Effects of an Intense Prescribed Forest Fire: Is It Ecological Restoration?

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Abstract

Relatively intense burning has been suggested as a possible alternative to the restoration of pre-European settlement forest conditions and fire regime in mixed conifer forests, in contrast to thinning of trees and light prescribed burning. In 1993 a management-ignited fire in a dense, never-harvested forest in Grand Canyon National Park escaped prescription and burned with greater intensity and severity than anticipated. We sampled the burned site and an adjacent unburned site (270 ha each) 6 years after the fire to assess burn effects on tree structure (species composition, size and age distributions, regeneration, and snags), forest floor fuels, and coarse woody debris. Tree structure before fire-regime disruption (1879 CE) was reconstructed with dendroecological techniques. By 6 years after burn the fire reduced average tree density (331 trees/ha) and basal area (28.5 m^2/ha) to levels similar to pre-European reference conditions (approximately 246 trees/ha and 28.5 m^2/ha). Mortality was concentrated in fire-susceptible species, especially white fir, restoring dominance by fire-resistant ponderosa pine. Forest floor fuels were reduced, and regeneration by aspen and understory plants was vigorous. Densities of large snags and logs were high. However the fire also killed a high proportion of old-growth trees, especially aspen. Burning created more spatial variability in forest structure than was present before fire-regime disruption by killing many trees in some areas of the site but few in other areas. The intentional use of severe burning would be challenging to managers because of the increased risk of escaped fires, but the ecological outcome of this particular wildfire was not inconsistent with ecological restoration goals for this ecosystem type.

Key words: **aspen (*Populus tremuloides*), Grand Canyon, Kaibab Plateau, mixed conifer, North Rim, ponderosa pine (*Pinus ponderosa*).**

Introduction

Several different approaches have been suggested for restoring surface fire to western forests from which fire has been excluded for over a century. As described by Stephenson (1999) for Sequoia (*Sequoiadendron giganteum* [Lindl.] Buchholz) forests, process restoration advocates have sought to use fire as the primary tool to restore conditions similar to those of historical forests, while structural restoration advocates argued for deliberate thinning to restore near-historical forest structure before burning. Although the terms process and structural restoration are an oversimplification of a nuanced reality (Moore et al. 1999; Stephenson 1999), they are a useful framework for comparison. In a simulation study applying the forest gap model FACET in mixed conifer forest of the Sierra Nevada, Miller and Urban (2000) tested these two restoration approaches and added a third alternative: (1) simple reintroduction of surface fire at the historical frequency (process only); (2) tree thinning (structural manipulation) followed by reintroduction of fire; and (3) intense prescribed fires, killing a number of trees, followed by surface fire reintroduction. They concluded that all three options eventually restored forests within the range of natural variability (RNV) (Landres et al. 1999), although much more time was required under the first alternative. However Miller and Urban (2000) recognized that the simulation results were uncertain because the fire behavior model they used could not simulate canopy burning, leaving unresolved the practical question of whether either low- or high-intensity prescribed fires (alternatives 1 and 3) could actually be applied in fire-excluded forests without initiating a crownfire.

The debate over restoration approach is highly relevant in the 2,950,000 ha of ponderosa pine (species names are summarized in Table 1) and lower mixed conifer forests in the Southwest (Fiedler et al. 2002; O'Brien 2002). In New Mexico over 90% of these forests are considered at moderate or high risk for crownfire due to dense stand structure and accumulated fuels (Fiedler et al. 2002). Over two decades of experimentation with fire reintroduction (alternative 1) show that prescribed fire at frequent intervals can be conducted effectively, but established young trees are very resistant to thinning from surface fires (Peterson et al. 1994; Sackett et al. 1996). Hence the trajectory of forest change does not move toward RNV, even in a multi-century simulation with the FIRESUM ecological process model (Covington et al. 2001). In contrast, experiments with structural manipulation—tree thinning—followed by surface fire (alternative 2) have been shown to rapidly

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Table 1. Tree species found at the study sites.

Species	Common Name	Code
Abies lasiocarpa (Hook.) Nutt.	Subalpine fir	ABLA
Abies concolor (Gordon & Glendinning) Lindl.	White fir	ABCO
Picea engelmannii Parry ex Engelm.	Engelmann spruce	PIEN
Pinus ponderosa P. & C. Lawson	Ponderosa pine	PIPO
Populus tremuloides Michx.	Quaking aspen	POTR
Pseudotsuga menziesii (Mirb.)	Douglas fir	PSME
Robinia neomexicana Gray	New Mexican locust	RONE

Species codes are derived from the genus and species names.

restore many key structural and functional forest attributes (Scott 1998; Lynch et al. 2000; Fulé et al. 2001, 2002*a*). Long-term simulation indicated that these conditions could be maintained after the original manipulation simply by burning at historical intervals (Covington et al. 2001; Fiedler et al. 2002; Fulé et al. 2002*a*). However there are constraints to the thinning-and-burning approach, including the high cost of thinning (Lynch et al. 2000; Fulé et al. 2002*a*), soil damage associated with mechanical equipment and slash disposal, exotic species, deleterious effects on some wildlife species (Tiedemann et al. 2000; Germaine & Germaine 2002), management policies restricting tree cutting in places such as national parks and wilderness areas, and finally the sheer scale of forest area at risk. Thus it would be helpful to have quantitative information about other restoration options, such as Miller and Urban's (2000) alternative 3: structural change achieved by intense prescribed fire rather than thinning.

