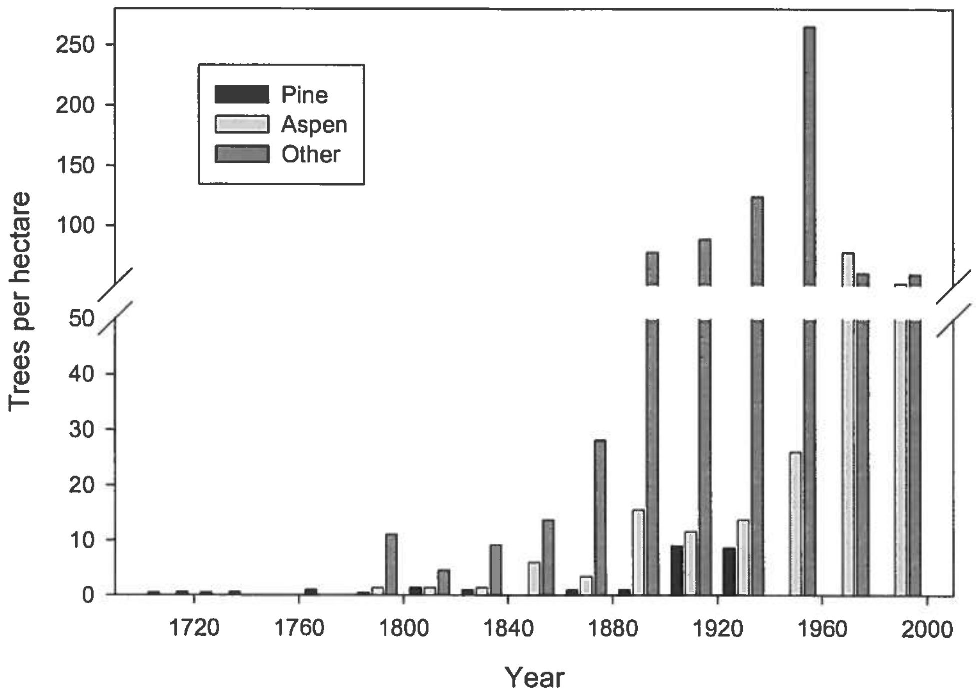


Figure 6a. Age distribution in spruce-fir forest.

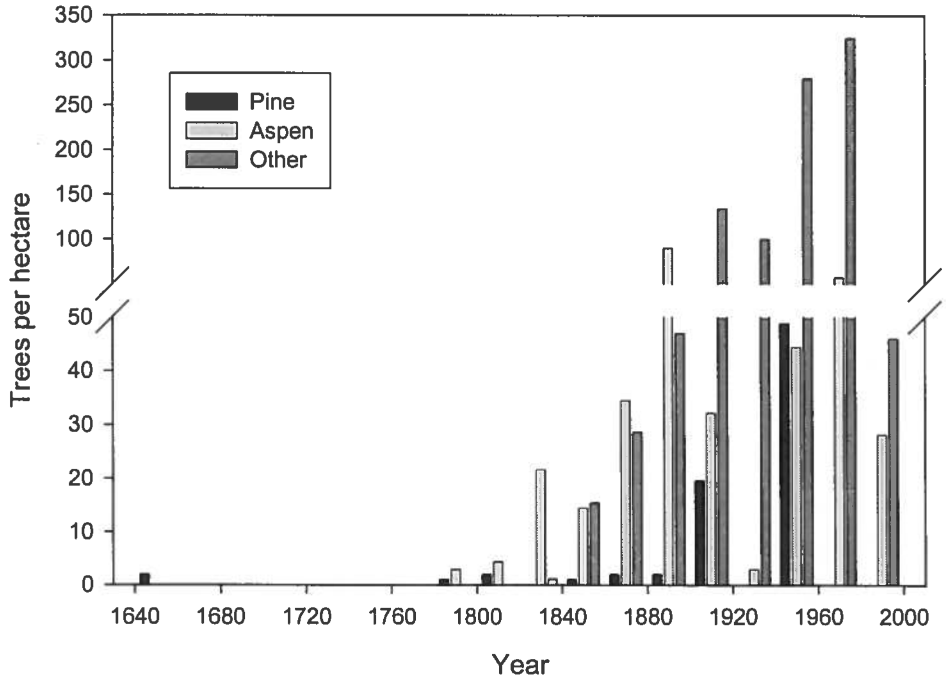


Figure 6b. Age distribution in aspen forest.

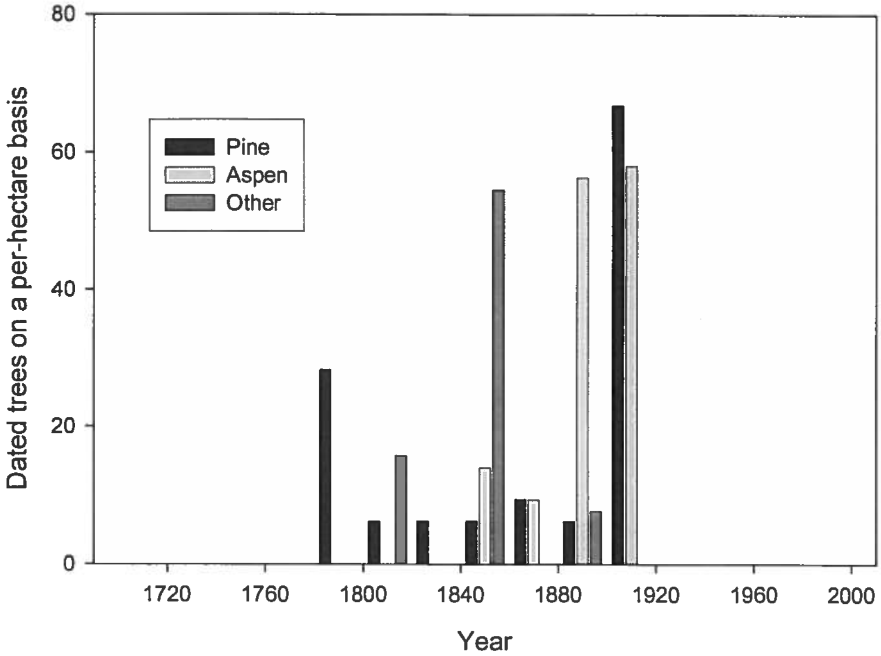


Figure 6c. Age distribution in pine forest. The sample size of dated trees in pine forest was inadequate to correctly express the data on a trees-per-hectare basis (see text).

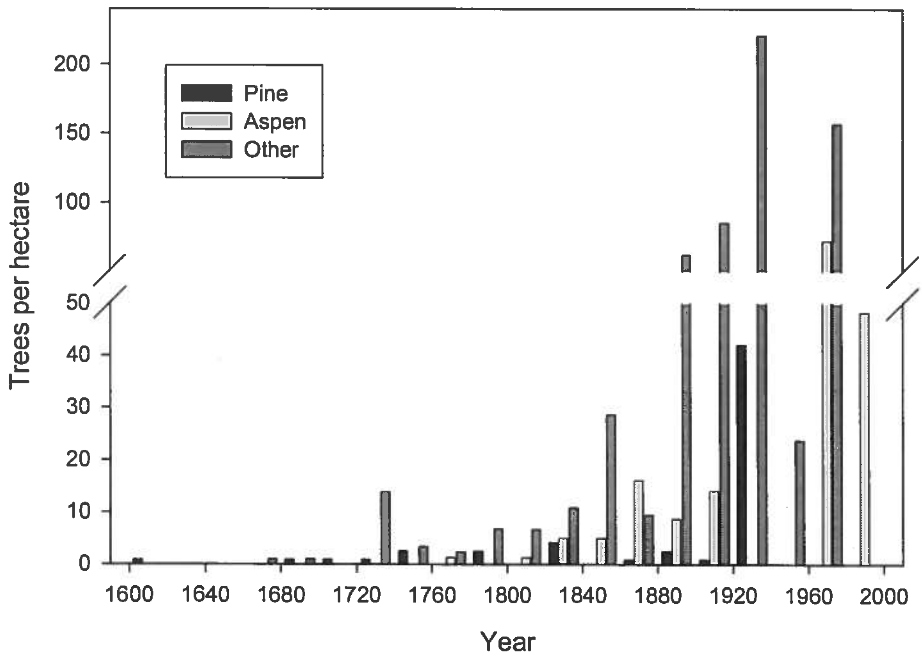


Figure 6d. Age distribution in mixed-conifer forest.

Changes in Canopy Fuels and Potential Fire Behavior 1880-2040: Grand Canyon, Arizona

Final Report to the Joint Fire Science Program, CA-1200-99-009-NAU 04 (Part 2)

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High-elevation forest on the North Rim of the Grand Canyon. At the time of the last extensive forest fire, in 1879, much of this forest was in early successional stages. By 2000, contiguous live and dead fuels had developed.

