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Changes in canopy fuels and potential fire behavior 1880–2040: Grand Canyon, Arizona

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Abstract

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 (canopy biomass, canopy 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. Canopy 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 canopy biomass in 1880 but had the lowest increase through 2040 (39%). Canopy bulk density increased roughly in parallel with canopy biomass; however, density values were considered less accurate in non-contemporary dates because they were based on assumptions about canopy 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 23–80% over the modeled period. Canopy fuel and CI values were mapped across the entire North Rim landscape. At a threshold windspeed of 45 km/h, 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. If managers choose to intervene to reduce canopy fuel mass and continuity, actions should be guided by the distinct ecological attributes of the different forest types.

Keywords: Kaibab Plateau; *Picea*; *Pinus*; *Abies*; *Pseudotsuga*; *Populus*; Crown fire; Fire hazard

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1. 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

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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 are 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 (Flannigan et al., 2000).

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 (Keane and Finney, 2003). 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, 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. The accuracy of long-term simulation may be easier to assess when dynamics are modeled retrospectively as well as into the future (Fuhlendorf et al., 1996). Stochastic simulation approaches, such as those used by He and Mladenoff (1999) in Wisconsin or Miller and Urban (2000) 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). About 90% of these forest types 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., 2003b). 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) and fuel accumulations (White and Vankat, 1993; Fulé et al., 2003b).

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.

2. Methods

2.1. Study area

The study area was a 6-km × 18-km transect on the Kaibab Plateau, the North Rim of the Grand Canyon, in northern Arizona. A series of study sites was arranged along an elevational gradient from Powell Plateau (approximately 2300 m elevation) up to the highest elevation in Grand Canyon National Park (2794 m in the Little Park East study site), comprising a total of 5345 ha. The forest structures and fire regimes at the study sites were described in detail by Fulé et al. (2002, 2003a, b).

Average annual precipitation at the North Rim ranger station (elevation 2542 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). Soils were derived from Kaibab limestone; soil textures ranged from coarse to fine loams (A. Dewall, National Resource Conservation Service, personal communication, 2002).

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 2500 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 and our observation of trees at the study site that had characteristics intermediate between the two species.

2.2. Image classification

Image classification was described in detail by Crouse and Fulé (2003) and is summarized here. We used a Landsat 7 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). Image classification and analysis was done using Erdas Imagine (Leica Geosystems AG, Switzerland). Field plot data (166 plots, described below) collected from 1997 to 2002 were used as training sites for classification. The classification scheme was developed within the National Vegetation Classification Standards (NVCS) framework (USGS, 2000). 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. Forest types were classified into four types: (1) aspen, (2) mixed conifer (sites where importance values were dominated by white fir or Douglas-fir), (3) ponderosa pine, and (4) spruce-fir. The two non-forest types were (1) grass, and (2) burn. The "burn" category was the Outlet Fire of 2000, a 5260-ha stand-replacing fire.

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

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

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 spruce do not occur at the lower elevations of the study area. The model "flagged" this vegetation type if it occurred below 2250 m. These pixels were then edited to reflect the correct vegetation type. Verification against an independent data set of 86 fire monitoring plots showed an overall classification accuracy of 74.7%.

2.3. *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 (Fig. 1). Sampling plots were 0.1 ha (20 m × 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 with 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 and 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 and >20 cm for aspen trees (Fulé et al., 2002). "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