A simulated test of this alternative is beyond the capabilities of fire behavior and effects models, and Sackett et al. (1996) pointed out the practical difficulties of actually experimenting with intense burning. Therefore the best data with which to assess intense burning comes from an actual example. We selected a 1993 prescribed fire in a ponderosa pine/mixed conifer forest in Grand Canyon National Park that burned more intensely than anticipated. The Grand Canyon forest has never been harvested, and most of it has had over a century of fire exclusion (White & Vankat 1993; Wolf & Mast 1998), leading to substantial increases in forest density (Fulé et al. 2002*b*). The area is valuable as a study site because extensive analysis of past and present forest structure and fire regime has been carried out (Fulé et al. 2002*b*, 2003). We measured paired burned and unburned sites to test the hypothesis suggested by Miller and Urban (2000): intense burning would be expected to preferentially kill small and fire-susceptible trees and reduce surface fuels and canopy cover, restoring a forest structure similar to the RNV. This hypothesis was bounded by two alternative hypotheses: (1) fire would have relatively little effect on forest structure, because trees would resist mortality (e.g., Sackett et al. 1996; Covington et al. 2001), or (2) fire would have excessively severe effects, altering forest structure in a manner inconsistent with the RNV and leading to long-term ecological degradation (e.g., Barton 2002).

Methods

Study Area

On 20 September 1993 a 490-ha prescribed fire unit called Northwest III was ignited by Grand Canyon National Park staff in ponderosa pine, aspen, and mixed conifer forests on Swamp Ridge, Kaibab Plateau, in northern Arizona (36°20′N, 112°15′W). Species names are summarized in Table 1. Surface fuels averaged 37.3 Mg/ha of woody debris and 28.3 Mg/ha of forest floor litter and duff (Duhnkrack 1993). On the day of ignition temperatures ranged from 15.6 to 20 °C, and relative humidity ranged from 15 to 32%. Unanticipated high winds gusting up to 48 km/hr led to relatively intense fire behavior beginning on 22 September, including tree torching and spotfires. The status of the fire was converted to a wildfire on 23 September 1993 and was suppressed by 30 September without escaping the original fire boundaries (Nichols et al. 1994).

Paired study sites were selected in the geographic center of the burned area (burn) and an adjacent unburned area (unburn), 270 ha each, separated by a road. Elevation ranged from 2,427 to 2,549 m. Soil information was derived from an ongoing soil survey (A. Dewall, National Resource Conservation Service, personal communication, 2002); soils were tentatively classified as the Elledge family (fine, mixed, superactive, and mesic Typic Paleustalfs). Average annual precipitation at the North Rim ranger station (elevation 2,564 m) is 64.7 cm, with an average annual snowfall of 356 cm. Temperatures range from an average July maximum of 25.1°C to an average January minimum of −8.2°C (Western Regional Climate Center, http://www.wrcc.dri.edu). The unburned study area (Swamp Ridge) was included in a study of forest structure and fire regime over an elevational transect on the Kaibab Plateau. Before 1879 CE, the fire regime was one of frequent surface fires, often covering large areas. The Weibull median probability interval, a statistic often used to assess the average frequency of fire recurrence (Swetnam & Baisan 2003), was 5.06 years and 8.70 years for fires scarring 25% or more of the sample trees, assumed to be larger fires (Fulé et al. 2003). No post-1879 fire was recorded on the site in park records nor was there evidence in the form of fire scarring, except for fire scarring on a single sample tree in 1954 (Fulé et al. 2003). The elimination of surface fires after 1879 was most likely the result of heavy grazing by domestic livestock introduced by European settlers (Altschul & Fairley 1989), removing herbaceous fuels, followed by fire suppression policies when the area was set aside as a reserve in 1893 and then a National Park in 1919 (Fulé et al. 2003).

Before fire exclusion, the frequent fire regime included both lightning- and human-ignited fires. Because it is generally difficult to estimate the cause of a fire from fire-scarred tree data, the role of human-ignited fires has been inadequately quantified in southwestern forests. However, Baisan and Swetnam (1997) and Allen (2002) used evidence from lightning density, fire seasonality, and comparison of populated with unpopulated areas to suggest that human-ignited fires may have played a relatively minor role because of saturation by lightning in southwestern uplands. The Grand Canyon region was not pristine (sensu Denevan 1992) in the sense of an absence of people before European settlement; in fact, the region has a very long history and pre-history of human presence (Altschul & Fairley 1989). However in contrast to areas where burning by indigenous inhabitants fundamentally regulated the fire regime (e.g., Australia [Lewis 1994] and California [Anderson 1996]), the scale of changes in forest ecosystem structure and disturbance regime at Grand Canyon associated with European settlement far exceeded the pre-European range of variability (Fulé et al. 2002*b*, 2003). Thus the date of European settlement/fire-regime disruption is a useful reference point against which to compare forest change due to recent human influence, rather than a switch between alternative anthropogenic fire regimes.