Executive Summary

We applied detailed forest reconstructions measured on broad-scale plot grids to initialize forest simulation modeling in 1880 and modeled spatially explicit changes in canopy fuels (crown biomass, crown bulk density, species composition) and potential fire behavior (crowning index) through 2040, a 160-year period. The study sites spanned a 500-m elevational gradient from ponderosa pine forest through higher-elevation mixed conifer, aspen, and spruce-fir forests on the North Rim of Grand Canyon National Park in northern Arizona. The simulations were relatively accurate, as assessed by comparing the simulation output in the year 2000 with field data collected in 1997-2001, because a regionally calibrated simulator was used (Central Rockies variant of the Forest Vegetation Simulator) and because we added regeneration by species and density in the correct historical sequence. Crown biomass increased at all sites, rising an average of 122% at the low-elevation sites and 279% at the high-elevation sites. The intermediate-elevation site, where mixed conifer vegetation predominated, began with the highest crown biomass in 1880 but had the lowest increase through 2040 (39%). Crown bulk density increased roughly in parallel with crown biomass; however, density values were considered less accurate in non-contemporary dates because they were based on assumptions about crown volume. Species composition of canopy fuels was consistent at low elevation (ponderosa pine) but shifted strongly toward mesic species at higher elevations, where ponderosa pine declined in absolute as well as relative terms. Potential crown fire behavior was assessed with the Nexus model in terms of crowning index (CI), the windspeed required to sustain active canopy burning. CI values decreased 23% to 80% over the modeled period. Canopy fuel and CI values were mapped across the entire North Rim landscape. At a threshold windspeed of 45 km/hr, only 6% of the landscape was susceptible to active crown fire in 1880 but 33% was susceptible by 2000. Implications of the changes over time and space include altered contemporary habitats and the high likelihood of rapid, broad-scale disturbance by fire or other agents. If managers choose to intervene to reduce canopy fuel mass and continuity, actions should be guided by the distinct ecological and geographic attributes of the different forest types.

Keywords

Kaibab Plateau, *Picea*, *Pinus*, *Abies*, *Pseudotsuga*, *Populus*, crown fire, fire hazard

Introduction

The Rodeo-Chediski fire complex in eastern Arizona in 2002 approached 200,000 ha in size, an order of magnitude greater than recent destructive and costly fires in the Southwest, such as the *circa* 20,000-ha Cerro Grande fire in New Mexico in 2000. As increasingly higher thresholds of crown fire are crossed, understanding the causes and characteristics of crown fire behavior has become urgent. Of the elements of the fire environment—weather, topography, and fuels— the features that can be modified by management include the mass, arrangement, and composition of fuels across the landscape. Thus information about vegetation patterns, change over time, and susceptibility to crown fire is highly useful for designing management strategies to mitigate undesirable fuel hazards, in concert with predictions of the effects of climate changes (Flannigan et al. 2000, Shafer et al. 2001)

Any fuel measurement and fire modeling approach should be driven by the intended use of the data—from broad planning purposes to detailed fire behavior predictions—within the constraints of financial resources and technological capabilities. Landscape fuels have been measured with a great variety of strategies (Keane et al. 2001). For example, at a broad scale, Schmidt et al. (2002) mapped several vegetation and fire regime variables at 1 km^2 resolution across the U.S., basing the vegetation maps on a synthesis of existing map data (e.g., Küchler's Potential Natural Vegetation map). At finer scale, Keane et al. (1998, 2000) mapped fire behavior fuel models (Anderson 1982) on large landscapes in Montana, Idaho, and New Mexico using a "triplet" of base vegetation characteristics: biophysical setting, cover type (species composition), and structural stage (vertical stand structure). The coupling of fuel data with vegetation simulation and fire behavior models provides an even broader array of approaches for modeling successional change, fire ignition, and fire spread. Rupp et al. (2001) classified landscape-fire-succession models along gradients of increasing complexity, for example, running from pathway or Markovian models with very generalized descriptions of plant communities to highly specific individual-plant and gap-phase models. Keane et al. (1999) integrated a detailed mechanistic biogeochemical process model, FireBGC, with a spatially explicit fire behavior model, FARSITE (Finney 1998), to link changes in climate with changes in fire regimes. Stochastic simulation approaches, such as those used by He and Mladenoff (1999) in Wisconsin or Miller and Urban (1999) in the Sierra Nevada, provide less precision about individual events but offer more realistic probability distributions about long-term change.

The Southwest is a microcosm of the variability in forests and fire regimes found throughout North America but forest types adapted to frequent surface-fire regimes predominate. Ponderosa pine and lower mixed conifer forests make up approximately 2,950,000 ha in Arizona and New Mexico (O'Brien 2002, Fiedler et al. 2002). These vegetation types comprise 86% of the high-elevation, non-woodland forests in Arizona (O'Brien 2002) and about 90% of these forests in New Mexico are considered at moderate or high risk for stand-replacing fire (Fiedler et al. 2002), due to increased stand density and fuel accumulation following fire exclusion and other management interventions (Covington and Moore 1994). The higher-elevation forest types, such as mixed conifer, aspen, and spruce-fir stands, are rare but important habitats, comprising only about 3% of Arizona's forests (O'Brien 2002). Although rarely studied, these forests are believed to include a mix of surface and stand-replacing fire regimes (White and VanKat 1993, Grissino-Mayer et al. 1995, Rollins et al. 2002, Fulé et al. in press). There is some evidence that recent successional changes in the fire exclusion period may be leading to compositional change (loss of aspen and meadows [Dahms and Geils 1997, Moore and Huffman in review]) and fuel accumulations (White and VanKat 1993, Fulé et al. in press).

The challenge for landscape management is to select the appropriate data and tools for modeling ecological change, then integrating process modeling with landscape pattern (Opdam et al. 2002). We reconstructed key fuel and fire behavior variables on a large southwestern landscape crossing a 500-m elevation gradient that encompassed forest types from ponderosa pine and mixed conifer to spruce-fir and aspen. We took advantage of detailed forest data collected on landscape-scale plot grids in a large, never-harvested forest at Grand Canyon National Park. The study objectives were to use a regionally-specific vegetation simulator and a fire behavior model to: (1) estimate the changes in canopy fuels across the sampled landscapes over time, from 1880 through the present and near future (2040); (2) estimate changes in the potential for crown fire behavior over time and space; and (3) assess ecological and management implications.

Methods

Study Area

The study area was a 6 X 18-km transect on the Kaibab Plateau, the North Rim of the Grand Canyon, in northern Arizona (Figure 1). A series of study sites was arranged along an elevational gradient from Powell Plateau (approximately 2,300 m elevation) up to the highest elevation in Grand Canyon National Park (2,794 m in the Little Park East study site), comprising a total of 5,345 ha. The forest structures and fire regimes at the study sites were described in detail by Fulé et al. (2002, 2003, in press).

Average annual precipitation at the North Rim ranger station (elevation 2,542 m) is 58 cm, with an average annual snowfall of 328 cm. Temperatures range from an average July maximum of 26° C to an average January minimum of -2° C (White and Vankat 1993). Soil information was derived from an ongoing soil survey (A. Dewall, National Resource Conservation Service, personal communication 2002). Soils at the western sites were Typic Paleustalfs. At the Little Park West and Little Park East sites, soil textures ranged from coarse to fine loams. Valley soils at these sites were Cumulic Haplustolls, soils on 15-40% slopes were Oxyaquic Paleustalfs, and flatter upland areas (2-15% slopes) were Cumulic Haplustolls.

Forests in the western study sites were dominated by ponderosa pine with Gambel oak and New Mexican locust (scientific names and species codes are given in Table 1). At intermediate elevation (Swamp Ridge study site, approximately 2,500 m), tree species included ponderosa pine, aspen, white fir, and Douglas-fir. At the highest elevation sites, Little Park West and Little Park East, tree species included ponderosa pine, aspen, white fir, subalpine fir, and spruces. Engelmann and blue spruce were combined in this study because of difficulties in distinguishing young trees (Moore and Huffman in review) and our observation of trees at the study site that had characteristics intermediate between the two species.

Image Classification

Image classification was described in detail by Crouse and Fulé (2003) and is summarized here. We used a Landsat Enhanced Thematic Mapper (ETM) image, acquired 6 June 2000. TM imagery has been shown to be the best product for consistently mapping vegetation for large land areas (Keane et al. 2000). Landsat Thematic Mapper imagery has been used extensively for vegetation mapping projects (Golden 1991, Congalton et al. 1993, Muldavin et al. 1998, Vanderzanden et al. 1999, Keane et al. 2000). Image classification and analysis was done using Erdas Imagine (Leica Geosystems AG, Switzerland). Field plot data (described below) collected from 1997 to 2002 were used as training sites for the classification process. The training sites were digitized directly on the ETM imagery to create a supervised training site signature file. Each training site polygon included at least 10 pixels including the pixel or pixels that corresponded with each EM plot center.

The classification scheme for this project was developed within the National Vegetation Classification Standards (NVCS) framework. The NVCS has been developed through the USGS-NPS Vegetation Mapping Program in association with USGS/BRD, EPA, National Park Service, The Nature Conservancy, Ecological Society of America, and others (USGS 2000). Field plot data were analyzed to determine the species labels that were assigned to each of the training sites. The species label for each training site was based on "importance value" (Taylor 2000), the sum of the relative frequency (percent stems) and relative abundance (percent basal area) for each species (Table 2).

A classification using a minimum distance algorithm was run using the combined training site signature set. The classified image was checked using aerial photographs (National High Altitude Photography program 1:40,000 color infrared prints) and individuals familiar with the area assisted in refining the classification. Misclassed pixels were identified using an elevation-based model developed in Imagine. For example, vegetation types such as Gambel oak and New Mexico locust do not occur at the higher elevations of the study area. The model "flagged" these vegetation types if they occurred above 2250 m. These pixels were then edited to reflect the correct vegetation type. Mixed conifer pixels below 2200 m were also identified and edited.

