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Comparing ecological restoration alternatives: Grand Canyon, Arizona

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Abstract

Three treatments designed to initiate the process of restoring the surface fire regime and open forest structure of a southwestern ponderosa pine forest were compared on the Kaibab National Forest along the Grand Canyon's South Rim. The treatments were: (1) *full restoration* (FULL)—thinning trees to emulate stand structure prior to fire regime disruption ca. 1887, forest floor fuel treatment, and prescribed burning, (2) *minimal thinning* (MIN)—removing young trees only around living old-growth (pre-1887) trees, fuel treatment, and prescribed burning, (3) *burn-only* (BURN)—representing the current management policy in Grand Canyon National Park (GCNP), and (4) CONTROL. Each treatment was applied to a 12 ha unit. Compared to reconstructed 1887 conditions, all study sites were much more dense prior to treatment (94–176 trees/ha in 1887, compared to 783–3693 trees/ha in 1997). However, basal area increases were less striking (12.6–20.3 in 1887, 17.5–27.0 m^2/ha in 1997), reflecting past harvest and dwarf mistletoe reduction treatments that removed many large pines. In 2000, 1 year after treatment, tree densities were reduced to 11, 23, and 37 of pre-treatment levels in the FULL, MIN, and BURN treatments, respectively. Understory plant communities showed significant declines in richness and plant frequency across years, probably due to a severe drought in 2000 (60% of average precipitation). No differences in plant communities were observed across treatments, despite the mechanized disturbance associated with tree removal in the FULL treatment. Prescribed fire behavior (flame length, flaming zone depth) and effects (bole char, crown scorch) were similar across all three burned treatments. Simulated fire behavior under dry, windy conditions was reduced in all three treatments compared to the control. The FULL treatment was much less susceptible to crownfire due to reduced crown bulk density and crown fuel load and increased crown base height. Crownfire susceptibility of the BURN treatment was only slightly reduced, while the MIN treatment was intermediate. Compared to the reference conditions of forest structure, the FULL treatment represented the most rapid and comprehensive restoration treatment, although the residual stand was at the low end of historical density. The BURN treatment thinned many small trees but had minor effects on crownfire susceptibility. Effects of the MIN treatment fell between FULL and BURN. The experimental treatments may be useful for the creation of defensible firebreaks near developments, roads, and boundaries with the FULL treatment, supplemented by MIN and BURN treatments over larger areas.
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1. Introduction

The largest stand-replacing forest fires in the recorded history of the southwest have occurred in 2 of the past 5 years, 1996 (Horseshoe, Hochderffer, Bridger-Knoll, Dome, Hondo) and 2000 (Cerro Grande, Outlet, Pumpkin, Viveash). Severe and costly crownfires in western ponderosa pine and related long-needled pine forests are recognized as a symptom of underlying ecosystem degradation (Leopold, 1924; Cooper, 1960; Moore et al., 1999). Contemporary conditions in virtually all frequent-fire adapted forests of western North America differ greatly from the evolutionary environments or range of natural variability of the biota (e.g. Covington et al., 1994; Arno et al., 1995a; Minnich et al., 1995; Fulé and Covington, 1997; Dahms and Geils, 1997; Millar and Wolfenden, 1999), leaving ponderosa forest ecosystems vulnerable to severe fires, pathogen outbreaks, or non-native species invasions. As the scale of disturbance size and intensity increases, forests may cross thresholds to alternative stable states, such as grasslands or shrublands (Holling, 1992; Romme et al., 1998).

Three distinct approaches exist for dealing with increasingly severe wildfires. First, in some situations there may not be an underlying ecological problem. For example, Swetnam et al. (1999) argued that increasing density in the twentieth century of pinyon trees at the northern end of its range in Colorado was not related to fire suppression or overgrazing, but was caused by northward expansion following long-term climate warming. Brown et al. (1999) and Shinneman and Baker (1997) presented evidence that relatively long fire-free periods (>100 years) and stand-replacing fires may have occurred in ponderosa pine forests of the central Rockies and Black Hills prior to European settlement. In such circumstances there may be no ecological rationale for management action.

A second approach is to focus on hazardous fuels with treatments including logging, thinning, chipping or utilization of slash, and/or prescribed burning (Kalabokidis and Omi, 1998; Scott, 1998a; McIver et al., 2001; USDA/USDI, 2000), as well as modification of fuels immediately surrounding structures (Cohen, 1995).

Ecological restoration is a third approach. Reversing recent deleterious changes and restoring more nearly natural conditions—that is, conditions characteristic of the evolutionary environment of an ecosystem—is central to restoration ecology (Society for Ecological Restoration, 1993). The paradigm of 'ecosystem management' is linked to ecological restoration because it takes the best possible understanding of the structure, function, and composition of intact, natural ecosystems as a point of reference for management strategies (Kaufmann et al., 1994; Landres et al., 1999). Where evolutionary environments can be maintained or restored at large enough scales, principles of conservation biology suggest that these habitats are most likely to perpetuate native plants and animals and allow their future evolution (Noss, 1991; Grumbine, 1992; Moore et al., 1999). Many ecosystem changes that have already occurred are permanent (species extinction) or have long-lasting effects (loss of old-growth trees, atmospheric CO_2 increase), leading some to argue that restoration goals may be misguided because the evolutionary environment cannot be fully regained or because the current environment might have superior qualities (Millar and Wolfenden, 1999; Tiedemann et al., 2000; Wagner et al., 2000). Certainly any management regime should proceed from careful assessment of ecological and social issues. In many cases there are specific reasons for managing ecosystems in conditions far removed from the range of natural variability. But the idea that native species and communities are most likely to benefit from restoration of conditions as close as possible to the evolutionary environment is integral to US resource policy (i.e. ecosystem management) and law. Under the Endangered Species Act, for example, it is not acceptable to permit the extinction of a native species under the argument that a "superior" exotic species is available instead. This link between conservation of native biological diversity and restoration of native habitats underscores the broad philosophical goal of a restorative approach, "not to revive the past . . . but to secure our future by restocking a dangerously depleted global inventory of natural areas" (Clewell, 2000, p. 217). While a laudable goal, testing of specific treatment alternatives is essential.

In ponderosa pine forests, restoration treatments started with the reintroduction of surface fire (Weaver, 1951; Biswell, 1972; Covington and Sackett, 1984). Fire alone was usually insufficient to restore the open, crownfire-resistant forest structure characteristic of pre-European settlement conditions (Sackett et al., 1996).

Therefore, recent restoration methods have focused on tree thinning as well as burning, guided by detailed dendroecological and historical reconstructions of pre-disruption conditions (Arno et al., 1995b; Southwest Forest Alliance, 1996; Covington et al., 1997; Scott, 1998a; Moore et al., 1999; Lynch et al., 2000).