Field Methods

Sampling plot centers were located on a systematic 300-m grid placed over each sampling site. The unburned site was sampled in July and August 1998, and the burned site was sampled in June and July 1999. We allowed a 6-year period after the fire to capture tree mortality, because conifers often take several years to die following fire (Sackett et al. 1996; Kaufmann & Covington 2001). Sampling plots were 0.1 ha (20×50 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, and the distance and bearing from a tagged reference tree to the center was recorded. Tree species and authors are summarized in Table 1. Trees greater than 15 cm diameter at breast height (dbh) were measured on the entire plot (1,000 m^2), and trees between 2.5 and 15 cm dbh were measured on one-quarter plot (250 m^2); all trees were tagged and tree condition was noted (1, live; 2, declining; 3, recent snag [standing dead tree]; 4, loose bark snag; 5, clean snag; 6, snag broken above breast height; 7, snag broken below breast height; 8, downed dead tree [also called log]; and 9, cut stump) (Thomas et al. 1979). All living and dead trees potentially old and large enough to have become established before European settlement (circa 1880) were identified as potentially pre-settlement trees in the field. All conifers with dbh greater than 37.5 cm or ponderosa of any size with yellowed bark (White 1985) were considered potentially pre-settlement trees. All living potentially pre-settlement trees and 10% of all post-settlement live trees were cored for the determination of age and past size. 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 & Block 1994) was recorded at each of the 332-point intercept locations along two 50-m point intercept transects along the outer plot edges. 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 timelag/size classes described by Anderson (1982), where the term timelag refers to the average time needed for the debris to come into equilibrium with atmospheric moisture content: 1-hr timelag (woody fuels >0 and <0.64 cm in diameter), 10-hr (0.64–2.54 cm), 100-hr (2.55–7.62 cm), and 1,000-hr (>7.62 cm, divided into sound and rotten categories). Litter (L layer, undecomposed material) and duff (F and H layers, decomposing material or humus) were measured every 1.52 m along each transect. Eight photopoints were established at each plot from the corners and quarter corners.

Laboratory Methods

Plot areas were corrected for slope. Tree increment cores were surfaced and crossdated (Stokes & Smiley 1968) with local tree-ring chronologies. 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). Fuel loadings were calculated from the planar transect data (Brown 1974; Sackett 1980). Pre-settlement forest structure was reconstructed at the time of disruption of the frequent fire regime, 1879 (Fulé et al. 1997, 2002*b*). Tree 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). To estimate the growth between the fire exclusion date and death date, we developed local species-specific relationships between tree diameter and basal area increment ($r^2 = 0.45$ to 0.90). An analogous process of growth estimation was used to estimate the past diameter of the small proportion of living pre-settlement era trees for which an intact increment core could not be extracted due to rot.

Assessment of fire effects is based on the assumption that the burned and unburned sites were similar before burning, because no pre-fire data exist. This assumption is reasonable in light of the close proximity, shared management history, and similar topography of the sites. Comparisons of forest variables between sites were made with multivariate analysis of variance (SYSTAT, SPSS, Chicago, IL, U.S.A.). Alpha level was 0.05. Variables were square-root transformed where necessary to meet ANOVA assumptions of normality and homoscedasticity. Following a statistically significant ANOVA result, treatment means were compared with a post hoc Tukey's test.

The Forest Vegetation Simulator (FVS) (Van Dyck 2000), Central Rockies/Southwestern Ponderosa Pine variant, was used to simulate stand development in the burned and unburned sites for the next 50 years. Three different scenarios were compared: (1) fire exclusion; (2) repeated prescribed burning of the burn site; and (3) removal of aspen regeneration to simulate severe browsing by ungulates. The prescribed burning was simulated with the fire/fuels extension (FFE) to FVS. Burns were set at 15-year intervals with wind speed at 8.5 km/hr, dry fuel moisture level, and ambient temperature 18.3°C. We chose the 15-year burning interval as a compromise between the historical fire frequency (5–9 years) versus budget and logistical constraints on management. An 8-year interval was also tested for comparison. Severe ungulate browsing occurred historically on the North Rim in the 1920s (Rasmussen 1941; Mitchell & Freeman 1993), and exceptionally heavy browsing of aspen remains common in forests with high elk populations (Dahms & Geils 1997; White et al. 1998; Ripple & Larsen 2000). In contrast to most western forests, elk have been absent from the North Rim for most of the twentieth century, but a small herd has become established recently (R. Sieg, Arizona Game and Fish Department, personal communication, 2002).

Results

The burned site was significantly lower in tree density and basal area than the unburned site (Table 2). Total tree density in the burn averaged 35% that of the unburned site, but the density difference was predominantly in smaller diameter trees, because the burn basal area was 69% that of the unburned site. The reductions were concentrated in fire-susceptible species. White fir and aspen were the two most common species in the unburned site, making up 77% of tree density, but these species made up only 48% of tree density in burn. Fire-resistant ponderosa pine and Douglas fir averaged somewhat lower in density and basal area in burn than in the unburned site, but the differences were not statistically significant. The maximum pine density at any plot, 594 trees/ha, occurred in the burn. Pine made up 60% of the total basal area in the burn but only comprised 48% of the basal area in the unburned site. Engelmann spruce was encountered only in the burn at low density (<12 trees/ha), and New Mexican locust was encountered only in the unburned site, also at low density (<6 trees/ha). Structural variability was greater in the burn site, which included plots with no living trees. In contrast, the minimum values of density and basal area at the unburned site were 309 trees/ha and 18.5 m^2/ha, respectively.

The sites differed significantly in log density but not in snag density. The burned site averaged 5.0 logs >30 cm in diameter per hectare and only 1.0 log >50 cm in diameter per hectare, significantly fewer than the 43.5 logs >30 cm and 21.2 logs >50 cm in diameter per hectare in the unburned area. The burned site averaged 63.3 snags >30 cm in diameter per hectare and 22.8 snags >50 cm in diameter per hectare, not significantly different from the averages of 45.1 snags >30 cm and 24.3 snags >50 cm in diameter per hectare in the unburned area.