Canopy cover was mapped into four forested classes and four non-forested classes (Table 3) using photo-interpreted training sites. During the supervised classification process, training sites were added as needed to refine the classification. A total of fifty-eight training sites were used to develop the supervised classification.

Forest Measurements

Vegetation was measured on plots based on the National Park Service's Fire Monitoring plots (Reeberg 1995), with modifications to collect detailed tree condition and dendroecological data. This plot design was chosen to correspond with NPS monitoring and because the relatively large plots are useful for capturing variability of clumps of old trees (Fulé et al. 1997). A total of 166 plots was measured between 1998 and 2001 (Figure 1). Sampling plots were 0.1 ha (20 x 50 m) in size, oriented with the 50-m sides uphill-downhill to maximize sampling of variability along the elevational gradient and to permit correction of the plot area for slope. Plots were permanently marked to permit re-measurements in the future. Iron stakes were sunk flush to the forest floor at the corners and center of each plot and a large tree was tagged with the distance and bearing to the plot center. Photos were taken at the corners and quarter-corners of each plot.

Trees larger than 15 cm diameter at breast height (dbh) were measured on the entire plot (1000 m^2) and trees between 2.5-15 cm dbh were measured on one quarter-plot (250 m^2); all trees were tagged. Tree attributes measured were: species, dbh, height, crown base height, dwarf mistletoe rating (0-6, Hawksworth and Geils [1990]), and tree condition. To assess changes in forest structure, we differentiated between trees established prior to 1880 and those that established later. Previous research in northern Arizona showed that ponderosa pines with dbh > 37.5 cm or ponderosa of any size with yellowed bark could be conservatively identified in the field as being of pre-1880 origin (White 1985, Mast et al. 1999). We used the same diameter breakpoint, 37.5 cm, for other conifers (Fulé et al. 2002) and > 20 cm for aspen trees. "Conservative" identification meant that these criteria included all pre-1880 trees as well as numerous post-1880 trees. Tree status (pre/post 1880) was later corrected in the laboratory using age data. All living trees meeting the field criteria above were considered potentially pre-1880 trees and were cored at 40 cm above ground level. A random 10% sample of all trees that did not meet the field criteria was also cored. Seedling trees, those below 2.5 cm dbh, were tallied by species, condition, and height class in a 50 m^2 subplot. Canopy cover measured by vertical projection (Ganey and Block 1994) was recorded at 30 cm intervals along the two 50-m sidelines of each plot.

Forest plot area was corrected for slope by multiplying the 50-m dimension by the slope correction factor. Tree increment cores were surfaced and visually crossdated (Stokes and Smiley 1968) with tree-ring chronologies we developed. Rings were counted on cores that could not be crossdated, especially younger trees. Additional years to the center were estimated with a pith locator (concentric circles matched to the curvature and density of the inner rings) for cores that missed the pith (Applequist 1958). From previous studies (Fulé et al. 2000, 2003, in press), we know that fire exclusion began after 1879 on most of the North Rim. Forest structure in 1880 was reconstructed using dendroecological methods (Fulé et al. 1997, 2002, in press) as follows: tree size at the time of fire exclusion was reconstructed by subtracting the radial growth measured on increment cores since 1879. We developed local species-specific relationships between tree diameter and basal area increment (r^2 = .45 to .90) and applied these relationships to estimate past size for trees without increment cores (dead or rotten centers). For dead trees, the date of death was estimated based on tree condition class using diameter-dependent snag decomposition rates (Thomas et al. 1979).

Modeling and Canopy Fuels

A schematic diagram of the modeling process is shown in Figure 2, leading from plot data to modeling of vegetation change and fire behavior. Forest change was simulated on a per-plot basis at 10-year intervals using the Forest Vegetation Simulator (FVS, Van Dyck 2000), Central Rockies variant. This model was chosen because it is a highly precise and site-specific statistical model for short-term simulation of forest change (Edminster et al. 1991), realistically simulating the specific 160-year period under study, in contrast to a long-term ecological process modeling approach (e.g., Keane et al. 2002) that could provide a broader but less precise range of outputs. Simulations were initialized with reconstructed 1880 conditions. Actual regeneration data by species and decade were added to each plot. Regeneration densities were increased by 40%, an empirically determined value, to account for density-dependent mortality. For example, if data for a given plot showed that 10 ponderosa pine trees established between 1880-1890, then we inserted 14 trees (140%) into the simulation year 1890. Simulations were run until 2040. Simulation data at the year 2000 were checked against field plot data that were collected between 1998 and 2002. The simulation values were within ± 20% of field values for tree density (trees/ha) and basal area.

Tree lists were exported from FVS for each plot and decade. Crown biomass was estimated with dbh-based allometric equations for foliage and fine twigs, the canopy fuels that are available for burning in crown fires (Scott and Reinhardt 2001). Allometric equations developed in or near the Southwest were selected as follows: ponderosa pine (Fulé et al. 2001), Gambel oak (Clary and Tiedemann 1986), juniper (Grier et al. 1992), white fir (Westman 1987), Douglas-fir (Gower et al. 1992), and equations compiled by Ter-Mikaelian and Korzukhin (1997) were used for aspen, subalpine fir, and Engelmann spruce. Crown volume was estimated on a per-plot basis by the maximum tree height (top of the canopy) minus crown base height (bottom of the canopy). Plots measured before 1999 had a single average crown base height value per plot. From 1999 onward (sites LPW and LPE), crown base heights were measured per tree and the average of these measurements was used for the plot value. Crown bulk density was calculated as crown biomass divided by crown volume.

Fire behavior was modeled with the Nexus Fire Behavior and Hazard Assessment System (Scott and Reinhardt 2001). Nexus required inputs of canopy biomass and bulk density, as described above. Foliar moisture content was set at 100% (Agee et al. 2002). Field measurements of plot slope were used. Hot and dry weather inputs were used to simulate fire behavior under extreme conditions. Fire weather extremes representing the 90^{th} and 97^{th} percentiles of low fuel moisture, high winds, and high temperature were calculated for 1970-2001 at the Bright Angel weather station (North Rim, Grand Canyon National Park, elevation 2560 m) using the FireFamily Plus program (Bradshaw and Brittain 1999). Weather values were calculated for June, historically the month with the most severe fire weather (Table 4). The wind reduction factor was 0.3 for all simulations. Fire behavior fuel models (Anderson 1982) used were 9 (sites below 2,500 m elevation) and 10 (sites above 2,500 m elevation).

A variety of fire behavior outputs are provided by Nexus, including both surface and crown fire behavior variables. As discussed below (see *Data Reliability*), our assumptions about past values of crown base height and fuel model would have heavily influenced interpretation of other fire behavior variables such as flame length or torching index (windspeed required for passive crown fire). Therefore we focused on crown fire behavior as measured by a more reliable variable, the crowning index (CI), defined as the windspeed at which active canopy burning could be sustained (Scott and Reinhardt 2001).

The variables of crown biomass, crown bulk density, and crowning index were mapped to the plot coordinates. Landscape patterns of these variables at the six study sites were estimated using negative exponential interpolation between plots. We also attempted to model CI on the entire North Rim landscape by looking for relationships between CI and slope, azimuth, elevation, and canopy cover within each vegetation type. Although some of the correlations between variables were statistically significant at an alpha level of 0.05, the predictive capabilities were very low ($r^2 < 0.1$) so these models were not used. Similar difficulties in relating fuel or fire characteristics to topographic factors or environmental gradients have been reported in other fuel modeling studies (Keane et al. 2000). Instead, we reclassified the image using the 1880 and 2000 CI values as training sites. The classification gave realistic outputs but the accuracy of the map, based on linked model results, cannot be quantified.

Results

Changes in Canopy Fuels, 1880-2040

Crown biomass increased at all study sites between 1880 and 2040, but the changes were most dramatic at the high-elevation LPW and LPE sites (Figure 3, see Figure 1 for site names). At the three lower-elevation sites, PP, FP, and RP, crown biomass ranged from 3,800 to 5,600 kg/ha in 1880 (mean 4,473 kg/ha). By 2040, crown biomass increased to 8,510 to 9,940 kg/ha (mean 9,022 kg/ha), an average increase of 122%. The higher-elevation sites LPW and LPE also had low crown biomass values in 1880, 3,800 and 4,400 kg/ha, respectively. But by 2040, LPW crown biomass increased to 14,270 kg/ha and LPE crown biomass rose to 16,800 kg/ha, an average increase of 279%. The intermediate-elevation site, SR, had the highest 1880 crown biomass (8,370 kg/ha) but increased by the smallest proportion of any study site, only 39%, to a value of 11,700 kg/ha in 2040.

Species composition changes differed by elevation. At the three low-elevation sites, almost all of the crown biomass (> 99%) consisted of ponderosa pine throughout the simulated 160-year period. But the proportion of ponderosa pine crown biomass declined at the higher sites. In the intermediate-elevation SR site, at approximately 2,500 m elevation, the crown biomass contribution from non-pine species (white fir, Douglas-fir, and aspen) increased from 31% in 1880 to 61% in 2040. At site LPW, approximately 2,650 m elevation, the non-pine component rose from 65% in 1880 to 86% in 2040. Engelmann spruce and subalpine fir alone made up 50% of total crown biomass by 2040, rising from only 19% in 1880. The highest site, LPE, approximately 2,700 m elevation, started with 86% non-pine crown biomass in 1880 and increased to 96%. Engelmann spruce and subalpine fir alone made up 36% of total crown biomass in 1880 and 67% in 2040.