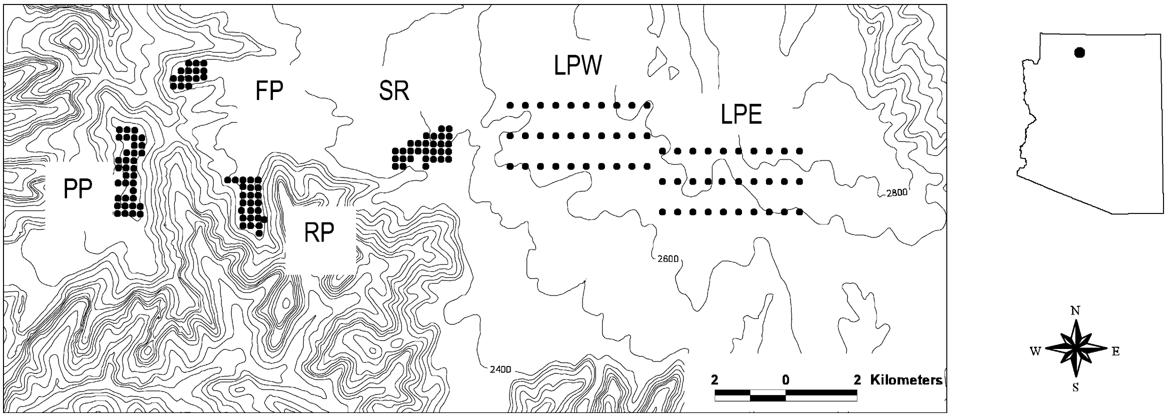

Fig. 1. Study sites on the North Rim of Grand Canyon National Park, Arizona. From west to east, sites are Powell Plateau (PP), Fire Point (FP), Rainbow Plateau (RP), Swamp Ridge (SR), Little Park West (LPW) and Little Park East (LPE). Forest structure and fire regimes of the first four sites were described by Fulé et al. (2002, 2003a). The Little Park area is described in Fulé et al. (2003b). The dots at each study site represent the locations of permanent plots. Each plot is 20 m × 50 m in size.

(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., 2002, 2003a, b), 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, 2003b) 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 ($r^2 = 0.45$–0.90) 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).

2.4. Modeling and canopy fuels

A schematic diagram of the modeling process is shown in Fig. 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. FVS is a statistical model of forest growth, originally developed in the northern Rocky Mountains and expanded to serve as a nationwide platform for U.S. Forest Service simulation. The specific growth and yield model for the Central Rockies variant was largely developed in northern Arizona, close to the site of the present study. We chose this model because it is a highly precise and site-specific statistical model for short-term simulation of forest change (Edminster et al., 1991), realistically

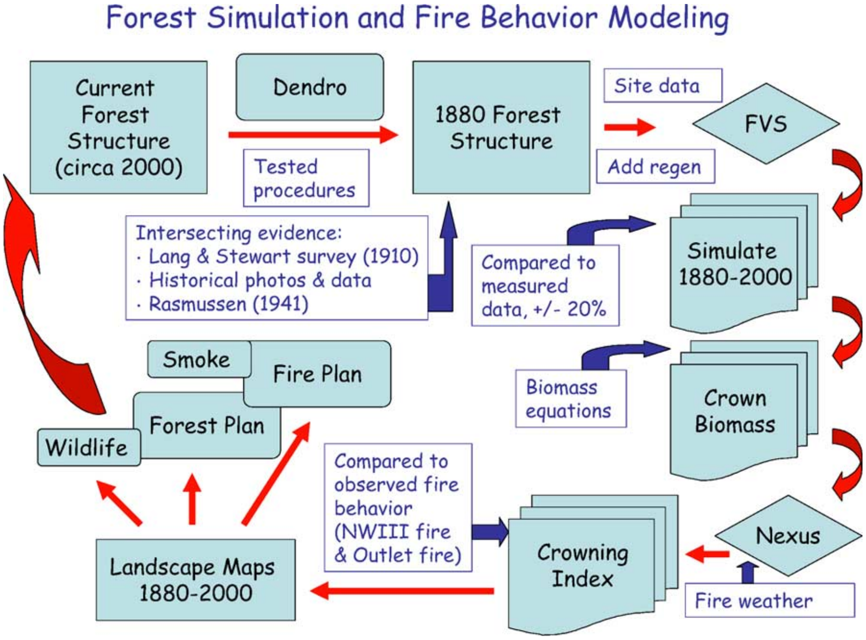

Fig. 2. Schematic of forest data and modeling process, beginning with data collection on contemporary forest structure (upper left), dendroecological reconstruction of forest structure in 1880, simulation modeling with FVS (upper right), crown fuel modeling, fire behavior modeling with Nexus (lower right), 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.

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 and 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 per hectare) and basal area.

Tree lists were exported from FVS for each plot and decade. Canopy 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., 2001a), 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 Korzuhkin (1997) were used for aspen, subalpine fir, and Engelmann spruce.