Canopy cover in the burn (30%) averaged less than half the canopy cover in the unburned site (63.2%) (Table 3). Seven percent of sample plots in the burn had no canopy cover and 50% of plots had canopy cover below 32%, which was the minimum value of canopy cover in the unburned site. Seedling or sprout densities (Table 4) were nearly equal

Table 2. Comparison of forest structure at burned and unburned sites.

	ABCO	*PIEN*	*PIPO*	*POTR*	*PSME*	*RONE*	*Total*
Density (trees/ha)							
Burn	113.1 ± 24.8	11.7 ± 9.7	135.7 ± 28.3	46.1 ± 29.9	23.7 ± 9.2	0	330.5 ± 57.0
	(0–500.9)	(0–290.5)	(0–594.2)	(0–891.6)	(0–240.9)		(0–1,242.2)
Unburn	466.5 ± 62.4	0	156.6 ± 22.9	255.8 ± 59.8*	56.3 ± 19.9	5.6 ± 5.6	940.7 ± 104.3
	(0–1,132.5)		(20.2–524.4)	(0–1,411.1)	(0–475.3)	(0–168.0)	(309.2–2,417.7)
Basal area (m^2/ha)							
Burn	9.6 ± 4.9	0.2 ± 0.2	17.1 ± 2.2	0.8 ± 0.3	0.8 ± 0.3	0	28.5 ± 4.9
	(0–151.2)	(0–5.0)	(0–48.8)	(0–9.7)	(0–9.0)		(0–153.8)
Unburn	14.5 ± 1.2	0	19.9 ± 1.9	5.3 ± 1.0	1.6 ± 0.5	0.004 ± 0.004	41.3 ± 1.9
	(0–26.8)		(1.6–39.8)	(0–19.7)	(0–10.0)	(0–0.1)	(18.5–62.2)

*Significantly different means.
Species codes are summarized in Table 1. Data are presented as mean ± standard error (range).

Table 3. Canopy cover at burned and unburned sites.

	No. of Plots	*Minimum (%)*	*Maximum (%)*	*Mean ± Standard Error (%)*
Burn	30	0	60.2	30.0 ± 3.0*
Unburn	30	31.6	84.9	63.2 ± 2.2*

*Significantly different means.

across the two sites for the smallest size category (<30 cm tall). Seedlings or sprouts between 30 and 200 cm were only 35% as dense in burn as in the unburned site, a ratio dropping to 23% for regeneration >200 cm tall, but the differences were not statistically significant.

Forest floor litter was significantly higher in burn, but duff was significantly lower (Table 5). All categories of fine and coarse woody debris were significantly lower in burn, except for large sound woody debris (1,000 hr), which was nearly equal in both sites. Total woody debris was 87% higher in the unburned site than in the burn.

Spatial patterns of forest structure showed strong contrasts between the sites (Fig. 1). Levels of structural variables were relatively evenly distributed across the unburned site, but the lowest values of canopy cover, duff depth, and tree density were grouped in the burn (Fig. 2). The west side of the burn had five contiguous plots with canopy cover <15%. As a first approximation of patch size, these aggregated plots represent an area of 45 ha or 30% of the study site. Three of these plots also had basal area <15 m^2/ha, and four of these plots had a density of <100 trees/ha. In contrast, the highest values of structural variables tended to occur on the east side of the burn, but the pattern was less grouped and no distinct patches of dense forest structure were evident. Regeneration density (seedlings and sprouts) had a distinctly different pattern, with patches of dense regeneration occurring sporadically across the landscape of both the burned and the unburned sites.

The fate of old-growth trees is important both for their ecological roles and for their importance in dendroecological forest reconstruction. Because the sites have never been harvested, a relatively high proportion of trees of pre-1879 origin (old-growth) are still living: 43% of the pre-1879 trees in the unburned site were alive in 1998. However fewer old-growth trees were alive in the burn: 66.3 trees/ha compared to 107.9 trees/ha in the unburned site. Most of the difference was in old-growth aspen, 3.7 and 39.5 trees/ha in burned and unburned sites, respectively. Living old-growth ponderosa pine trees were nearly equal in density, 53.9 and 50.9 trees/ha in burned and unburned sites, respectively. The unburned and burned sites were also similar in the density of old-growth snags that died relatively recently (condition classes 3 through 5, as defined in the *Field Methods*), 30.1 and 27.7 trees/ha, respectively, as well as older or broken snags (condition classes 6 and 7), 64.0 and 51.6 trees/ha, respectively. Density of downed old-growth trees was nearly three times higher at the unburned site, however (52.2 trees/ha vs. 18.1 trees/ha in the burn).

Reconstructed forest structure in 1879, at the end of the frequent fire regime, was significantly less dense in the burned than the unburned site, but aspen was the only individual species with a significant difference in reconstructed density (Table 6). Tree density in the burned site was 77% that of the unburned site; basal area was only 69%. As with the other variables measured in this study, these differences might represent actual differences between the two sites or the effect of the fire in consuming dead wood that is important for accurate forest reconstruction. Under the assumption that both sites were similar before the fire, 17% of the burned site (five plots) had current basal area values below the minimum basal area value reconstructed in 1879 in the unburned site. Four of these plots were contiguous, approximately representing a patch of 36 ha (13% of the study site).