The National Park Service was a pioneer among US resource management agencies in recognizing and attempting to reverse the deleterious effects of fire exclusion (Pyne, 1982; Sellars, 1997). Since the 1970s, Grand Canyon National Park (GCNP) has intensively pursued restoration of the natural role of fire in its approximately 38,500 ha of ponderosa pine, ponderosa pine/Gambel oak, and mixed conifer forest. Despite many successful burns, intense fire behavior due to dense forest conditions led to several costly escaped fires. Reviews of the fuel situation by Davis (1981) and the Interagency Task Force (Nichols et al., 1994) reached similar conclusions, recommending aggressive treatment of accumulated fuels and an expansion of prescribed burning. The 1994 review suggested using mechanical treatments in limited circumstances to thin ‘‘understory trees which have grown into the area following fire exclusion’’ in order to ‘‘assist in restoration of natural fire regimes, ecosystem stability, and protection of mature overstory trees’’.

In 1997, we developed a cooperative project with Grand Canyon and the Tusayan District of the Kaibab National Forest to test several ecological restoration approaches on small areas. All treatments included restoration of surface fire but varied in tree thinning and fuel treatments.

2. Methods

2.1. Study area

The experiment was conducted on a 50 ha site in the Tusayan Ranger District of the Kaibab National Forest on the border of GCNP. The elevation was approximately 2290 m with gentle slopes, averaging 7%. The forest habitat type is ponderosa pine/Gambel oak (Larson and Moir, 1987). Soils in GCNP adjacent to the site are classified as fine, smectitic, mesic, Vertic Paleustalfs and Haplustalfs, clay soils weathered from calcareous sandstone parent material (Lindsay, 2000, personal communication). Average precipitation is 36.8 cm, including average annual snowfall of 177.5 cm, at Grand Canyon Village, approximately 2097 m elevation and 20.4 km NW of the study area (GCNP, 1992). Temperatures range from an average maximum of 29 °C in July to 8 °C in January. Precipitation varied substantially in the specific measurement years of this study: the water year (1 October–30 September) for 1997 recorded 43.3 cm precipitation (117% of average) while the water year for 2000 recorded only 22.4 cm precipitation (61% of average).

Human influence in the Grandview area, as evidenced by prehistoric cultural resources, was likely linked to the ponderosa pine/pinyon–juniper ecotone where a diversity of valued plants and animals would have been attractive to both nomadic and sedentary groups (Hevly, 1988). The region around the study site included the first Euro-American developments at Grand Canyon: Hance and Hull cabins, the Grandview trail and mine, several early tourist camps and the Grandview Hotel. Euro-American settlement was associated with the cessation of the frequent forest fire regime after 1887, livestock grazing, predator control, tourism, mining, and logging for construction materials and mine timbers. Recent management impacts included the Grand Canyon mistletoe control project of 1949–1952 in which infected trees were pruned or thinned along a broad swath of the South Rim (Lightle and Hawksworth, 1973). Past harvest included individual tree selection and group selection methods focusing on mature ponderosa pine trees. The site was not grazed by livestock since prior to 1997 and was fenced in 1999 as part of this experiment. Heavy use by mule deer and Rocky Mountain elk was observed in the drought year of 2000, possibly due to the placement of a temporary water reservoir as a firefighting contingency just north of the study site.

2.2. Treatment design

Four treatments were developed with input from Park Service and Forest Service staff as well as public comments: (1) a *full ecological restoration* treatment (FULL), designed to emulate the structure of presettlement forests, treat fuels, and restore fire in prescription; (2) a *minimal thinning* treatment (MIN), designed to reduce fire hazard and facilitate prescribed fire; (3) a *burn-only* treatment (BURN); (4) a *control* treatment (CONTROL). These treatments cover a range of alternatives in taking the first steps toward restoring ecosystem characteristics. The experiment was originally designed in three blocks, each containing a replicate of the four treatments ($N = 12$). Pre-treatment measurements were carried out on all three blocks, the Kaibab National Forest site described here and two GCNP sites. The Grand Canyon sites were not treated due to obstacles in the environmental assessment process. As a result, the experimental design was reduced to a before–after control-impact (BACI) design (Stewart-Oaten et al., 1992) with $N = 4$.

The reference condition selected for the FULL thinning was the pre-settlement pattern of tree species composition and spatial arrangement (White, 1985; Fulé et al., 1997; Covington et al., 1997; Mast et al., 1999). Living pre-settlement trees of all species were retained. In addition, wherever evidence of remnant pre-settlement material was encountered (snags, stumps, logs), several of the largest post-settlement trees of the same species within 9.1 m were retained as replacements. If suitable trees were not found within 9.1 m, the search radius was extended to 18.2 m. Each remnant was replaced with 1.5 trees (i.e. three replacements per two remnants) if the replacements were 40.6 cm diameter at breast height (dbh) or larger, otherwise each remnant was replaced with 3 trees.

The MIN treatment focused on protecting living old-growth trees from crownfire. To a limited extent, the MIN was also expected to reduce tree competition (Biondi, 1996). Around each old tree, young trees were thinned to interrupt the continuity of crown fuels both horizontally and vertically, converting the fuel complex in the immediate vicinity to a savannah type (roughly fire behavior fuel model 2, Anderson, 1982) instead of a forest type (fuel model 9). The thinning radius ranged from 12 to 18 m, proportional to the height of the focal tree. The thinned area was anisotropic with the longest radii to the SW and/or downhill on sloping sites, in order to provide maximum protection from the prevailing SW wind direction and/or from downslope fires. Within the thinning area, cutting nearest the old tree was similar to the FULL treatment, with nearly all young trees removed. Toward the outer edges of the thinning radius, the thinning was feathered into the unthinned surrounding stand.

The presence of ponderosa pine trees severely infested with dwarf mistletoe affected the thinning designs. Hawksworth and Geils (1990) estimated the mean time to death of 50% of infected trees at Grand Canyon as 7–25 years for dwarf mistletoe rating (DMR) of 5–6 on a scale of 0–6. We chose to retain all pines of pre-settlement origin, irrespective of mistletoe infestation. But since post-settlement pines with a DMR of 5 or 6 were highly likely to die in the near future, they were not considered acceptable replacement trees.

In both the FULL and MIN treatments, accumulated forest floor fuels were raked approximately 30 cm away from the base of the boles of old-growth trees in order to minimize cambial girdling by fire (Sackett et al., 1996). The BURN treatment was intended to imitate current management practices at GCNP so fuels were not raked from tree boles.

2.3. Field methods

Treatments were randomly assigned to four forest units, each nominally 12 ha in size (actual range 11.9–13.4 ha). Twenty permanent monitoring plots were established in each unit (total $N = 80$ plots) between 20 August and 3 November 1997. Plots were located on a 60 m grid, corresponding to a measured experimental area of 7.2 ha per treatment unit. Plot centers were established with tape and compass from surveyed reference points, such as section corners. Global positioning systems were used to geo-reference plot grids. Centers were permanently marked with iron stakes and slope and aspect were recorded. Photos were taken to plot center from 11.28 m NE.