Canopy 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). We used maximum tree height per plot as a proxy for “dominant height” (Research Working Group #2, 1999), the average height of dominant and co-dominant trees. Even though the tallest tree on each plot was used, the measurement is not equivalent to “predominant height” (the tallest trees in the stand; Research Working Group #2, 1999), because the tallest trees in the stand were not evenly captured on the plots. Therefore, the top of canopy measured by our method fell in the upper portion of the tree height distribution but not at the maximum. For example, the average maximum height of PIPO trees on a per-plot basis was 28.2 m, a value equivalent to the 85th percentile of the distribution of all PIPO tree heights. The bottom of the canopy can also be measured in varying ways (Scott and Reinhardt, 2001). In previous studies where we explored stand vulnerability to torching, we used the lowest quintile value (Fulé et al., 2001a). In the present study, however, the uncertainty of estimating long-term characteristics of crown base height were high so we chose a simple average and did not attempt to model torching. 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 was used for the plot value. Canopy bulk density was calculated as canopy biomass divided by canopy volume.

Fire behavior was modeled with the Nexus Fire Behavior and Hazard Assessment System (Scott and Reinhardt, 2001). Nexus integrates a deterministic semi-empirical model of surface fire behavior (Rothermel, 1972) with models of crown fire transition (Van Wagner, 1977) and crown fire spread (Rothermel, 1991). 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 90th and 97th percentiles of low fuel moisture 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), a statistical software package designed to analyze distributions of fire-related weather variables. Values were calculated for June, historically the month with the most severe fire weather (Table 2).

Table 2
Fuel moisture and temperature for the Bright Angel weather station (North Rim, Grand Canyon National Park, elevation 2560 m), 1970–2001

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
Temperature (°C)	30	32.2

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-h timelag fuels, “10 H” to 10-h timelag, etc. (Anderson, 1982). Foliar moisture content was set to 100% (Agee et al., 2002).

Fire behavior fuel models (Anderson, 1982) used were 9 (sites below 2500 m elevation) and 10 (sites above 2500 m elevation). Fuel models are mathematical summaries of key input variables for the surface fire behavior model in Nexus, such as bulk density and surface-area-to-volume ratios.

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 canopy biomass, canopy 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.

3. Results

3.1. Changes in canopy fuels, 1880–2040

Canopy biomass increased at all study sites between 1880 and 2040, but the changes were most dramatic at the high-elevation LPW and LPE sites (Fig. 3, see Fig. 1 for site names). At the three lower-elevation sites, PP, FP, and RP, canopy biomass ranged from 3800 to 5600 kg/ha in 1880 (mean 4473 kg/ha). By 2040, canopy biomass ranged from 8510 to 9940 kg/ha (mean 9022 kg/ha), an average increase of 122%. The higher-elevation sites LPW and LPE also had low canopy biomass values in 1880, 3800 and 4400 kg/ha, respectively. But by 2040, LPW canopy biomass increased to 14,270 kg/ha and LPE canopy biomass rose to 16,800 kg/ha, an average increase of 279%. The intermediate-elevation site, SR, had the highest 1880 canopy biomass (8370 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 canopy biomass (>99%) consisted of ponderosa pine throughout the simulated 160-year period. But the proportion of ponderosa pine canopy biomass declined at the higher sites. In the intermediate-elevation SR site, at approximately 2500 m elevation, the canopy 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 2650 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 canopy biomass by 2040, rising from only 19% in 1880. The highest site, LPE, approximately 2700 m elevation, started with 86% non-pine canopy biomass in 1880 and increased to 96%. Engelmann spruce and subalpine fir alone made up 36% of total canopy biomass in 1880 and 67% in 2040.

The changes in canopy biomass were interpolated across the landscape of the study sites at 60-year intervals (1880, 1940, and 2000) (Fig. 4). In 1880, the entire landscape had relatively low canopy biomass values, with the highest values at mid-elevation (SR). By 1940, however, higher canopy 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.

Canopy bulk density values closely paralleled canopy biomass values (Fig. 5), as expected, but with minor variation due to plot-to-plot differences in canopy volume. The increases in canopy bulk density ranged from 34% (SR) to 291% (LPE) (Table 3). Summarized by vegetation type rather than by study site, the spruce-fir type had the highest increase