The changes in crown biomass were interpolated across the landscape of the study sites at 60-year intervals (1880, 1940, and 2000) (Figure 4). In 1880, the entire landscape had relatively low crown biomass values, with the highest values at mid-elevation (SR). By 1940, however, higher crown biomass values appeared in LPW and LPE, a pattern that became stronger with time. By 2000, heavy crown fuels covered about 2/3 of the LPE site.

Crown bulk density values closely paralleled crown biomass values (Figure 5), as expected, but with minor variation due to plot-to-plot differences in crown volume. The increases in crown bulk density ranged from 34% (SR) to 291% (LPE) (Table 5). Summarized by vegetation type rather than by study site, the spruce-fir type had the highest increase (371%), while ponderosa pine had the lowest (78%). In 1880, the absolute values of crown bulk density were below 0.05 kg/m^3 in every vegetation type and every study site except SR. By 1960, all vegetation types and all study sites except the lowest elevation sites (PP and RP) had crown bulk density > 0.05 kg/m^3. By 2040, four of the six study sites and three of the four vegetation types had crown bulk density > 0.07 kg/m^3 (Table 5).

Changes in Potential Fire Behavior, 1880-2040

High winds were required for crown fire spread at all sites in 1880, as shown by values of crowning index (CI), the windspeed required to sustain active crown fire, plotted for each study site in Figure 6. The lower-elevation sites dominated by ponderosa pine, PP, FP, and RP, had CI values ranging from 96 to 156 km/hr (average 128 km/hr) in 1880, declining to 55-71 km/hr (average 65 km/hr) in 2040, a decrease of 49%. CI values at the intermediate-elevation site, SR, dropped from 62 to 48 km/hr between 1880 and 2040, a 23% decline. At the higher-elevation sites, LPW and LPE, CI values were extremely high in 1880: 272 and 371 km/hr, respectively. By 2040, the CI dropped to 54 km/hr at LPW (80% decline) and to 52 km/hr at LPE (86% decline). Approximately half of the SR study site and 1/3 of the LPW and LPE study sites were susceptible to crown fire at windspeeds of 45 km/hr or less by 2000.

Modeled across the entire North Rim, 33% of the landscape was susceptible to crown fire at a windspeed threshold of 45 km/hr in 2000, compared to less than 6% in 1880 (Figure 7). The large area classified as "burn" in Figure 7 is the Outlet fire (May, 2000). Both in 1880 and 2000, the lowest values of CI were clustered in mixed conifer forests located just above the ponderosa forests on the canyon rim. By 2000, many higher-elevation spruce fir forests at the center of the plateau were also susceptible, as well as linear spruce-fir stands located on north slopes of drainages.

Discussion

Data Reliability

Model results should always be used with caution and accuracy is important in any study modeling ecological characteristics over time and space. The most accurate data were the direct measurements on the field plots, providing contemporary values of tree density and size by species. The accuracy of the forest reconstructions at these sites was compared in detail by Fulé et al. (2002, in press) to multiple lines of evidence from a historical inventory (Lang and Stewart 1910), other historical accounts (e.g., Dutton 1882), and ecological data (e.g., Rasmussen 1941, Mitchell and Freeman 1993). Reconstructed forest structure matched closely with the earliest inventory data: forest density was within 5-10% of the historical data, closely matching the historical distribution by species in multi-species sites (Fulé et al. 2002, in press). We expected vegetation simulation to be relatively accurate because we chose a local statistical model, the Central Rockies variant of the Forest Vegetation Simulator (FVS), and because we input plot-specific data on regeneration density by year and species. This expectation was borne out by the fact that simulated densities and basal areas in 2000 were within ± 20% of field-measured values.

Crown biomass was estimated with published regression equations with high predictive capability ($r^2 \sim .8\text{-}.9$) and local equations were selected whenever possible, indicating that crown biomass changes over time should be relatively accurate. The next step in canopy fuel modeling was to calculate crown bulk density. This estimate required assumptions about crown volume that probably affected the results. We calculated crown volume from the top (height) and bottom (crown base height) of the contemporary plots and assumed the same volume for all simulated decades. An alternative choice would have been to use FVS height and crown ratio estimates to estimate crown volume. However, while FVS was accurate in simulating tree numbers and diameters, we found many inconsistencies in height even though sites were initialized with local site index data. Our use of contemporary crown volume measurements has two implications. First, it meant that current crown bulk density estimates (e.g., 1980-2020) were probably more accurate than distant estimates (e.g., 1880-1920). In particular, crown bulk density was most likely underestimated in early decades in the regenerating stands at high elevation (sites LPW and LPE, Figure 5). Although crown biomass was very low on many of those plots due to severe fires, especially in 1879 (Fulé et al. in press), crown volume would also have been lower due to the small stature of the young trees. Second, the fact that we were not able to estimate crown base height in the past or future led us to remove torching index (windspeed required for passive crown fire) from the fire behavior analysis, because simulated torching is highly sensitive to crown base height (Scott and Reinhardt 2001).

The use of allometric crown biomass equations produces estimates that are approximately 40-60% lower than the crown bulk density values based on the data of Brown (1978). For example, the "high cover" mesic mixed conifer stands in the Gila National Forest (New Mexico) were estimated to have crown bulk density values of 0.25 kg/m^3 (Keane et al. 2000), in contrast to maximum values of 0.094 kg/m^3 in the present study (62% lower). Therefore comparisons of crown bulk density values across studies that used different methods should be based on relative, not absolute, differences. The FARSITE model (Finney 1998) is calibrated to accept higher crown bulk density values, while the Nexus model (Scott and Reinhardt 2001) used in the present study provides realistic crown fire behavior estimates with the lower crown bulk density values (E. Reinhardt, personal communication, 2003).

The assumptions and uncertainties associated with fire behavior modeling have been discussed elsewhere (Van Wagner 1977, Rothermel 1991, Finney 1998, Scott 1998, Scott and Reinhardt 2001). Our objective was not to improve upon existing models but rather to use the Nexus model with detailed local data to develop a fair comparison of potential fire behavior in relative terms. Thus it would be unrealistic to expect that CI values are precise estimates of the exact windspeed at which any real crown fire will be sustained. However, it is reasonable to compare CI values across space and time to assess crown fire susceptibility in relative terms.

Spatial patterns were relatively consistent between sites and decades (Figures 4-6) because sampling was done on regular grids, a strategy for minimizing interpolation error (cite). Spatial accuracy was high at each mapped plot (within the ± 15 m accuracy of handheld global positioning system units) and unknown between plots. However, the same interpolation procedures were used for each map, so comparisons between maps are consistent.

Ecological Implications

Substantial changes in forest composition, structure, and susceptibility to fire occurred over the large landscapes of the study sites since 1880 and similar trends are forecast for the next few decades, barring disturbance or management intervention. At sites ≥ 2,500 m elevation, the majority of the increases occurred in mesic species, reversing in a century the 10,000-year trend of upward movement of xeric species on the Kaibab Plateau (Weng and Jackson 1999). Future disturbances are likely to be widespread and severe in terms of tree mortality, whether from fire or biological pathogens, especially if influenced by warmer and drier climate (Flannigan et al. 2000).

The western ponderosa pine sites (PP, FP, and RP) were described by Fulé et al. (2002, 2003) as rare examples of unharvested forests with relatively undisrupted surface fire regimes. Each of these sites had two or three large surface fires after 1879 and several smaller fires (Fulé et al. 2003). Forest structural change as measured by basal area was relatively low at these sites, rising from an average of 18.5 m^2/ha in 1880 to 28.3 m^2/ha at present; this 53% increase in basal area was much less than the 152% increase observed in another unharvested ponderosa forest where fire was completely excluded on the South Rim of Grand Canyon (Fulé et al. 2002). The rise in crown biomass at the western ponderosa sites, (89% for 1880-2000) was disproportionately larger than the basal area increase (53%). The western sites are relatively open, 193-249 pines/ha (Fulé et al. 2002), compared to an Arizona-wide average in 1999 of approximately 618 pines/ha (O'Brien 2002), and arguably present a lesser fuel hazard due to the dominance of tall, old, fire-pruned trees. Nonetheless, even these relatively natural forests have increased in canopy fuel loading, perhaps reflecting an intermediate effect of the reduced, but not eliminated, surface fire regime.

At the higher elevation sites, shifts in species composition were as notable as the increase in crown biomass. In every case, fire-resistant ponderosa pine declined in relative terms, as has been observed in other mixed conifer settings in the absence of fire (e.g., Minnich et al. 1995, Stephenson 1999). The absolute value of ponderosa pine biomass was also estimated to decline, as early as 1940 at the intermediate-elevation SR site and after 1960 at the higher sites (Figure 3). Douglas-fir, a species with intermediate fire resistance (relatively susceptible as a seedling but resistant when mature [Miller 2000]), retained a relatively consistent proportion of crown biomass (Figure 3). The major increases in crown biomass occurred in white fir (Figure 8), subalpine fir, and spruce (Figure 3), relatively fire-susceptible species.

Habitat changes associated with forest alteration include the loss of early-successional environments at high elevation. Approximately 60% of the LPW and LPE landscapes were initiated by stand-replacing fire between 1782 and 1879 (Fulé et al. in press), creating open forests with abundant snags. These conditions may be important for bird diversity in general and especially for certain guilds such as woodpeckers, flycatchers, and seedeaters (Hutto 1995).