Simulated stand development for 50 years under the first scenario, no fire, maintained high basal area at the unburned site and caused basal area to increase to high levels, greater than 40 m^2/ha, in the burned site (Fig. 3). In contrast repeated prescribed burning at a 15-year interval maintained low basal area at the burned site, approximately 20 m^2/ha. Repeated prescribed burning in the unburned site was also predicted to result in low basal area, approximately 23 m^2/ha. Burning at an 8-year interval

Table 4. Density of seedlings or sprouts (number/hectare) in burned and unburned sites.

	ABCO	*ABLA*	*PIEN*	*PIPO*	*POTR*	*PSME*	*RONE*	*Total*
Seedlings <30 cm tall								
Burn	3,336.3	80.1	167.0	173.8	1,444.1	402.4	0	5,603.7
Unburn	2,397.0	0	0	54.1	2,185.5	27.0	640.9	5,304.5
Seedlings 30–200 cm tall								
Burn	261.1	46.8	160.3	20.1	702.3	6.7	0	1,197.3
Unburn	1,653.2	0	0	40.8	1,362.9	6.7	347.7	3,411.3
Seedlings >200 cm tall								
Burn	6.7	13.4	6.7	0	6.7	0	0	33.5
Unburn	60.6	0	0	0	74.2	13.4	0	148.2

Table 5. Forest floor and coarse woody debris at burned and unburned sites.

	Litter (cm)	*Duff (cm)*	*1 hr (Mg/ha)*	*10 hr (Mg/ha)*	*100 hr (Mg/ha)*	*1,000 hr Sound (Mg/ha)*	*1,000 hr Rotten (Mg/ha)*	*Total Wood (Mg/ha)*
Burn	0.6 ± 0.04*	1.3 ± 0.09*	0.5 ± 0.07*	1.7 ± 0.2*	3.9 ± 0.6*	35.9 ± 6.1	13.6 ± 4.1*	55.7 ± 7.3*
Unburn	0.4 ± 0.05*	3.6 ± 0.3*	1.1 ± 0.1*	3.2 ± 0.3*	8.9 ± 1.0*	34.0 ± 5.2	57.5 ± 13.9*	104.7 ± 14.3*

*Significantly different means.
Data are presented as mean ± standard error. Woody debris is described by timelag class (1 hr, etc.).

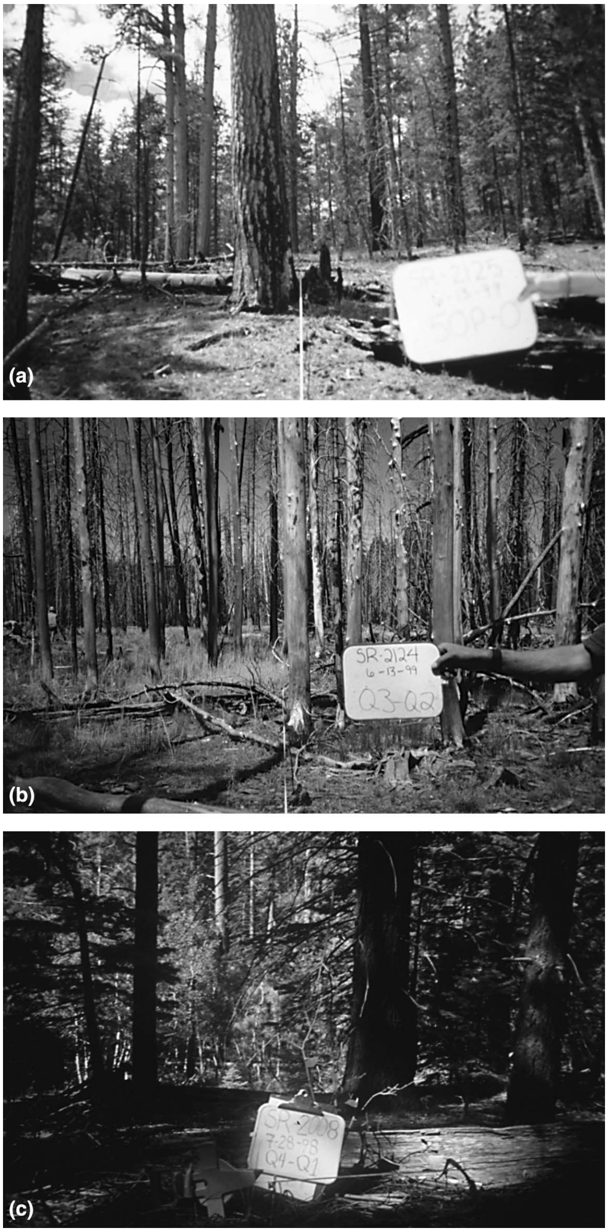


Figure 1. Contrasting scenes within the burn site: the upper plot (a) had complete overstory mortality and a vigorous understory response, while many mature, fire-resistant ponderosa pines survived in the lower plot (b). (c) The unburned site was characterized by dense tree structure, many large, rotten, logs, and minimal understory plant growth. Fuel ladders and fuel jackpots like those shown here contributed to the intense fire at the burned site.