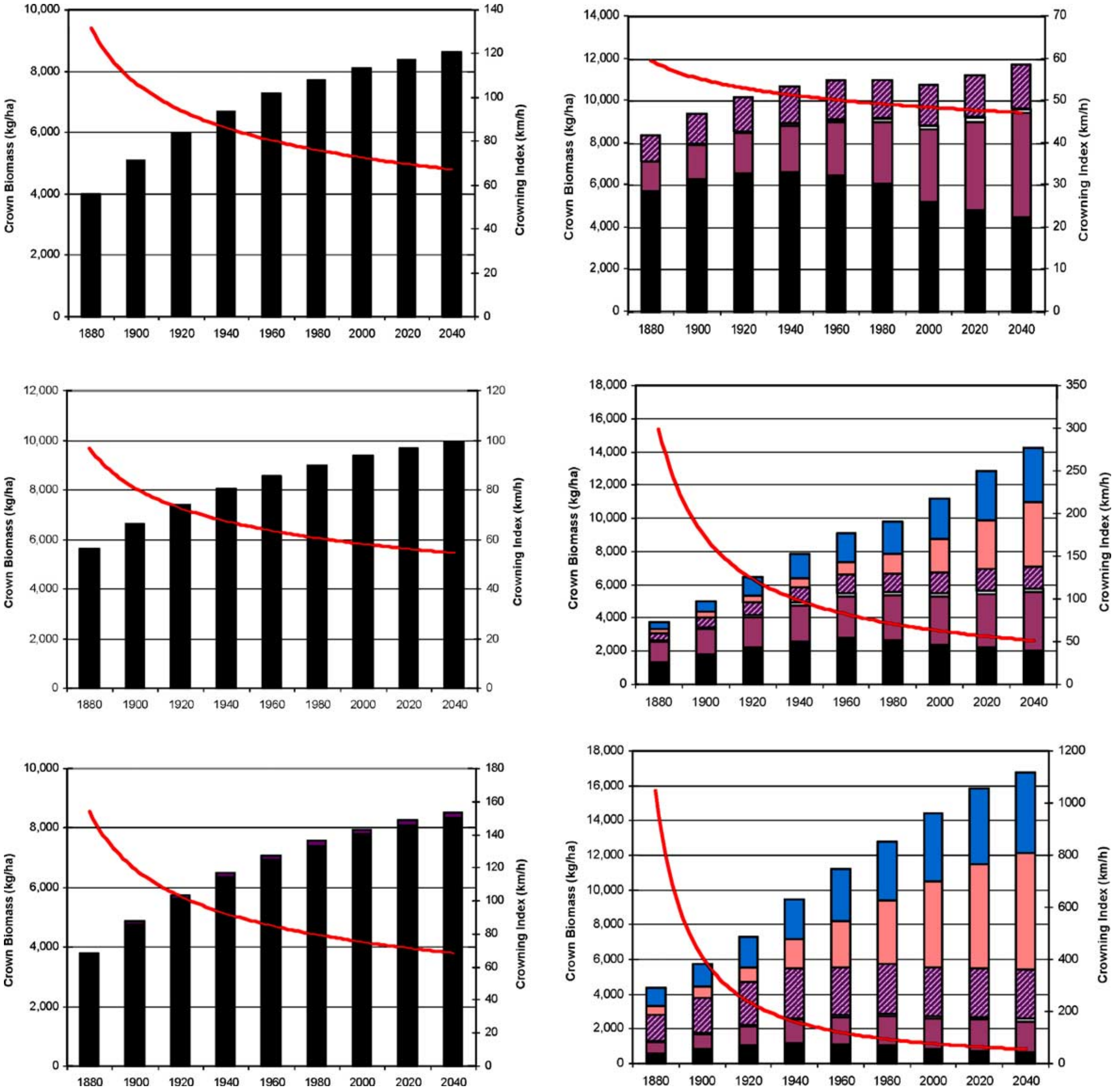

Fig. 3. Changes in biomass of foliage and fine twigs (kg/ha) and crowning index (km/h), 1880–2000. Left column: PP (top), FP (middle), RP (bottom). Ponderosa pine made up >99% of canopy biomass at these sites. Right column: SR (top), LPW (middle), LPE (bottom). Species in each column are (listed lower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

(371%), while ponderosa pine had the lowest (78%). In 1880, the absolute values of canopy bulk density were below 0.05 kg/m^3 in every vegetation type and every study site except SR. The canopy bulk density value of 0.05 kg/m^3 is not necessarily a threshold for crown fire behavior (see comments below on alternative methods of canopy bulk density calculation), but values of 0.05 kg/m^3 or higher have been associated with passive or active crown fire in other studies using Nexus in southwestern forests (Fulé et al., 2001a, 2001b). By 1960, all vegetation types and all study sites except the lowest elevation sites (PP and RP) had canopy bulk density >0.05 kg/m^3. By 2040, four of the six study sites and three of the four vegetation types had canopy bulk density >0.07 kg/m^3 (Table 3).