Changes in composition of mixed stands affect potential fire behavior in several ways. First, since 1880 surface fuels have become increasingly comprised of relatively more tree litter and relatively less herbaceous fuels, tending to reduce fire rate of spread but to increase surface heating (Anderson 1982). Second, the proportion of long-needled ponderosa pine litter, a resinous and loosely compacted fuel, has declined relative to short-needled conifer litter (Douglas-fir, spruce, and fir) that is less flammable and more compacted, tending to reduce both rate of spread and surface heating. However, the overall effect on surface fire behavior is counteracted by the presence of numerous jackstrawed fallen logs in mixed conifer and spruce-fir stands (P.Z. Fulé, unpublished data), leading us to select fuel model 10 ("light logging slash," Anderson 1992) for modeling. In terms of crown fire behavior, the historical shift toward mesic species is likely to have lowered crown base heights because spruce and firs have relatively long crowns (Miller 2000), facilitating crown fire initiation, as well as the substantial decline in crowning index estimated in this study.

Finally, the only deciduous species at high elevation, aspen, formed a very small fraction of crown biomass (< 3%) at any site or date. In part this is due to differences in tree architecture, with aspen bearing relatively fewer and lighter-mass leaves high in the canopy compared to the longer crowns and denser leaves of conifers. The absence of fire after 1879 probably reduced aspen regeneration but a greater impact may have been selective herbivory by deer during the early twentieth century (Adams 1925, Rasmussen 1941, Mitchell and Freeman 1993). Aspen regeneration occurred on the study sites even in the absence of fire but a large demographic gap was observed corresponding with the period of high deer populations (Fulé et al. 2002, in press). If aspen comprised a relatively higher proportion of contemporary forest fuels, crown fire hazard would have been reduced (lower crown bulk density and high crown base height) and surface fire behavior would have been characterized by reduced rate of spread and intensity (Hély et al. 2000).

Management Implications

National Park management in the U.S.A. is oriented toward restoration of keystone natural processes, such as fire, as an integral part of conserving natural ecosystems (e.g., Parsons and Van Wagtendonk 1996, Stephenson 1999). Grand Canyon National Park fire management policy is to restore the natural process of fire disturbance "to effectively manage wildland fire and provide for the protection of life, property, and cultural resources, while ensuring the perpetuation of park ecosystems and natural resources. The restoration of natural fire regimes to park ecosystems is an important objective in managing the natural resources of GRCA" (Fire Management Plan 1992). The concept of the range of natural (or historical) variability has been used to characterize the variability of desired future conditions (Landres et al. 1999, Moore et al. 1999, Stephenson 1999, Swetnam et al. 1999).

The increasingly dense and connected crown fuels of the North Rim have been capable of supporting broad-scale crown fire for several decades, as evidenced by the severe fires of recent years such as the NWIII fire (1993) and Outlet fire (2000). But the differences in fire ecology and historical variability across the elevational gradient should be reflected in distinct management goals and actions for the different forest types.

Canopy fuels have increased substantially since 1880 in the ponderosa pine-dominated sites at low elevation. In contrast to the higher elevation forests, there is no evidence of historic crown fire at scales greater than 1-2 ha (Fulé et al. 2003). Managers could choose to encourage wildland fires to burn for resource benefits, supplemented by management-ignited fires, to maintain or restore a frequent surface regime. The data in the present study suggest that even if burns resulted in substantial mortality (e.g., 40-50% reduction in crown biomass), the resulting forest conditions would still be within the range of historical conditions. Our western ponderosa sites were rare areas even within the park because they had not undergone complete fire exclusion. But there is now substantial evidence from the park's experience with fire use and prescribed fire to show that broad-scale burning can be reintroduced safely in fire-excluded ponderosa pine sites such as the Walhalla Plateau (Kaufmann and Covington 2001), Galahad Point (Tower fire, 2001, S. Powers unpublished data), and Point Sublime, Widforss Point, and other areas (K. Kerr, personal communication, 2000).

In a study conducted in high-elevation forests adjacent to the Little Park East study site, White and VanKat (1993) suggested that fuels had become more homogeneous over the past century, presenting a threat of larger crown fires. The results of the present study support this assessment.

However, Fulé et al. (in press) compared the severity of burning in the 5,260-ha Outlet fire (Bertolette and Spotskey 2001) with the reconstructed forest structure at high elevation in 1879, arguing that the effects of the modern wildfire might not have been outside the range of natural variability in fire behavior and effects. At high elevations, therefore, management planning could incorporate recognition of the increase in fire hazard simultaneously with the perspective that severe fires have an ecological role in the high-elevation fire types. If this ecological role were explicitly addressed in management plans and environmental documentation, then at least some high-elevation crown fires might be managed more appropriately as natural disturbances rather than being treated as catastrophes that require costly and often futile wildfire suppression efforts.

Intermediate-elevation mixed conifer forests present the greatest management challenge. These forests had the highest historical and contemporary crown bulk density (Table 5) but had a regime of frequent surface fires before 1880 (Fulé et al. 2003). Trees in only 20% (3 of 15) mixed conifer plots in the LP study sites were initiated by stand-replacing fire (Fulé et al. in press). Previous reviews of hazardous fuels highlighted the difficulty of prescribed burning in heavy mixed conifer fuels and suggested the use of fuel treatments such as tree thinning to help reduce fuels (Davis 1981, Nichols et al. 1994). An ongoing experiment that includes thinning treatments was designed with a mixed conifer replicate, located in the SR study site (Grand Canyon National Park 2002). Fuel treatments could be strategically located to try to intercept crown fires (Finney 2000). Fuelbreaks would also be valuable for circumscribing secure burn blocks, widening the window for prescribed burning and reducing costs. Finally, relatively severe burning was suggested by Miller and Urban (1999) as a tool to rapidly restore historical conditions by preferentially killing fire-susceptible tree species and smaller trees. Severe prescribed burns would not be simple to execute but there is evidence from the 1993 NWIII fire, adjacent to the SR study site, that forest composition and density were reduced to historical levels while maintaining a native herbaceous understory (Fulé et al. in review, Huisinga et al. in review).

The changes in fuel mass, composition, and spatial arrangement across the North Rim landscape are large in both relative and absolute terms. The rise in canopy fuels is consistent with the growing scale of crown fires in the Southwest and similar dry forests across the West. In the lower-elevation forests, large-scale stand-replacing fires appear to be a novel disturbance regime with uncertain successional outcomes (Mast and Savage 2003). Even though crown fire plays an important ecological role in many of the higher-elevation forest types, it may be argued that the high fire susceptibility across the contemporary landscape has also crossed a threshold to represent a novel condition (Romme et al. 1998). New environments might emerge from broad-scale crown burning instead of the highly mixed pattern of burn severity that prevailed on the North Rim before 1880 (Fulé et al. in press). This seems especially likely in the high-elevation forest types of the Southwest because they are commonly restricted to small and isolated "sky island" habitats, with limited opportunity for re-introduction of propagules. In addition to the ecological effects, the consequences of unusually large crown fires on the North Rim have management, economic, and social implications that should be considered. Such implications are beyond the scope of this study, but a more comprehensive analysis may find merit in accepting certain costs or risks (increased funding for fire management, higher risk of escaped fires, more smoke) in exchange for a more aggressive approach to reducing landscape fuels and interrupting fuel continuity.

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Table 1. Tree species found on sampling plots at Grand Canyon study sites.

Species	Common Name	Code
Abies lasiocarpa (Hook.) Nutt.	Subalpine fir	ABLA
Abies concolor (Gordon & Glendinning) Hoopes.	White fir	ABCO
Picea engelmannii Parry ex Engelm.	Engelmann spruce	PIEN
Picea pungens Engelm.	Blue spruce	Combined with PIEN
Pinus ponderosa var. *scopulorum* P. & C. Lawson	Ponderosa pine	PIPO
Populus tremuloides Michx.	Quaking aspen	POTR
Pseudotsuga menziesii (Mirb.) Franco var. *glauca* (Beissn.) Franco	Rocky Mountain Douglas-fir	PSME
Quercus gambelii Nutt.	Gambel oak	QUGA
Robinia neomexicana Gray	New Mexican locust	RONE

Table 2. Hierarchical vegetation type classification scheme. Species codes are based on the GEnus and SPecies (e.g., *Abies concolor* = ABCO)

mixed conifer	abco/pien
	abco/pipo
	abco/potr
	abco/psme
	psme/pien
	psme/pipo
pipo	pipo
	pipo/abco
	pipo/pied
	pipo/potr
	pipo/psme
	pipo/quga
	pipo/rone
potr	potr/abco
	potr/abla
	potr/pien
	potr/pipo
quga	quga/pipo
spruce-fir	abla/pien
	abla/pipo
	pien
	pien/abco
	pien/abla
	pien/pipo
	pien/potr
burn	burn
grass-shrub	grass-shrub
meadow	meadow
Mohave desert scrub	Mohave desert scrub

Table 3. Forest canopy cover classes and non-forest categories in classified Landsat imagery.

Category	Class
Forest Canopy	11-25%
Forest Canopy	26-40%
Forest Canopy	41-70%
Forest Canopy	71-100%
Non-Forest	Below Rim Vegetation
Non-Forest	Burn
Non-Forest	Meadow
Non-Forest	Grass/Shrub

Table 4. Fuel moisture, wind, and temperature for the Bright Angel weather station (North Rim, Grand Canyon National Park, elevation 2560 m), 1970-2001. The 90th and 97th percentiles are shown for the month of June (7-day analysis periods), historically the month with the most severe fire weather. The 97th percentile conditions were used for fire behavior analysis. "1 H" refers to 1-hour timelag fuels, "10 H" to 10-hour timelag, etc. (Anderson 1982). Foliar moisture content was set to 100% (Agee et al. 2002).