Overstory trees taller than breast height (137 cm) were measured on a 400 m^2 (11.28 m radius) circular fixed-area plot. Species, condition (living or snag/log classes (Thomas et al., 1979)), diameter at breast height (dbh), and a preliminary field classification of pre-settlement or post-settlement origin, were recorded for all live and dead trees over breast height, as well as for stumps and downed trees that surpassed breast height while alive. Potentially pre-settlement ponderosa pine trees were identified based on size (>40 cm diameter at "stump height" (dsh), 40 cm above ground level) or yellowed bark (White, 1985). Trees of all other species, oaks, pinyons, and junipers, were considered as potentially pre-settlement if dsh >20 cm. All potentially pre-settlement trees, as well as a random 10% subsample of other trees, were cored with an increment borer at 40 cm above ground level to determine age and past size, as described below. Diameter at stump height was recorded for all cored trees. All overstory trees were marked with aluminum tags at breast height and tree locations were mapped.

Regeneration (trees below breast height) and shrubs were tallied by condition class and by three height classes (0–40, 40.1–80, and 80–137 cm) on a nested 100 m^2 (5.64 m radius) subplot. The point-line intercept method (line transect) was used to collect herbaceous and shrub data on all plots. Plant species, substrate, and overstory canopy cover (vertical projection of canopy taller than 137 cm) were recorded every 30 cm along a 50 m line transect oriented upslope with 25 m above and 25 m below the plot center. Species were also recorded within 5 m to either side of each transect, forming a 10 m wide belt transect on each plot. Dead woody biomass and forest floor material were measured on a 50 ft planar transect in a random direction from each plot center.

Trees were marked for retention and thinning was carried out under a Forest Service contract. Total costs (marking, thinning, fuel raking, fencing, and prescribed burning) were US$ 748/ha in FULL and US$ 566/ha in MIN. The prescribed fire costs alone (BURN treatment) averaged US$ 44/ha (Johnson, 2000, personal communication).

Forest floor fuels were re-measured on 1 September 1999, prior to burning. The treatment units were burned with strip headfires on the afternoons of 18 and 20 October 1999. Winds ranged from 0 to 7 km/h, wind direction was primarily from the north and east, and relative humidity varied from 14 to 28%. Average flame lengths were 25–120 cm (corresponding to fireline intensities of 13–383 kW/m (Agee, 1993)) with maximum flame lengths reaching 2.5–3 m (1884–2799 kW/m). Flaming zone depths ranged from 0.25 to 1.8 m. Fires burned primarily on the surface, but some passive crownfire (torching) was observed in the BURN unit. Plot 1 in the CONTROL unit was unintentionally burned with high tree mortality. This plot was removed from analysis.

After burning, fuels were re-measured on 5 November 1999. Burn severity codes (five categories, unburned to complete consumption) were recorded at each forest floor measurement point. All variables on all permanent plots were re-measured during 4–8 August 2000. In addition, total height, crown base height, crown scorch (height and percent), bole char (minimum and maximum height), and dwarf mistletoe rating (0–6) were measured on all trees. Burn severity codes (four categories, unburned to completely burned) were assigned to vegetation and substrate (litter, rock, soil, wood, scat, bole) at each intercept point along the line transects.

2.4. Laboratory, statistical, and modeling analysis

Increment cores were surfaced and visually crossdated (Stokes and 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, 1887, 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). To estimate growth between the fire exclusion date and death date, we developed local species-specific relationships between tree diameter and basal area increment ($r^2 = 0.45–0.90$). 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.

Comparisons of forest variables on the 20 sample plots between treatments and over time (pre- and post-treatment) were made with repeated-measures analysis of variance (ANOVA) using Systat (SPSS Inc., Chicago, IL, 1998). Alpha level was 0.05. Variables were square-root transformed where necessary to meet ANOVA assumptions of normality and homoskedasticity. Following a statistically significant ANOVA result, treatment means were compared with a post-hoc

Tukey's procedure. Collapsing a replicated experiment to a single realization of each treatment substantially limited statistical inference because of the lack of knowledge of within-treatment error, an undesirable but common occurrence in environmental field studies (Eberhardt and Thomas, 1991). Significant differences mean that the populations in each unit differ but treatment cause-and-effect cannot be inferred in a statistical sense. In a practical sense, however, it is logical to infer that intentional treatments, such as cutting trees and burning, *caused* direct effects such as declines in tree density or consumption of fuels. A much weaker level of inference would be appropriate for assessing more subtle alterations or those where mechanisms are less apparent.

Herbaceous community data analysis included calculations of plant and substrate frequencies, species richness, Simpson's index (SI, richness weighted by frequency), height classes, and cumulative species curves. PC-ORD (MjM Software, Gleneden Beach, OR, 1999) was used for community analyses, including species area curves, cluster analyses, ordinations (non-metric multidimensional scaling (NMDS)), and indicator species analysis.

Fire behavior was modeled with the Nexus Fire Behavior and Hazard Assessment System (Scott and Reinhardt, 1999). Crown biomass was estimated with allometric equations for foliage and fine twigs of ponderosa pine (Fulé et al., 2001), Gambel oak (Clary and Tiedemann, 1986), and pinyon and juniper (Grier et al., 1992). Crown volume was estimated by the averages of maximum tree height (top of the canopy) and crown base height (bottom of the canopy). Crown bulk density was calculated as crown biomass divided by crown volume. Crown base height values measured in the CONTROL treatment in 2000 were used to estimate pre-treatment values for all stands. Since the majority and presumably the largest of the pre-settlement era trees had been previously logged on the study site, pre-settlement stand height (20 m) and crown base ($r^2 = 0.45-0.90$) height (4.88 m) values were estimated at the 90th percentile of trees >10 m tall in 1997.

Fire weather extremes representing the 90th and 97th percentiles of low fuel moisture, high winds, and high temperature were calculated from 34 years of data on the Kaibab National Forest (Tusayan weather station) using the FireFamily Plus program (Bradshaw and Brittain, 1999). Weather values were calculated for the entire fire season (23 April–16 October) as well as for June, historically the month with the most severe fire weather (Table 1). Fire behavior information from two of the two largest wildfires in northern Arizona, the 1996 Horseshoe (May) and Hochderffer (June) fires, was used to estimate wind gusts during periods of extreme fire behavior. Sustained winds of 51 km/h were commonly observed on these fires.