Fig. 4. Canopy biomass values interpolated across the study site landscapes. See Fig. 1 for study site locations.

3.2. 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 Fig. 6. The lower-elevation sites dominated by ponderosa pine, PP, FP, and RP, had CI values ranging from 96 to

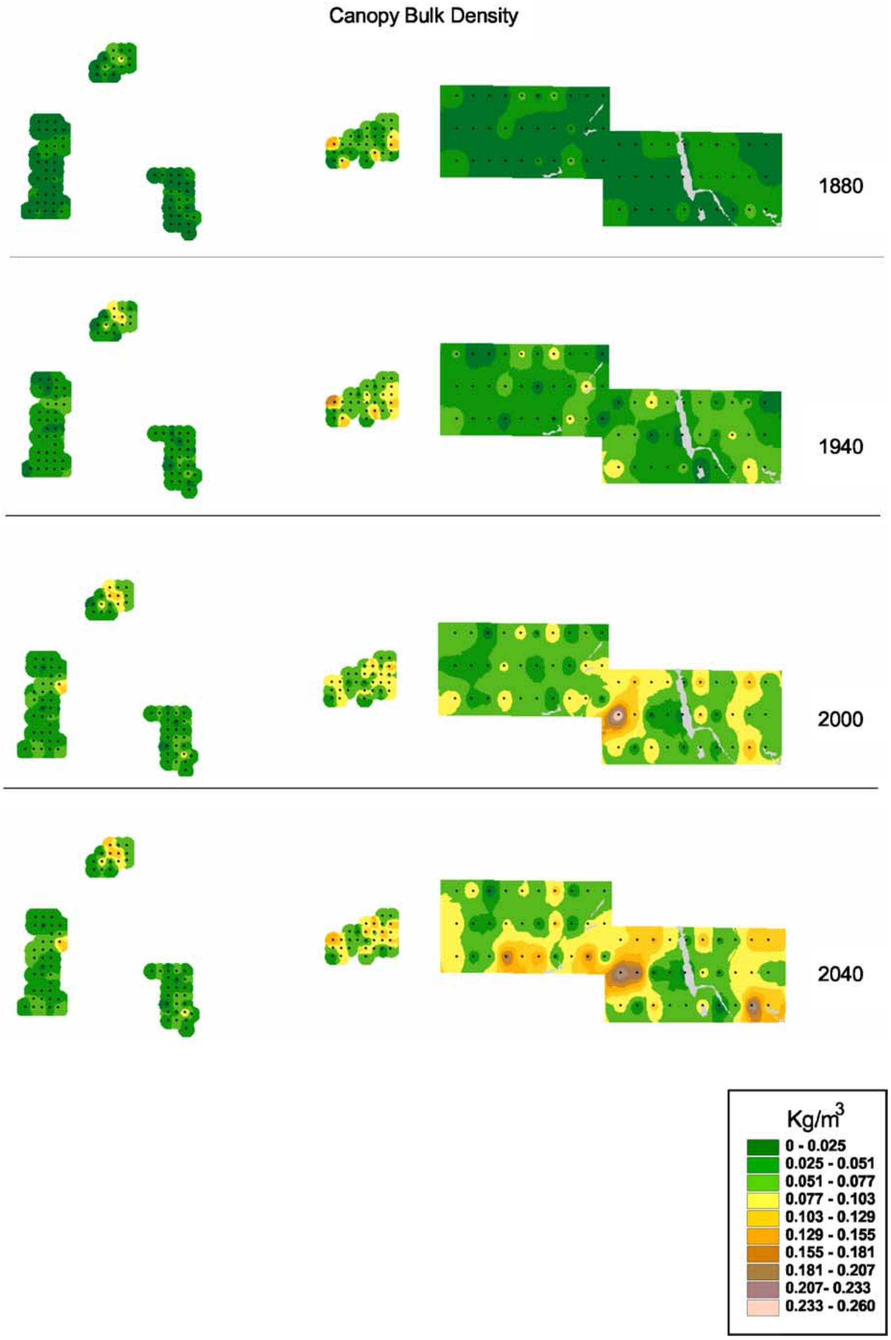

Fig. 5. Canopy bulk density values interpolated across the study site landscapes. See Fig. 1 for study site locations.