Variable	June 90th percentile	June 97th percentile
1 H moisture (%)	1.7	1.4
10 H moisture (%)	2.8	1.8
100 H moisture (%)	3.2	3.2
Wind speed (km/h)	25.5	31
Temperature (°C)	30	32.2

Table 5. Crown bulk density values (kg/m^3) and (standard errors) summarized by study site (elevational gradient) and by vegetation type. Total change (%) from 1880 to 2040 is summarized in the final column.

	1880	1920	1960	2000	2040	% Change
Study Site (Elevation)						
PP (2,296 m)	.022 (.0018)	.033 (.0022)	.041 (.0025)	.046 (.0028)	.049 (.0030)	122%
FP (2,338 m)	.042 (.0065)	.054 (.0074)	.062 (.0080)	.068 (.0083)	.072 (.0086)	71%
RP (2,320 m)	.020 (.0023)	.032 (.0029)	.040 (.0034)	.045 (.0038)	.049 (.0040)	145%
SR (2,482 m)	.061 (.0057)	.074 (.0053)	.078 (.0040)	.075 (.0042)	.082 (.0058)	34%
LPW (2,650 m)	.021 (.0029)	.035 (.0038)	.050 (.0044)	.061 (.0043)	.079 (.0074)	276%
LPE (2,724 m)	.023 (.0032)	.038 (.0044)	.059 (.0058)	.078 (.0088)	.090 (.0102)	291%
Vegetation Type						
Aspen	.027 (.0060)	.037 (.0057)	.051 (.0061)	.062 (.0063)	.086 (.0142)	219%
Mixed Conifer	.041 (.0035)	.060 (.0042)	.071 (.0044)	.080 (.0053)	.094 (.0084)	129%
Ponderosa Pine	.032 (.0028)	.044 (.0029)	.051 (.0027)	.054 (.0026)	.057 (.0028)	78%
Spruce-fir	.017 (.0026)	.031 (.0037)	.052 (.0058)	.072 (.0087)	.080 (.0077)	371%

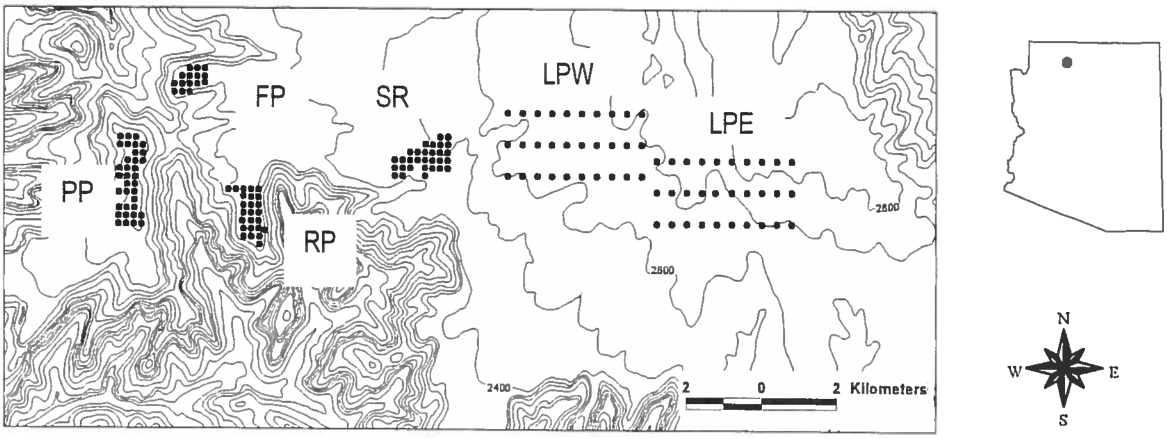

Figure 1. Study sites on the North Rim of Grand Canyon National Park, Arizona. From west to east, sites are Powell Plateau (PP), Fire Point (FP), Rainbow Plateau (RP), Swamp Ridge (SR), Little Park West (LPW) and Little Park East (LPE). Forest structure and fire regimes of the first four sites were described by Fulé et al. (2002, 2003). The Little Park area is described in Fulé et al. (in press).

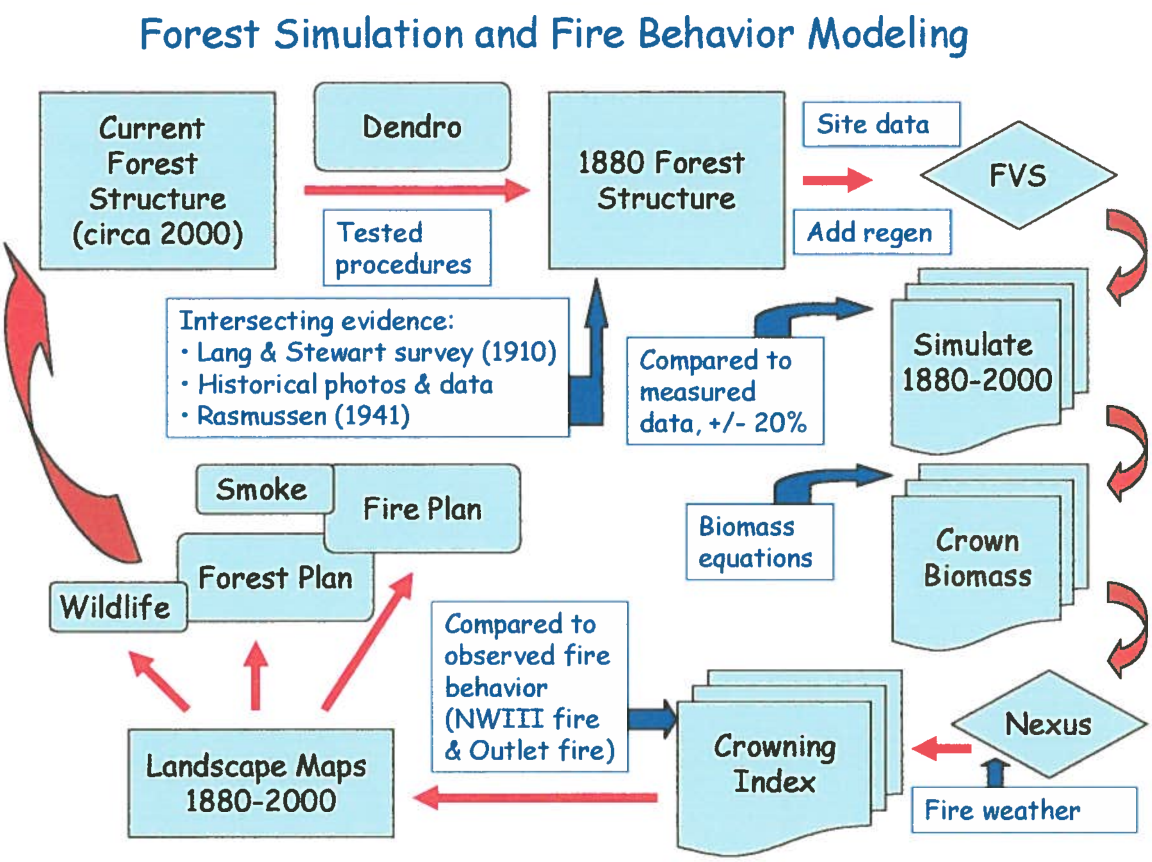

Figure 2. Schematic of forest data and modeling process, beginning with data collection on contemporary forest structure (upper left), dendroecological reconstruction of forest structure in 1880, simulation modeling with FVS (Forest Vegetation Simulator, upper right), crown fuel modeling, fire behavior modeling with Nexus (lower right), and development of fire hazard maps (lower left) that provide information for forest and fire planning. In turn, management actions based on these plans affect forests on the Grand Canyon landscape.