Table 1
Fuel moisture, wind, and temperature for the Tusayan weather station (Kaibab National Forest), 1966–1999

Variable	Fire season (23 April–16 October)		June	
	90th percentile	97th percentile	90th percentile	97th percentile
1H moisture (%)	3.3	2.6	2.3	1.7
10H moisture (%)	4.4	3.4	3.0	3.0
100H moisture (%)	6.8	6.4	4.5	4.5
Wind speed (km/h)	22.1	28.7	25.5	32.3
Temperature (°C)	29.3	29.3	32.2	32.2

"Average" stand conditions fail to represent the dispersed fuel ladders that facilitate the transition to the crown in real fires, making simulated fires difficult to crown even though real fires crowned under similar or even less severe conditions. Taking the variability of the data into account to simulate more realistic fire behavior, we ranked the crown base height data by quintiles (20% categories) and compared fire behavior and treatment effects on both the stand averages and the susceptible quintiles.

Growth of retained trees for 40 years following the treatments was simulated with the forest vegetation simulator (FVS, Van Dyck, 2000), central Rockies/southwestern ponderosa pine variant. Canopy fuels were calculated for the stands in 2040 as described above.

3. Results

3.1. Changes in tree structure

In 1887, at the end of the pre-settlement frequent-fire regime, forest structure was relatively open (93.8–176.3 trees/ha, 10.6–20.3 m^2/ha), with ponderosa pine

Table 2
Forest basal area (m^2/ha) at the time of fire regime disruption (1887), prior to treatment (1997), and 1-year post-treatment (2000)[a]

Treatment	1887			1997			2000		
	Mean	Range	S.E.M.	Mean	Range	S.E.M.	Mean	Range	S.E.M.
CONTROL									
JUOS	0			0.0001	0–0.001	0.00007	0.0001	0–0.001	0.00007
PIED	0			0.003	0–0.6	0.003	0.003	0–0.6	0.003
PIPO	18.5	0.4–48.1	3.3	16.9	0–39.3	2.6	16.9	0–39.3	2.6
QUGA	1.8	0–19.4	1.0	5.7	0–28.6	1.5	5.7	0–28.6	1.5
Total	20.3 a	0.4–48.1	3.3	22.7 ab	5.4–41.6	2.6	22.6 a	5.3–41.5	2.6
FULL									
JUOS	0			0.1	0–1.3	0.07	0		
PIED	0			0.02	0–0.35	0.02	0		
PIPO	12.3	0–30.8	1.7	13.8	2.5–27.8	1.8	4.2	0–15.4	1.1
QUGA	0.7	0–3.3	0.2	3.5	0–11.0	0.8	1.9	0–10.5	0.6
Total	13.0 ab	0.2–33.5	1.9	17.5 b	2.7–36.8	1.8	6.2 b	0–16.6	1.1
MIN									
JUOS	1.7	0–17.0	1.0	3.3	0–24.0	1.6	2.4	0–18.7	1.3
PIED	0.007	0–0.1	0.007	0.2	0–2.0	0.1	0.1	0–1.9	0.1
PIPO	10.6	0–47.2	2.5	18.1	2.5–33.2	2.3	10.4	0–24.5	1.6
QUGA	0.2	0–2.2	0.1	0.9	0–7.2	0.4	0.5	0–3.0	0.2
Total	12.6 ab	0.8–47.3	2.4	22.5 ab	6.3–36.0	2.1	13.4 c	3.0–28.0	1.4
BURN									
JUOS	0.3	0–5.0	0.3	0.6	0–10.0	0.5	0.5	0–9.0	0.4
PIED	0			0.002	0–0.04	0.002	0		
PIPO	9.8	0–31.5	2.5	21.6	2.3–40.5	2.5	16.8	2.3–31.3	2.0
QUGA	0.5	0–2.3	0.2	4.8	0–20.5	1.4	4.4	0–20.5	1.3
Total	10.6 b	0–32.1	2.5	27.0 a	4.8–48.4	2.5	21.7 ac	4.8–47.1	2.4

[a] JUOS: *Juniperus occidentalis*; PIED: *Pinus edulis*; PIPO: *Pinus ponderosa*; QUGA: *Quercus gambelii*. Total basal area differed significantly across years. Letters indicate significantly different means by treatment within years. $N = 20$ for all treatments except control ($N = 19$).

making up 84–95% of basal area (Table 2). By 1997, all the study units were significantly more dense. The units differed in tree structure prior to treatment, with tree densities ranging from 782.9 to 3692.5 trees/ha and basal areas from 17.5 to 27.0 m^2/ha (Tables 2 and 3). Ponderosa pine dominated all sites, making up 53–91% of trees/ha and 74–80% of total basal area. Treatments had rapid and substantial effects on tree structure (Fig. 1). In FULL treatment, tree density was reduced to 11% of pre-treatment density, from 1337.5 to 153.8 trees/ha. Basal area declined to 35% of pre-treatment levels, from 17.5 to 6.2 m^2/ha. Sixty-nine percent of the tree density decline was due to tree thinning and 31% to mortality from fire or other causes. The MIN treatment reduced density to 23% of pre-treatment density, from 2935.0 to 683.8 trees/ha. Basal area declined to 59% of pre-treatment levels, from 22.5 to 13.4 m^2/ha. Fifty percent of the tree density decline was due to tree thinning and 50% to mortality from fire or other causes. The BURN treatment reduced density to 37% of pre-treatment density, from 3692.5 to 1383.8 trees/ha. Basal area declined to 80% of pre-treatment levels, from 27.0 to 21.7 m^2/ha. All of the tree density decline in the BURN treatment was due to mortality from fire or other causes. The untreated CONTROL site changed by less than 4% in tree density and less than 1% in basal area during the same period.

Relative dominance by ponderosa pine was reduced slightly in the MIN and BURN treatments, declining a

Table 3
Forest density (trees/ha) at the time of fire regime disruption (1887), prior to treatment (1997), and 1-year post-treatment (2000)[a]