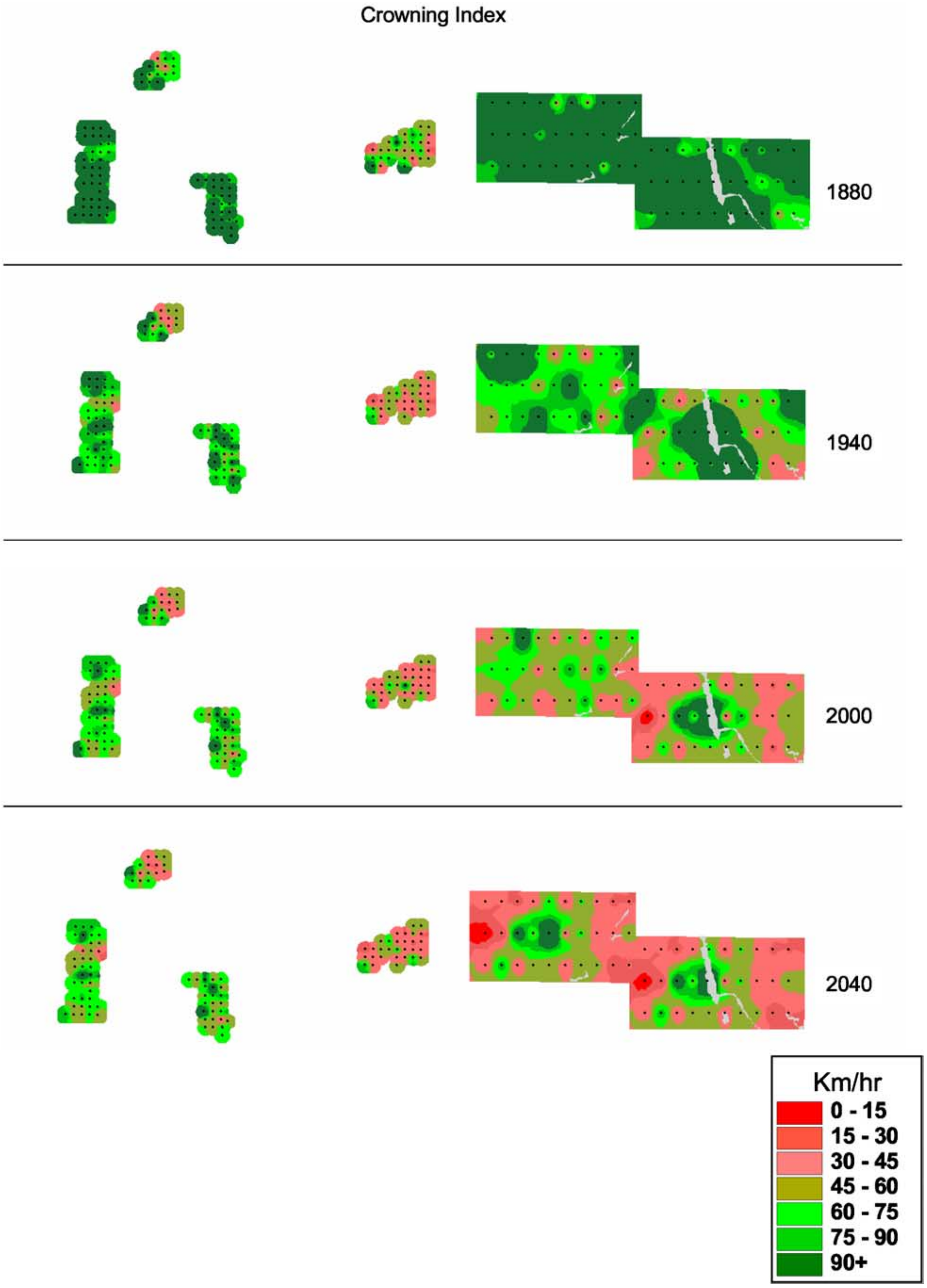

Fig. 6. Values of crowning index, the windspeed required to sustain active crown fire, were estimated with the Nexus model and interpolated across the study site landscapes. See Fig. 1 for study site locations.