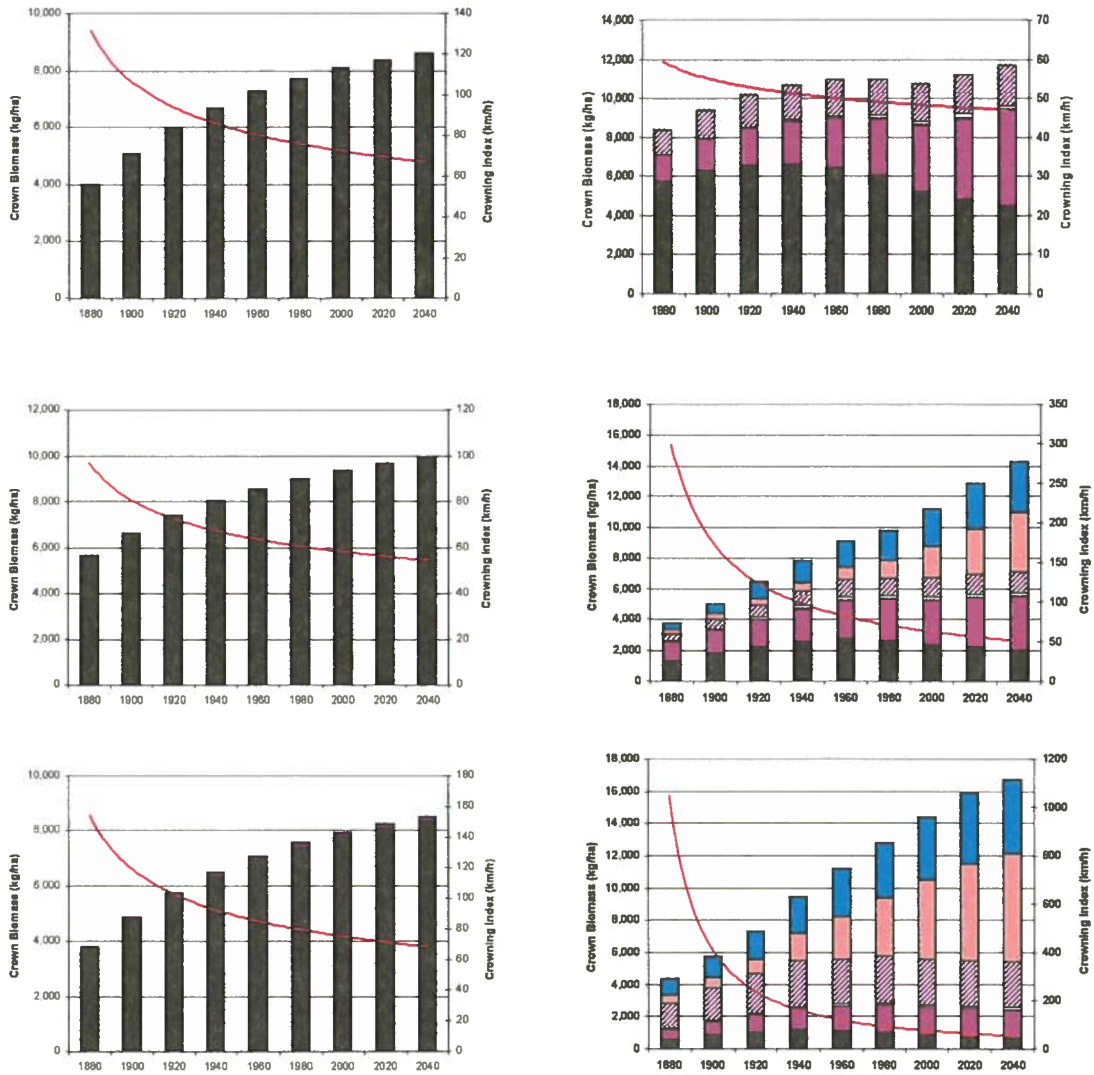

Figure 3. Changes in biomass of foliage and fine twigs (kg/ha) and crowning index (km/h), 1880-2000. Left column: PP (top), FP (middle), RP (bottom). Ponderosa pine made up >99% of crown biomass at these sites. Right column: SR (top), LPW (middle), LPE (bottom). Species in each column are (listed lower to upper): PIPO, ABCO, POTR, PSME, ABLA, PIEN.

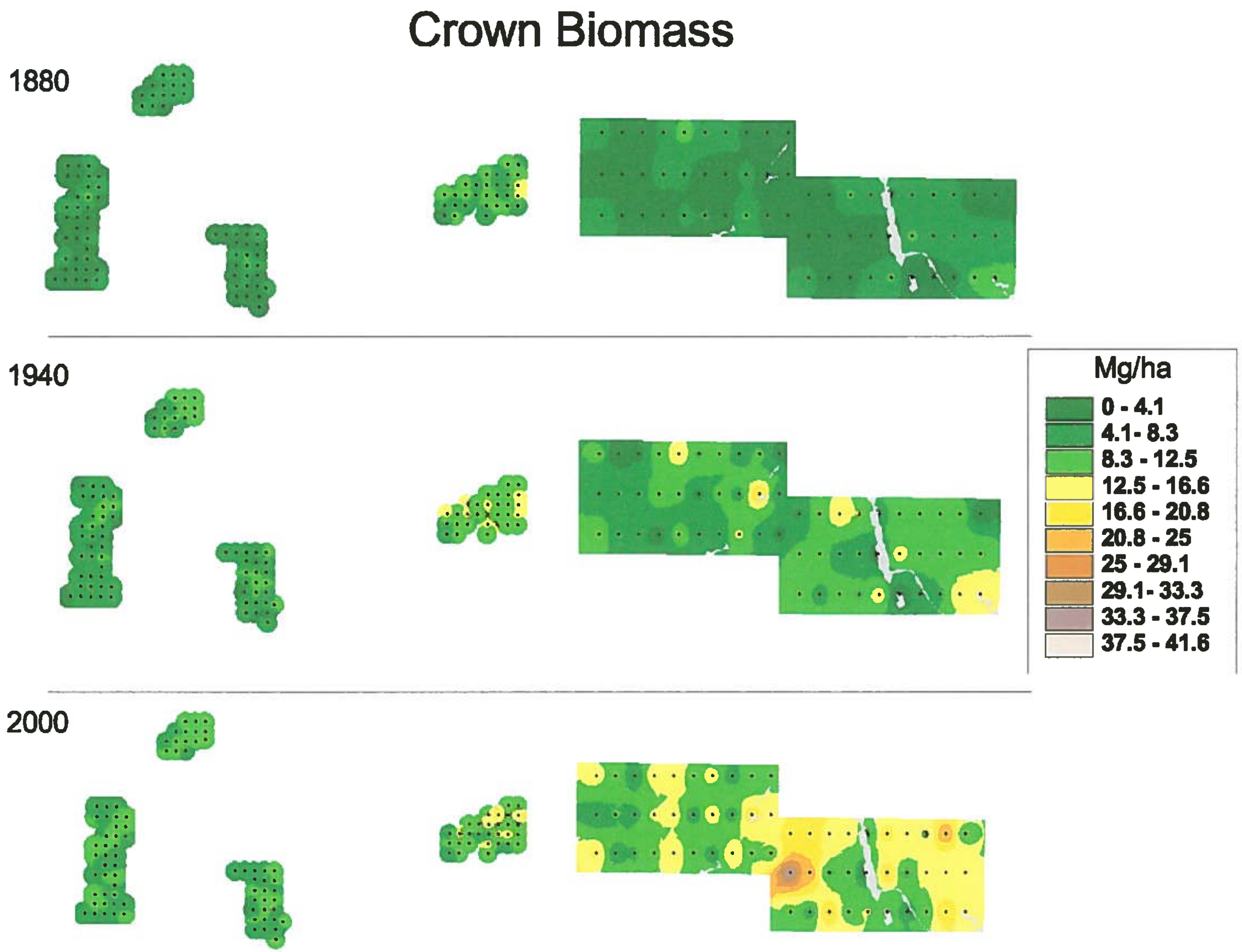

Figure 4. Crown biomass values interpolated across the study site landscapes. See Figure 1 for study site locations.

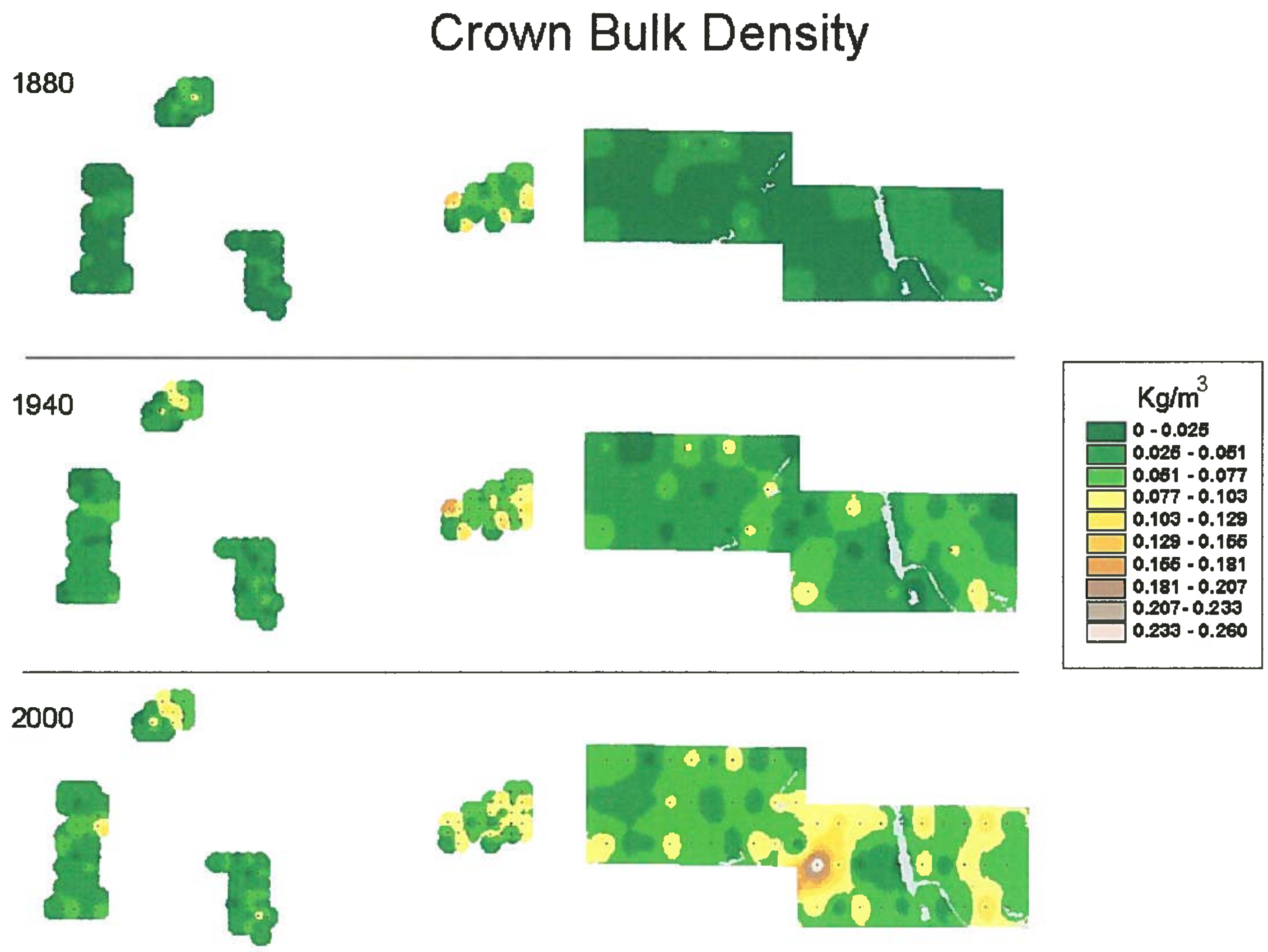

Figure 5. Crown bulk density values interpolated across the study site landscapes. See Figure 1 for study site locations.