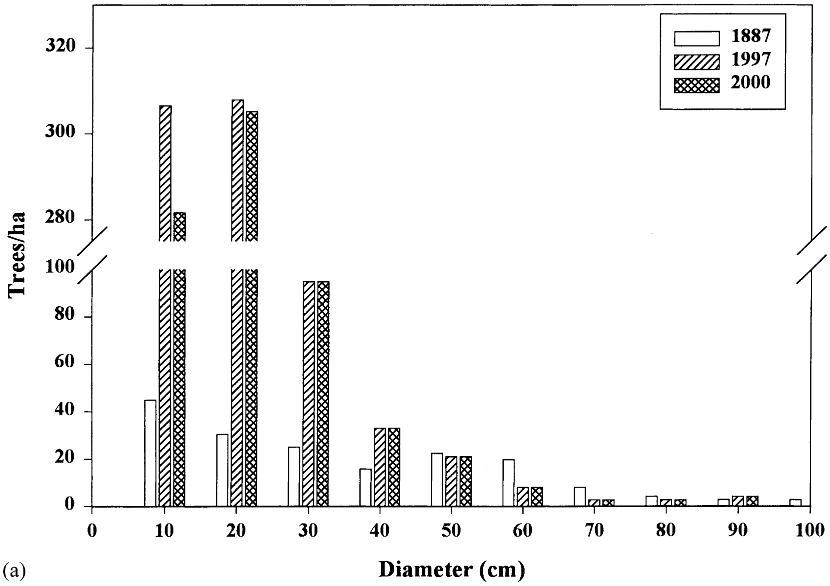
Treatment	1887				1997				2000			
	Mean	Range	S.E.M.	QMD[b]	Mean	Range	S.E.M.	QMD[b]	Mean	Range	S.E.M.	QMD[b]
CONTROL												
JUOS	0				2.6	0–25	1.8	0.7	2.6	0–25	1.8	0.7
PIED	0				1.3	0–25	1.3	5.4	1.3	0–25	1.3	5.4
PIPO	106.6	25–225	13.6	47.0	411.8	0–1525	96.0	22.9	406.6	0–1525	94.5	23.0
QUGA	69.7	0–250	16.1	18.1	367.1	0–1825	100.5	14.1	344.7	0–1775	96.1	14.5
Total	176.3 a	25–325	17.2		782.9 a	150–2450	135.2		755.3 a	150–2375	131.9	
FULL												
JUOS	0				18.8	0–125	7.7	8.2	0			
PIED	0				6.3	0–50	3.1	6.4	0			
PIPO	60	0–125	6.9	51.1	873.8	125–3825	234.9	14.2	42.5	0–125	8.7	35.5
QUGA	41.3	0–125	10.1	14.7	437.5	25–2225	121.4	10.1	111.3	0–550	33.5	14.7
Total	101.3 b	25–250	14.6		1337.5 ab	350–4575	273.3		153.8 b	0–600	35.0	
MIN												
JUOS	40.0	0–300	19.5	23.3	176.3	0–950	56.9	15.4	78.8	0–350	27.5	19.7
PIED	1.3	0–25	1.3	8.3	20.0	0–100	7.6	11.3	6.3	0–50	3.6	14.2
PIPO	58.8	0–225	12.5	47.9	2581.3	75–8750	604.5	9.4	555.0	0–2000	124.3	15.4
QUGA	13.8	0–100	6.1	13.6	157.5	0–850	54.7	8.5	43.8	0–250	15.4	12.1
Total	113.8 b	25–300	18.6		2935.0 bc	550–9050	592.4		683.8 a	125–2325	136.8	
BURN												
JUOS	2.5	0–50	2.5	39.1	50.0	0–600	30.0	12.4	16.3	0–150	8.6	19.8
PIED	0				3.8	0–50	2.7	2.6	0			
PIPO	45.0	0–125	9.2	52.7	3343.8	175–18600	989.9	9.1	1160.0	75–2950	187.0	13.6
QUGA	46.3	0–250	15.0	11.7	295.0	0–900	71.4	14.4	207.5	0–900	62.8	16.4
Total	93.8 b	0–300	16.9		3692.5 c	350–18750	973.9		1383.8 a	75–3425	186.8	

[a] Species codes are the same as in Table 2. Total density differed significantly across years. Letters indicate significantly different means by treatment within years. $N = 20$ for all treatments except control ($N = 19$).
[b] QMD is the quadratic mean diameter (cm), i.e. the diameter of the tree with average basal area.

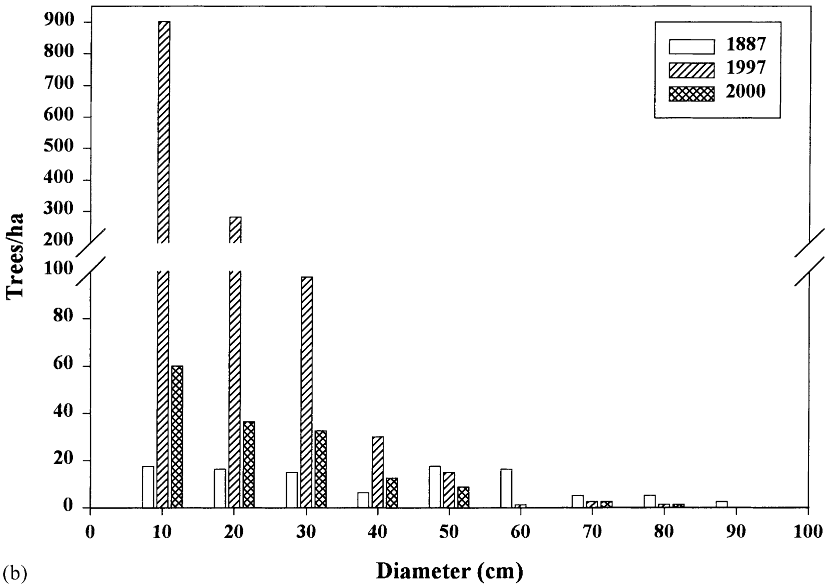
maximum of 7% in tree density and 3% in basal area. In contrast, the FULL treatment resulted in numerical dominance by Gambel oak. Ponderosa pine made up only 28% of post-treatment tree density, although the species remained dominant in basal area (68%). Canopy cover did not differ significantly between treatments in 1997, ranging from 39.5 to 52.7%. After the treatment, the canopy cover in both the FULL and MIN treatments was significantly lower (Table 4).

Tree regeneration density declined substantially in the CONTROL as well as the other treatments (Table 5), but neither the treatment nor time factors were found to be statistically significant. Regeneration in all size classes was dominated by highly variable patches of Gambel oak sprouts, but ponderosa pine seedlings were relatively better represented in taller size classes. Density of regeneration dropped an average of 81% (range 70–86%) in the tallest class and 79% (range 68–88%) in the middle class. Only in the shortest regeneration class was a difference observed among treatments: the CONTROL, MIN, and BURN densities all declined (average 47%, range 34–65%), but the FULL treatment increased by 166% due to a large increase in oak sprouts. Even after the decline observed in 2000, all treatments retained over 1500 seedlings or sprouts per hectare.

No significant differences were found in any crown scorch or bole char variables across the three burned treatments (Table 6). Crown scorch averaged from 3.8 to 4.8 m with maxima 65–142% higher. Scorched crown volume averaged 22.6–29.5%, but maximal values reached 73.8% (MIN) treatment and some trees were apparently killed by fire, as noted above. Minimal and maximal bole char heights ranged from 0.4 to 1.4 m, respectively. Average scorch height was significantly correlated with average scorch percent ($r = 0.71$), minimum char height ($r = 0.67$), and maximum char height ($r = 0.64$). However, pre-burn fuel loadings in any fuel category were not well-correlated with scorch or char (maximum $r = 0.32$).