Table 3
Canopy bulk density values (kg/m^3) and (standard errors) summarized by study site (elevational gradient) and by vegetation type

	1880	1920	1960	2000	2040	Change (%)
Study site (elevation)						
PP (2296 m)	0.022 (0.0018)	0.033 (0.0022)	0.041 (0.0025)	0.046 (0.0028)	0.049 (0.0030)	122
FP (2338 m)	0.042 (0.0065)	0.054 (0.0074)	0.062 (0.0080)	0.068 (0.0083)	0.072 (0.0086)	71
RP (2320 m)	0.020 (0.0023)	0.032 (0.0029)	0.040 (0.0034)	0.045 (0.0038)	0.049 (0.0040)	145
SR (2482 m)	0.061 (0.0057)	0.074 (0.0053)	0.078 (0.0040)	0.075 (0.0042)	0.082 (0.0058)	34
LPW (2650 m)	0.021 (0.0029)	0.035 (0.0038)	0.050 (0.0044)	0.061 (0.0043)	0.079 (0.0074)	276
LPE (2724 m)	0.023 (0.0032)	0.038 (0.0044)	0.059 (0.0058)	0.078 (0.0088)	0.090 (0.0102)	291
Vegetation type						
Aspen	0.027 (0.0060)	0.037 (0.0057)	0.051 (0.0061)	0.062 (0.0063)	0.086 (0.0142)	219
Mixed conifer	0.041 (0.0035)	0.060 (0.0042)	0.071 (0.0044)	0.080 (0.0053)	0.094 (0.0084)	129
Ponderosa pine	0.032 (0.0028)	0.044 (0.0029)	0.051 (0.0027)	0.054 (0.0026)	0.057 (0.0028)	78
Spruce-fir	0.017 (0.0026)	0.031 (0.0037)	0.052 (0.0058)	0.072 (0.0087)	0.080 (0.0077)	371

Total change (%) from 1880 to 2040 is summarized in the final column.

156 km/h (average 128 km/h) in 1880, declining to 55–71 km/h (average 65 km/h) in 2040, a decrease of 49%. CI values at the intermediate-elevation site, SR, dropped from 62 to 48 km/h 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/h, respectively. By 2040, the CI dropped to 54 km/h at LPW (80% decline) and to 52 km/h 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/h or less by 2000. Windspeeds of 45 km/h or more are common as gusts and occasionally as sustained winds in northern Arizona during severe fires (McCoy, 1996).

Modeled across the entire North Rim, 33% of the landscape was susceptible to crown fire at a windspeed threshold of 45 km/h in 2000, compared to less than 6% in 1880 (Fig. 7). 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

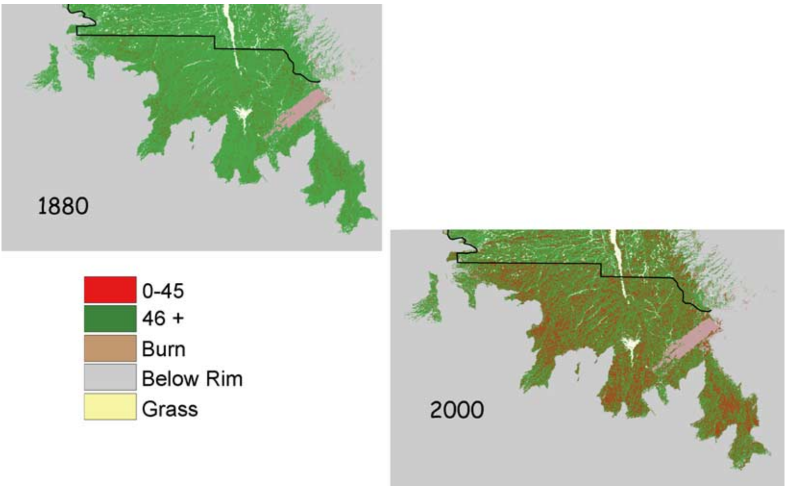

Fig. 7. Comparison of crowning index values, km/h, on North Rim forests in 1880 (relatively resistant to crown fire) and 2040 (relatively susceptible to crown fire). The large area classified as “burn” is the outlet fire, which burned in May, 2000.

at the center of the plateau were also susceptible, as well as linear spruce-fir stands located on north slopes of drainages.