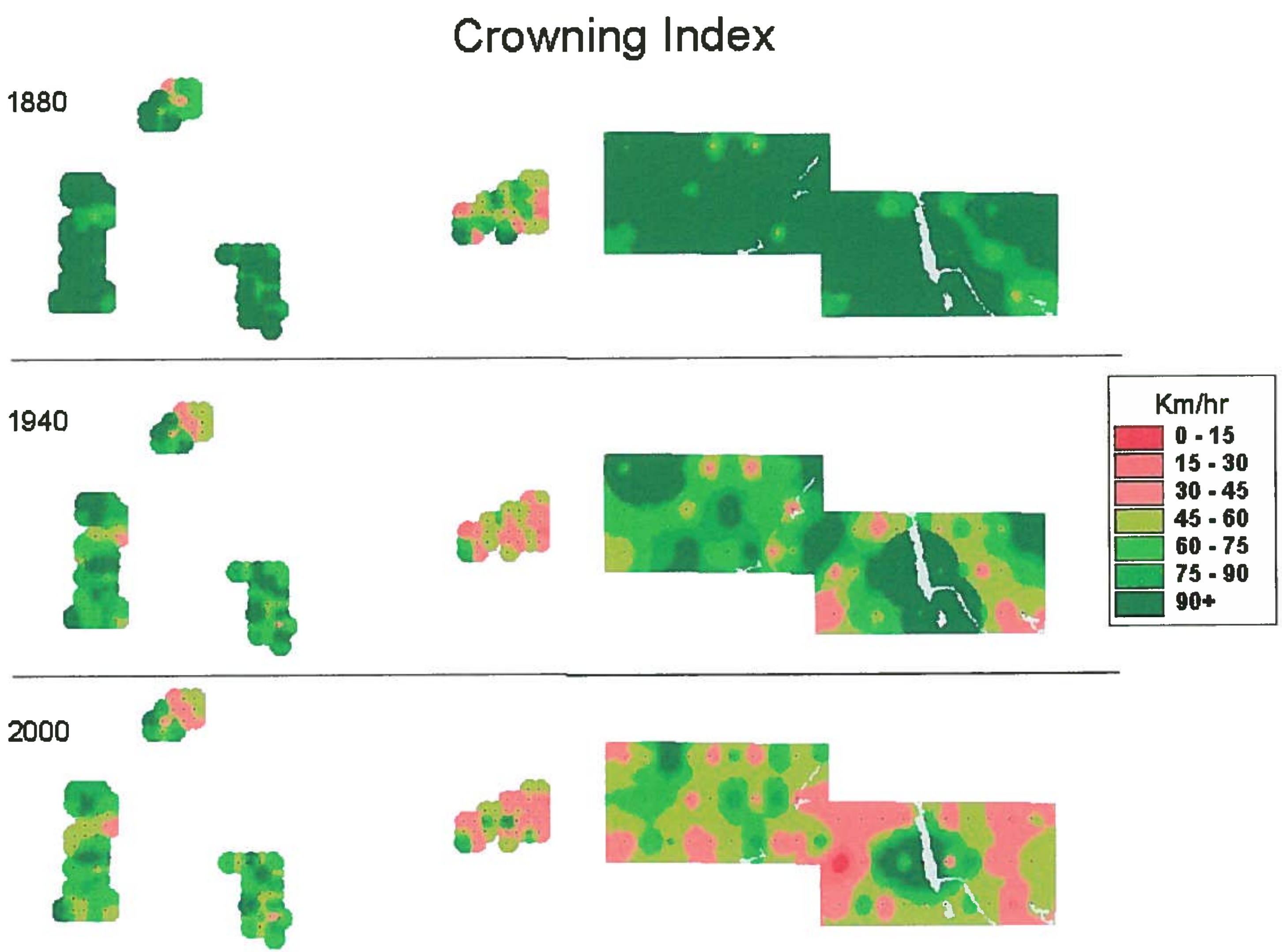

Figure 6. Values of crowning index, the windspeed required to sustain active crown fire, were estimated with the Nexus model and interpolated across the study site landscapes. See Figure 1 for study site locations.

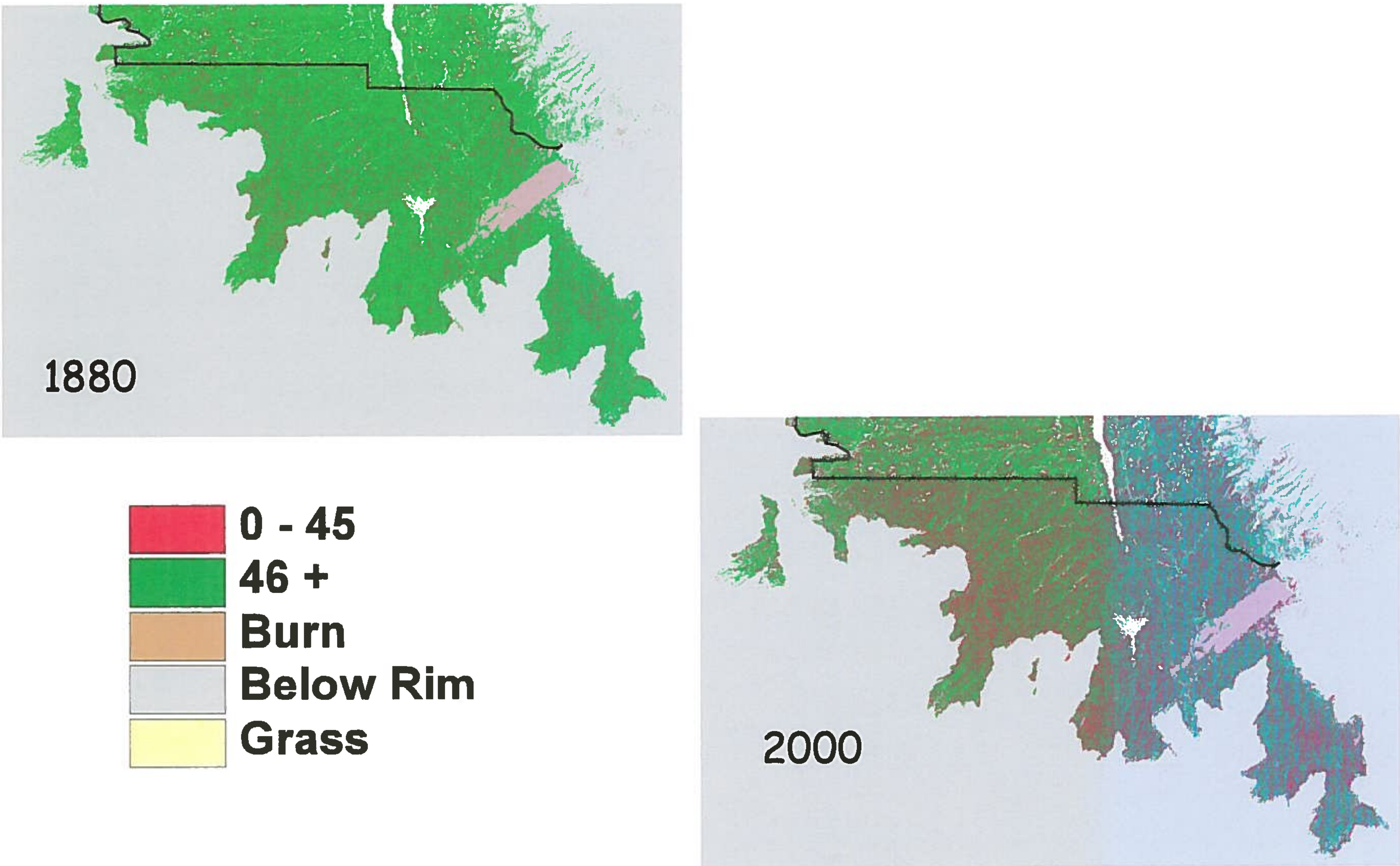

Figure 7. Comparison of crowning index values, km/hr, on North Rim forests in 1880 (relatively resistant to crown fire) and 2040 (relatively susceptible to crown fire). The large area classified as "burn" is the Outlet fire, which burned in May, 2000.

Figure 8. Older ponderosa pine overstory is being replaced by young white fir trees in this scene from the intermediate-elevation SR study.