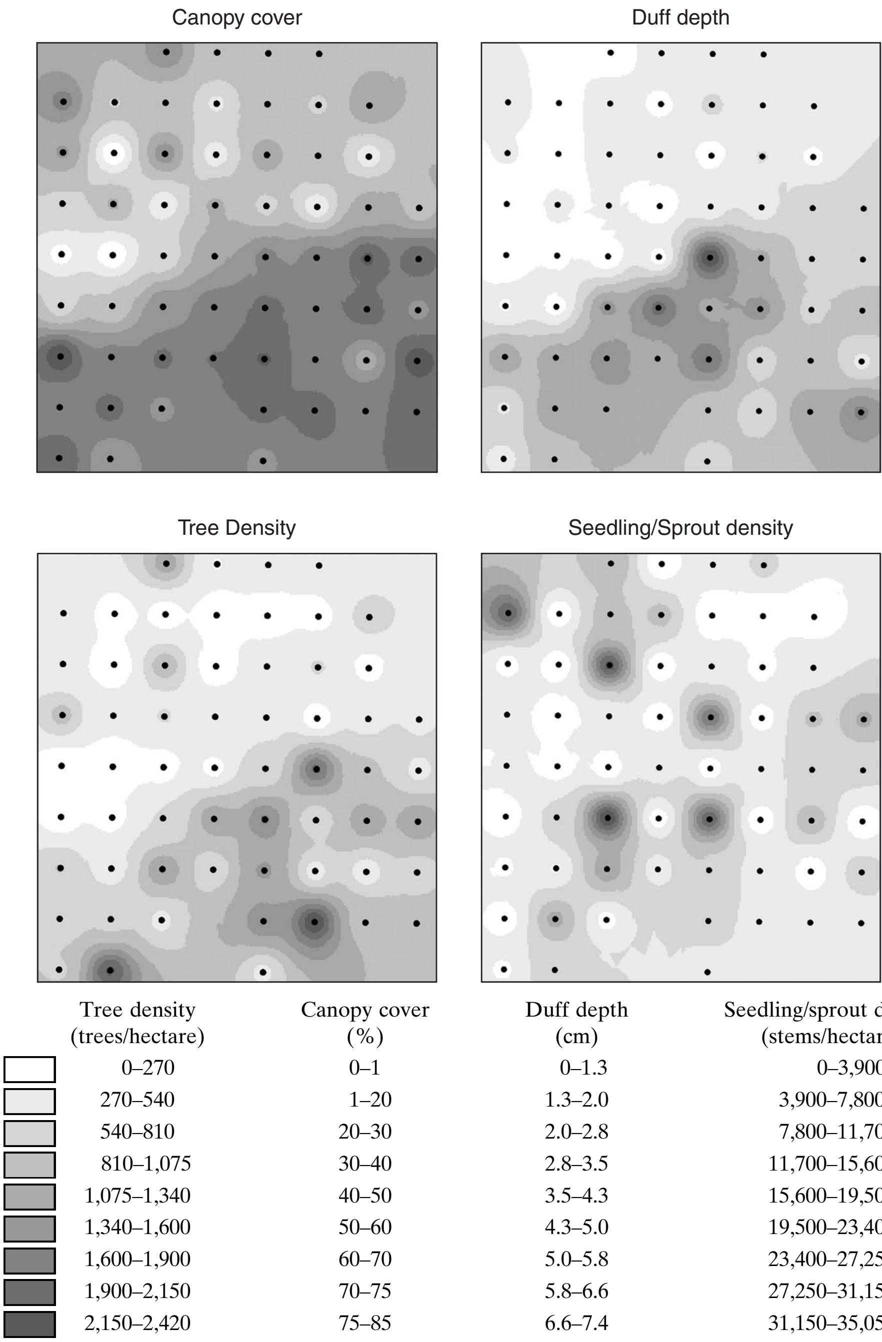
led to similar results but slightly lower basal area (data not shown). In the third scenario, with aspen regeneration removed, basal area in both the burned and unburned sites reached high levels close to 40 m^2/ha, similar to the first scenario. However the forest was increasingly dominated by conifers as the aspen failed to regenerate: the proportion of tree density made up by aspen dropped from 8 to 24% at the start of the simulation in the burned and unburned sites, respectively, to <4% in both sites after 50 years.

Discussion

Reliability and Consistency of Data

A central problem in fire ecology research is the scarcity of replicated experimentation (Whelan 1995). In the present study the burn was not randomly assigned or replicated, and pre-burn data were not collected, constraining the scope of inference. We assessed fire effects by comparing the two sites under the assumption that the burned and unburned sites were essentially equivalent before the fire, allowing the unburned site to serve as a reference or control area. While this assumption is reasonable, there could have been pre-existing differences between sites, and the variability of effects between independent fires is unknown, because the sampling plots in this study are pseudoreplicates (van Mantgem et al. 2001). The study of unreplicated perturbations is nonetheless valuable in circumstances where even limited information is needed for management decisions (including the possible decision to undertake an experiment) (Eberhardt & Thomas 1991).

A second issue related to data reliability is the direct consumption of evidence of pre-1879 forest structure. The reconstructed 1879 forest structure at the unburned site was highly consistent with the data measured in the earliest survey of the Kaibab Plateau, by Lang and Stewart (1910) (Fulé et al. 2002*b*). Under the assumption that burned and unburned sites were similar before the fire, approximately 34 dead and downed old-growth trees/hectare may have been burned without trace, comprising about 60% of the difference in reconstructed 1879 forest structure between the sites. Tree evidence of past forest structure was probably lost in all categories of dead trees (snags and logs). The burned area had reconstructed forest density amounting to 77% of the unburned density and 69% of the unburned basal area, suggesting that roughly one-third of the pre-1879 structural evidence was