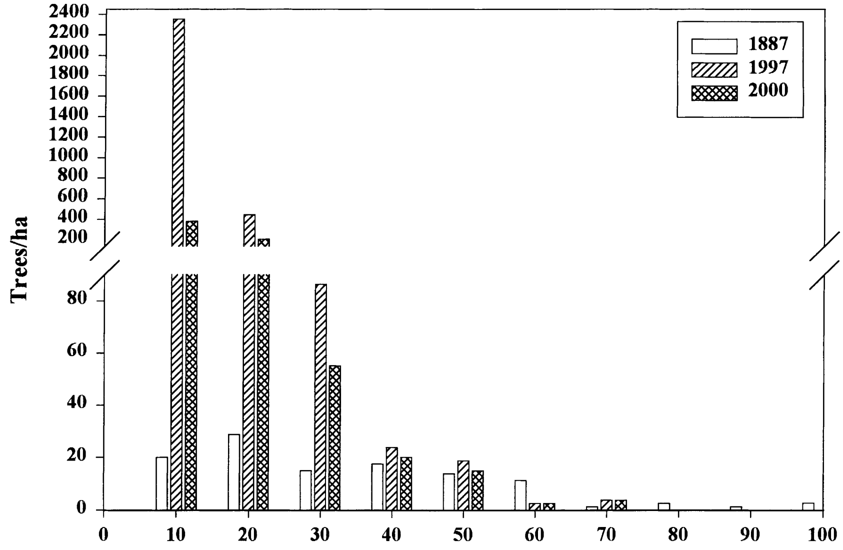


(a)

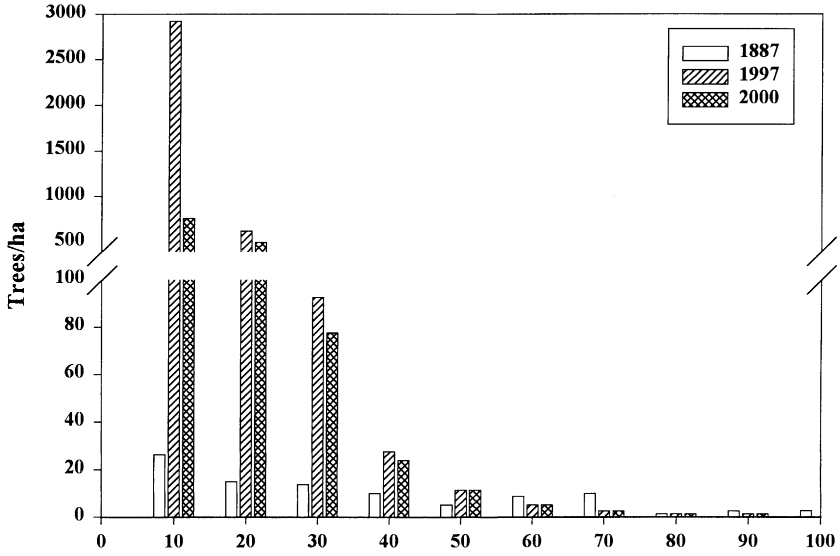


(b)

Fig. 1. Diameter distributions in 1887 (dendroecological reconstruction), 1997 (pre-treatment), and 2000 (post-treatment) in the (a) CONTROL (b) FULL (full restoration), (c) MIN (minimal thinning), and (d) BURN (burn-only) treatment units.



(c)



(d)

Fig. 1. (*Continued*).

Dwarf mistletoe rating (DMR) of ponderosa pine trees, measured only post-treatment, was significantly highest in CONTROL (average 1.5 out of 6). The three treated units had low DMR values (FULL = 0.3, MIN = 0.1, BURN = 0.01) that did not differ significantly.

3.2. Understory vegetation

The predominant trend throughout the understory vegetation data was a substantial decline in species richness and frequency between 1997 (pre-treatment) and 2000 (post-treatment). The declines were consistent

Table 4
Canopy cover (%) before and after treatments[a]

Treatment	Pre-treatment				Post-treatment			
	Mean	Minimum	Maximum	S.E.M.	Mean	Minimum	Maximum	S.E.M.
CONTROL	45.5 a	13.8	80.1	4.5	52.3 a	18.4	81.2	4.2
FULL	39.5 a	10.2	77.7	4.3	24.9 b	0	50.6	3.5
MIN	46.3 a	15.1	68.1	3.0	37.6 b	27.1	79.5	2.8
BURN	52.7 a	7.2	78.9	4.2	53.7 a	9.6	84.8	4.4

[a] Letters identify significantly different means within years.

and statistically significant over time but were inconsistent and rarely significant between treatments, suggesting that the change was due primarily to drought, herbivory, or other non-treatment factors.

Litter and plants were the most frequent ground covers recorded along herbaceous transects (Table 7) in both 1997 and 2000. Post-treatment litter frequency increased between 19 and 37%, in place of the plant cover that was found before treatment. Plant cover decreased between 21 and 34% after treatment. In 2000, bare soil increased (0.1–6.0%) in all treatments except the CONTROL. The increase in bare soil was lower in the MIN treatment (0.1%) than in the FULL treatment (6.0%).

Between 51 and 58 additional species not recorded on line transects were found on belt transects in both 1997 and 2000 (Table 8). Species richness in the CONTROL treatment areas was significantly higher than the other three areas in 1997, before experimentation began, and over all treatments, years, and methods of measurement. The largest 1997–2000 decreases were observed in the BURN treatment (17–32 species) and the CONTROL (15–19 species). The lowest decrease was found in the FULL treatment (5–13 species). SI, a widely used dominance measure of diversity, ranged from 2.0 to 11.8 with averages ranging from a low of 5.63 in the BURN treatment to a high of 5.84 in the MIN treatment prior to treatment (Fig. 2). SI decreased significantly in all treatments from 1997 to 2000. Post-treatment SI ranged from 0 to 0.75 with averages ranging from a low of 0.11 in the BURN treatment to a high of 0.19 in the FULL

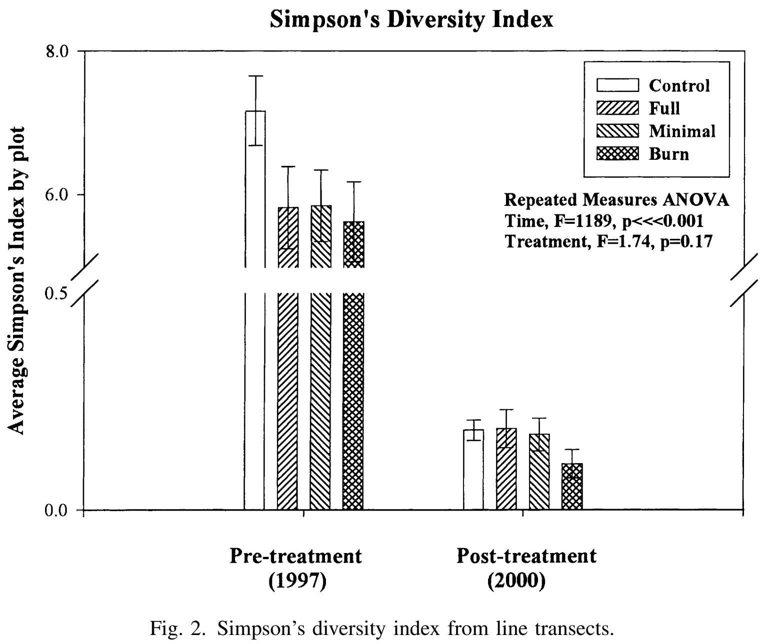


Fig. 2. Simpson's diversity index from line transects.