4. Discussion

4.1. 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, 2003b) 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, similar to the historical distribution by species in multi-species sites (Fulé et al., 2002, 2003b). We expected vegetation simulation to be relatively accurate because we chose a local statistical model, the Central Rockies variant of the 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. The tradeoff for precision, however, was that the simulation model lacked process relationships to explore the potential ecological effects of changing climate and disturbance regimes (e.g., Peng and Apps, 1999). Therefore we limited future projections to the near term, the next 40 years.

Canopy biomass was estimated with published regression equations with high predictive capability ($r^2 = 0.8$–0.9) and local equations were selected whenever possible, indicating that canopy biomass changes over time should be relatively accurate. However, the calculation of canopy bulk density required assumptions about canopy volume. We calculated canopy 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 canopy 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. Most importantly, all living trees always grew substantially in height in FVS modeling, leading to modeled heights in 2000 that were higher than measured heights and continuous expansion of canopy volume over time. Since most plots were dominated by mature trees at the outset, this approach would have been problematic. Finally, since we used an independent set of diameter-based allometric biomass equations, rather than FVS-modeled crown fuel values, there is no inherent linkage between models of tree height and canopy biomass. Our use of contemporary canopy volume measurements has two implications. First, it meant that current canopy bulk density estimates (e.g., 1980–2020) were probably more accurate than distant estimates (e.g., 1880–1920). In particular, canopy bulk density was most likely underestimated in early decades in the regenerating stands at high elevation (sites LPW and LPE, Fig. 5). Although canopy biomass was very low on many of those plots due to severe fires, especially in 1879 (Fulé et al., 2003b), canopy volume would also have been lower due to the small stature of the young trees. Furthermore, the dates of stand-replacing fires varied widely and past fire severity was highly mixed (Fulé et al., 2003b), so taller surviving trees were commonly in close proximity to stand-replacing fire patches. 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). However, by basing our canopy volume estimates on measured rather than modeled values, the comparisons over time are relatively realistic and consistent.

The use of allometric canopy biomass equations produces estimates that are approximately 40–60% lower than the canopy 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 canopy 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 canopy 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 canopy 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 canopy 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 (Figs. 4–6) because sampling was done on regular grids, a strategy for minimizing interpolation error. Spatial accuracy was high at each mapped plot (within the ±15 m accuracy of hand-held global positioning system units) and unknown between plots. However, the same interpolation procedures were used for each map, so comparisons between maps are consistent.

4.2. 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 $\geq$2500 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, 2003a) 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., 2003a). 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 western sites remain 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 relatively low 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 canopy 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 (Fig. 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 canopy biomass (Fig. 3). The major increases in canopy biomass occurred in white fir (Fig. 3), subalpine fir, and spruce (Fig. 3), relatively fire-susceptible species.

Habitat changes associated with forest alteration include the decline 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., 2003b), 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 presence of numerous jackstrawed fallen logs in mixed conifer and spruce-fir stands (P.Z. Fulé, unpublished data), lead us to select fuel model 10 (“light logging slash,” Anderson, 1982) 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.

Aspen, the only deciduous species at high elevation, formed a very small fraction of canopy 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 to some extent 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, 2003b). If aspen comprised a relatively higher proportion of contemporary forest fuels, crown fire hazard would have been reduced (lower canopy 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).

4.3. Management implications

National Park management in the U.S. 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” (Grand Canyon National Park, 1992, p. 57).

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. 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 at low, mid, and high elevation. In the ponderosa pine-dominated sites at low elevation, canopy fuels have increased substantially since 1880 but remain lower than in the other forest types. 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 reduction in canopy biomass, up to 40–50%, the resulting forest conditions could still be within the range of historical conditions if older trees tended to survive. In a study conducted in high-elevation forests adjacent to the LPE 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, management planning could balance 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.

Mid-elevation mixed conifer forests present the greatest management challenge. These forests had the highest historical and contemporary canopy bulk density (Table 3) but had a regime of frequent surface fires before 1880 (Fulé et al., 2003a). Tree groups in only 20% (3 of 15) mixed conifer plots in the

LP study sites were initiated by stand-replacing fire (Fulé et al., 2003b). Alternatives include testing tree thinning treatments (Grand Canyon National Park, 2002), especially where strategically located to try to intercept crown fires (Finney, 2000), and using relatively severe burning as suggested by Miller and Urban (2000) 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 press).

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 western U.S. 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., 2003b). 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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