Tree density (trees/hectare)	Canopy cover (%)	Duff depth (cm)	Seedling/sprout density (stems/hectare)
0–270	0–1	0–1.3	0–3,900
270–540	1–20	1.3–2.0	3,900–7,800
540–810	20–30	2.0–2.8	7,800–11,700
810–1,075	30–40	2.8–3.5	11,700–15,600
1,075–1,340	40–50	3.5–4.3	15,600–19,500
1,340–1,600	50–60	4.3–5.0	19,500–23,400
1,600–1,900	60–70	5.0–5.8	23,400–27,250
1,900–2,150	70–75	5.8–6.6	27,250–31,150
2,150–2,420	75–85	6.6–7.4	31,150–35,050

Figure 2. Comparison of spatial patterns of canopy cover, duff depth, tree density, and seedling/sprout density across the sampling grids. The intensity of the dark shading is proportional to the values of each variable. The burned site is represented by the dark-colored sampling points, located to the N and NW. The unburned site, with light-colored sampling points, is located to the S and SE.

destroyed by the fire. In a harvested forest, where much of the evidence would have been in the form of dry and decomposing stumps, the loss might have been proportionally much greater. It would not be logical to address Bonnicksen and Stone's (1985: 479) challenge that "we are burning the past in our National Parks" by trying to prevent fire indefinitely, but there is a good basis to argue that time is running out to capture important data on fire regimes and forest structure, because the loss of dendroecological evidence to fire and decay may ultimately lead to greater uncertainty or even misinterpretation of the RNV.

Consistency is the third important issue associated with the interpretation of this study: would another fire in similar forests burn in the exact same way? Probably not,

Table 6. Reconstructed forest structure at the unburned site in 1879.

	ABCO	*PIEN*	*PIPO*	*POTR*	*PSME*	*RONE*	*Total*
Reconstructed 1879 density (trees/ha)							
Burn	36.8 ± 5.8 (0–110.7)	0.7 ± 0.7 (0–20.1)	124.0 ± 13.3 (0–280.7)	22.7 ± 6.0* (0–120.2)	75.4 ± 2.6 (0–70.4)	0	189.6 ± 11.3* (80.1–312.2)
Unburned	31.8 ± 5.2 (0–90.1)		131.5 ± 10.7 (20.2–261.6)	67.9 ± 14.1* (0–250.3)	14.6 ± 4.5 (0–80.9)	0	245.7 ± 12.9* (90.1–373.6)
Reconstructed 1879 basal area (m^2/ha)							
Burn	3.7 ± 0.6 (0–10.6)	0.04 ± 0.04 (0–1.3)	14.9 ± 1.6* (0–27.6)	0.4 ± 0.1* (0–2.9)	70.6 ± 0.3* (0–7.2)	0	19.6 ± 1.6* (2.3–34.7)
Unburned	3.4 ± 0.6 (0–12.0)		21.3 ± 2.0* (5.5–49.7)	1.1 ± 0.3* (0–6.1)	72.7 ± 0.8* (0–15.2)	0	28.5 ± 1.8* (15.1–54.0)

*Significantly different means.
Statistics are presented as mean ± standard error (range).

because the characteristics of canopy burning and spotting are highly uncontrollable and unpredictable even to experienced fire managers. Fires in similar North Rim forests under similar prescriptions have spanned the range from surface fire (e.g., the adjacent Northwest I and Northwest II prescribed fires) to severe crownfire (e.g., the Outlet escaped prescribed fire). However, while the overall effects encountered in the burned site are unlikely to be replicated precisely in another fire, the range of effects provides a useful basis for assessing the impact of intense burning and addressing the research hypotheses.

Assessment of Alternative Hypotheses

In several important respects, the effects of the burn were consistent with Miller and Urban's (2000) hypothesis of structural change toward RNV conditions after intense burning. Quantitative reference data can be applied for direct comparison with forest structure in the burn. Using the reconstructed 1879 forest structure in the unburned site as the point of comparison, post-burn tree density, 331 trees/ha, was within 1.2 standard deviations of the density, 246 trees/ha. Average post-burn basal area was identical to the 1879 value (28.5 m^2/ha). Ponderosa pine was the dominant species after the burn, comprising 41% of tree density and 60% of basal area, compared to 54 and 75%, respectively, in the reconstructed forest. In contrast, white fir was numerically dominant in the absence of fire in the unburned site, and ponderosa pine made up $<50\%$ of basal area. Quadratic mean diameter averaged 33.1 cm in the burn, close to the reconstructed value of 38.4 cm and well above the unburned value of 23.6 cm. These differences suggest that fire-susceptible species and smaller trees were preferentially killed in the burn. Post-burn conditions of canopy cover and forest floor/woody debris were consistent with the expectations of relatively open canopy conditions (30%), thin duff layers (1.3 cm), and low levels of rotten woody debris (13.6 Mg/ha).

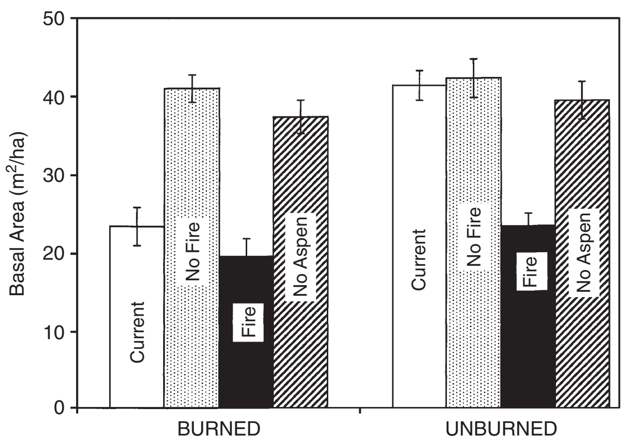


Figure 3. Changes simulated by the Forest Vegetation Simulator and fire/fuels extension in tree basal area over 50 years under three scenarios: (1) no fire; (2) prescribed fire at 15-year intervals; and (3) removal of aspen regeneration due to heavy ungulate browsing. Both the burned and unburned study sites converged on dense forest structure in the absence of fire. Aspen regeneration mortality led to increased conifer dominance. Only repeated burning maintained the current open forest structure of the burned site.