Table 5
Regeneration density (trees/ha) before and after restoration treatments[a]

Treatment	Pre-treatment total	JUOS	PIED	PIPO	QUGA	Post-treatment total	JUOS	PIED	PIPO	QUGA
Regeneration 0–40 cm in height										
CONTROL	2660 (632) 0–10900	10 (6.9) 0–100	0	25 (25) 0–500	2625 (629) 0–10900	1745 (341) 200–5800	10 (6.9) 0–100	0	65 (26) 0–400	1670 (330) 200–5500
FULL	1580 (280) 0–4500	15 (11) 0–200	10 (6.9) 0–100	30 (15) 0–200	1525 (281) 0–4500	4215 (990) 0–16000	0	5 (5) 0–100	30 (25) 0–500	4180 (984) 0–16000
MIN	2955 (701) 100–11400	125 (47) 0–700	40 (23) 0–400	180 (125) 0–2500	2610 (692) 0–10500	1675 (403) 200–8200	15 (11) 0–200	20 (16) 0–300	15 (11) 0–200	1625 (407) 0–8100
BURN	4180 (802) 600–12300	105 (75) 0–1500	5 (5) 0–100	160 (92) 0–1800	3910 (773) 0–12200	1445 (282) 0–4000	15 (11) 0–200	0	75 (33) 0–600	1355 (286) 0–4000
Regeneration 40–80 cm in height										
CONTROL	290 (106) 0–1600	0	5 (5) 0–100	10 (6.9) 0–100	275 (102) 0–1600	60 (24) 0–400	0	0	35 (17) 0–300	25 (16) 0–300
FULL	830 (289) 0–4300	0	0	45 (20) 0–300	785 (286) 0–4300	95 (43) 0–700	0	0	5 (5) 0–100	90 (43) 0–700
MIN	315 (86) 0–1200	15 (11) 0–200	0	30 (11) 0–100	270 (79) 0–1100	70 (26) 0–300	10 (10) 0–200	0	0	60 (26) 0–300
BURN	330 (58) 0–900	5 (5) 0–100	5 (5) 0–100	125 (41) 0–600	195 (63) 0–900	105 (29) 0–400	0	0	55 (21) 0–300	50 (22) 0–400
Regeneration 80–137 cm in height										
CONTROL	100 (45) 0–600	0	0	10 (6.9) 0–100	90 (45) 0–600	30 (21) 0–400	0	0	10 (6.9) 0–100	20 (20) 0–400
FULL	285 (115) 0–2000	0	0	85 (36) 0–500	200 (102) 0–1900	40 (28) 0–500	0	0	0	40 (28) 0–500
MIN	140 (39) 0–700	15 (15) 0–300	5 (5) 0–100	35 (13) 0–200	85 (41) 0–700	25 (18) 0–300	15 (15) 0–300	0	0	10 (10) 0–200
BURN	230 (59) 0–900	5 (5) 0–100	0	180 (56) 0–900	45 (30) 0–600	35 (11) 0–100	0	0	20 (9) 0–100	15 (8) 0–100

[a] Values are the mean (S.E.) and minimum to maximum range.

Table 6
Crown scorch and bole char following prescribed fire in October 1999[a]

Treatment	Average scorch height (m)	Maximum scorch height (m)	Average scorch (%)	Maximum scorch (%)	Minimum char height (m)	Maximum char height (m)
FULL	4.8 (0.9)	7.9 (1.2)	22.6 (5.1)	47.8 (8.5)	0.8 (0.3)	1.2 (0.4)
MIN	3.8 (0.5)	8.3 (0.7)	29.5 (3.8)	73.8 (6.1)	0.4 (0.1)	1.3 (0.4)
BURN	3.8 (0.7)	9.2 (1.5)	26.8 (5.1)	70.9 (8.5)	0.7 (0.3)	1.4 (0.4)

[a] Values are mean (S.E.). $N = 18$ for FULL treatment (two plots had no trees) and $N = 20$ for MIN and BURN treatments. Maximum values are per-plot averages of maximum measurements. No significant differences were found between treatments for any variable except maximum scorch (%) ($P = 0.049$, no significant pairwise differences).

Table 7
Substrate frequencies from line transects and percentage of increase or decrease from 1997 to 2000

Treatment	1997			2000			Increase/ decrease (%)
	Average frequency (%)	Range (%)	S.E.M.	Average frequency (%)	Range (%)	S.E.M.	
CONTROL							
Plant substrate	43.5	22.9–70.0	2.6	9.8	2.4–15.7	0.9	−33.7
Litter	44.1	22.3–71.7	2.8	81	57.2–94.0	1.8	+36.9
Rock	1.7	0–3.6	0.3	0.6	0–3.6	0.3	−1.1
Soil	8.9	1–30.7	1.8	6	0.6–32.5	1.6	−2.9
Wood	2	0–4.8	0.3	2.5	0–6.0	0.4	+0.5
Scat	0.3	0–1.2	0.2	0.5	0–0.6	0.1	+0.2
Bole	0.6	0.6–0.6	0	0.4	0–0.6	0.2	−0.2
FULL							
Plant substrate	34.5	14.5–48.2	2.2	6.8	1.8–13.3	0.8	−27.7
Litter	48.8	27.1–81.3	3.2	68.2	45.8–86.8	2.6	+19.4
Rock	0.8	0–3.0	0.2	1.3	0–10.2	0.7	+0.5
Soil	14.2	1.2–34.3	2.3	20.2	8.4–36.1	2	+6.0
Wood	1.5	0–6.6	0.4	3.5	0–12.0	0.6	+2.0
Scat	0.5	0–1.2	0.2	0.4	0–0.6	0.1	−0.1
Bole	0.8	0–1.8	0.4	0.3	0–0.6	0.2	−0.5
MIN							
Plant substrate	26.6	6.6–47.0	2.9	5.5	0–14.5	1	−21.1
Litter	59.1	25.9–83.7	3.9	78.2	50.6–94.6	2.1	+19.1
Rock	1.2	0–4.8	0.4	1	0–4.8	0.4	−0.2
Soil	12.2	1.2–33.7	1.8	12.3	3.0–28.9	1.6	+0.1
Wood	1	0–5.4	0.3	2.7	0–7.8	0.5	+1.7
Scat	0.3	0–1.2	0.2	0.4	0–0.6	0.1	+0.1
Bole	0.3	0–1.2	0.1	0.4	0–1.8	0.2	+0.1
BURN							
Plant substrate	31.2	7.8–65.1	3.8	6.5	0–22.2	1.3	−24.7
Litter	57.8	18.7–87.3	4.4	82.2	57.8–96.4	2.3	+24.4
Rock	0.9	0–2.4	0.2	0.7	0–3.0	0.2	−0.2
Soil	7.8	0–21.7	1.4	9.1	0.6–21.7	1.4	+1.3
Wood	2.9	0–10.8	0.8	1.8	0–7.8	0.6	−1.1
Scat	0.5	0–0.6	0.1	0.2	0–1.2	0.2	−0.3
Bole	0.8	0–1.8	0.2	0.7	0–1.8	0.3	−0.1