These values can be compared to modern reference sites at Powell Plateau and Rainbow Plateau, at slightly lower elevation (approximately 2,300 m) on the rim of the Grand Canyon where twentieth-century fire exclusion had minimal impact: canopy cover averaged 49.1%, duff depth 1.9 cm, and rotten woody debris 3.7 Mg/ha (P. Z. Fulé et al., unpublished data; Fulé et al. 2002*b*). Finally, these structural changes were accomplished without the negative impacts usually associated with mechanical intervention: roads, soil compaction, logging damage, and introduction of exotic species. Instead, understory plant communities in the burn were vigorous and nearly entirely composed of natives (K. Huisinga 2000, Northern Arizona University, personal communication).

Although the average conditions supported Miller and Urban's (2000) hypothesis, the variability in post-burn forest structure was greater than in 1879. The standard error of the mean (SEM) for tree density, 57 trees/ha, was 4.4 times greater than the SEM in 1879, 12.9 trees/ha. Similarly, the SEM for post-burn basal area was 2.7 times higher than the

1879 value, 4.9 and 1.8 m^2/ha, respectively. This variability was spatially aggregated, especially in the relatively deforested patches at the western end of the burn. As many as 90% of the old-growth aspens may have been killed by the fire. These comparisons underscore the complexity of evaluating alternative forest structures in a restoration context. Even though average post-burn values of forest density and basal area were very similar to reference conditions at the stand scale, habitats at finer scales may be very different.

The two bounding hypotheses—little effect of the fire versus excessive fire severity—were supported to a limited extent within the heterogeneous effects of the fire. Some portions of the site had relatively little tree mortality, with post-fire maximum density reaching 1,242 trees/ha, in contrast to the western side of the fire where heavy mortality occurred. However, the patches of high mortality were relatively small (one sample plot representing approximately 9 ha with complete mortality and seven plots representing approximately 63 ha of canopy cover <15%) and regeneration averaged over 2,800 seedlings or sprouts per hectare on the low-canopy plots; hence there was no evidence of large-scale conversion to a non-forest vegetation type, as observed by Barton (2002) in southern Arizona.

Implications for Management

The future scenarios, contrasted with FVS simulation modeling, illustrate the importance of active management in restoration and conservation of southwestern forests. The Northwest III fire caused a significant shift in forest structure toward reference conditions, but the effects disappeared within 50 years in the absence of fire. Recurring fire, even at relatively long intervals, was projected to maintain open forest characteristics. Like many other U.S. land management agencies, Grand Canyon National Park has an active policy of "wildland fire use for resource benefits," under which lightning-ignited fires can be allowed to burn within certain constraints. However it is relatively unlikely that the burned site would be re-burned exclusively with naturally ignited fires that are allowed to spread. The area remains surrounded by dense forest with heavy fuel loads along the park boundary, a location of special sensitivity to Park Service managers (Pyne 1989), making it difficult for managers to allow lightning fires to burn. Therefore, intentional prescribed burning of the burned site should be a high priority. When aspen regeneration was removed, simulating the effect of elk herbivory, the proportion of aspen dropped to unprecedented levels (<4%) within 50 years (aspen comprised 28% of trees in the reconstructed forest), but total basal area still rose to high levels.

The Northwest III fire marked an important turning point in forest management at Grand Canyon National Park. Pyne et al. (1996) outlined a historical series of "problem fire types" that forced paradigm shifts in American fire policy at approximately 20-year intervals. For example, the Wilderness fire era of the 1970s and 1980s was a period of prescribed burning and prescribed natural fire (a policy of allowing some lightning-ignited fires to burn), which underwent serious reconsideration after the Yellowstone fires of 1988. At Grand Canyon, the Northwest III fire was the realization of earlier predictions (Davis 1981) that fuel loads on the North Rim were too high for consistent control by prescribed burning. A post-fire review concluded that some level of mechanical treatment was a useful option (Nichols et al. 1994), facilitating an experiment to test combinations of thinning and burning on the North and South Rims (Covington et al. 1997). Evaluating the results from the first treatment block in this experiment, Fulé et al. (2002*a*) recommended that tree thinning could be used along roads and boundaries in the park to create defensible fire buffers, permitting fires in the interior of burn blocks to burn with greater intensity. In retrospect, the present study has suggested that the effects of the Northwest III fire were broadly consistent with restoration of RNV conditions. Deliberate use of intense burning will likely always pose a greater challenge for managers than underburning, because of the greater risk of escape and the perception that the burning may damage the forest. But administrative and public support can be enhanced if careful measurement of ecological effects shows that such burning can meet restoration goals. In remote settings like Grand Canyon, large-scale management tests of intense prescribed fire within secure boundaries may prove to be a more fruitful direction for adaptive management experimentation than continuing attempts to underburn dense, fire-excluded forests.

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