Table 8
Species richness from point line-intercept transects, belt transects, and combined from both methods, both pre-treatment (1997) and 1-year post-treatment (2000)

Treatment	Year	Species richness		
		Transect	Belt	Transect + belt
CONTROL	1997	52	96	109
	2000	33	81	90
FULL	1997	44	85	100
	2000	31	80	87
MIN	1997	39	79	94
	2000	26	68	77
BURN	1997	52	87	104
	2000	20	70	78

Table 9
Plant nativity from line transects

Treatment	Native (%)		Introduced (%)		Unknown (%)	
	1997	2000	1997	2000	1997	2000
CONTROL	86.1	97.3	13.7	2.7	0.2	0
FULL	94.6	97.3	5.2	2.3	0.2	0.4
MIN	99.5	98.8	0.5	1.2	0	0
BURN	90.4	100	9.6	0	0	0

treatment. Non-native species were highest in the CONTROL plots in both years (1997 and 2000) and non-existent in the post-treatment BURN plots (Table 9). Except in the MIN plots, percent nativity increased post-treatment. For both years and all treatments, *Bromus tectorum* (cheatgrass) was the most frequent non-native species on line transects (up to 7%). On belt transects, the most abundant non-native species recorded were *B. tectorum, Taraxacum officinale* (common dandelion), *Trifolium repens* (white clover), and *Verbascum thapsus* (common mullein).

Community-wide comparisons with NMDS showed differences over time but no differences between treatments either before or after treatment for either plant frequencies or species presence/absence (data not shown). There was very little variation between treatment in burn severity of either substrates (litter, duff, wood) or vegetation. Average burn severity codes fell between 3.07 and 3.95 where "4" denotes unburned.

3.3. Fuels

Surface fuels were similar at all units before treatment (Table 10). Forest floor depth averaged 2.4 cm (range 1.76–3.13 cm). Small woody fuels (1–100H (where 1H denotes 1 h timelag, 2H denotes 2 h timelag, etc.) timeclass, <7.62 cm diameter) averaged 4.30 Mg/ha (range 2.91–5.00 Mg/ha) and large woody fuels (1000H) averaged 10.34 Mg/ha (range 8.16–13.86 Mg/ha). After thinning, forest floor depth increased by 28 and 61% in FULL and MIN. Total woody fuels increased by 360% in FULL and decreased by 62% in MIN. Burning reduced forest floor depth 41–78% and woody fuels by 10–43%. Fuels were relatively stable between the 1999 burn and the 2000 re-measurement. Substantial variability between measurement periods was observed in the untreated CONTROL and standard errors were commonly ≥50% of the means throughout the fuel data.

Variation in crown bulk density (foliage and fine branches) prior to treatments ranged from 0.0501 to 0.0662 kg/m^3 (Table 11), relatively less than the variation in basal area or tree density (Tables 2 and 3). Crown bulk density was reduced 61% in the FULL treatment, 42% in the MIN treatment, and only 19.5% in the BURN treatment (Table 11). Pre-treatment crown fuel loads varied from 6.6 to 10.3 Mg/ha. After thinning and burning, crown fuels were reduced 66% in the FULL treatment, 48% in the MIN treatment, and 27% in the BURN treatment. Changes in the CONTROL crown bulk density and fuel load were <1% in the same period. Crown base heights were not measured before treatment, but they averaged at least 1.3 m higher than the CONTROL after treatment. All three treated units were similar in average crown base height in 2000 (range 0.32 m) but the lowest quintile of crown base height was more variable (range 0.85 m).

3.4. Potential fire behavior

The purpose of the modeling analysis was not to accurately estimate the behavior of a real fire but rather to compare the treatment alternatives. Model results should always be applied cautiously. There are a number of uncertainties in the models integrated in Nexus, reflecting the complexity of fire behavior (Scott, 1998b). Fire behavior models are highly

Table 10
Forest floor depth and woody debris biomass classified by moisture timelag class

Treatment	Litter (cm)	Duff (cm)	Forest floor (cm)	1H (Mg/ha)	10H (Mg/ha)	100H (Mg/ha)	1000H sound (Mg/ha)	1000H rotten (Mg/ha)	Wood < 1000H Mg/ha	Wood > 1000H Mg/ha
Pre-treatment (1997)										
CONTROL										
Mean	0.93	2.19	3.13	0.35	1.79	2.86	10.70	0.16	5.00	10.86
S.E.M.	0.2	0.3	0.5	0.1	0.4	1.2	4.0	0.1	1.3	4.0
FULL										
Mean	0.52	1.49	2.00	0.33	1.30	3.29	6.52	1.65	4.91	8.16
S.E.M.	0.1	0.3	0.3	0.08	0.3	1.6	2.1	0.9	1.6	2.6
MIN										
Mean	0.39	1.37	1.76	0.14	1.38	2.86	13.79	0.07	4.38	13.86
S.E.M.	0.05	0.2	0.3	0.05	0.4	1.7	9.4	0.07	1.9	9.4
BURN										
Mean	0.66	2.09	2.75	0.28	1.48	1.14	8.09	0.37	2.91	8.46
S.E.M.	0.1	0.3	0.3	0.08	0.4	0.8	5.0	0.3	1.0	5.1
Pre-burning (1999)										
FULL										
Mean	2.59	1.41	4.00	0.63	2.57	10.16	30.33	3.31	13.36	33.64
S.E.M.	0.6	0.2	0.7	0.1	0.7	3.1	15.9	1.9	3.7	16.3
MIN										
Mean	1.66	1.17	2.83	0.37	0.92	1.72	3.86	0	3.00	3.86
S.E.M.	0.7	0.2	0.7	0.1	0.3	0.8	1.6	0	1.1	1.6
Post-burning (1999)										
FULL										
Mean	0.50	0.38	0.87	0.20	1.19	6.63	22.93	11.34	8.03	34.27
S.E.M.	0.1	0.1	0.2	0.04	0.3	1.9	14.0	11.3	2.1	18.1
MIN										
Mean	0.68	0.97	1.66	0.14	0.76	2.14	3.19	0	3.04	3.19
S.E.M.	0.4	0.2	0.4	0.03	0.2	0.8	1.4	0	0.8	1.4
BURN										
Mean	0.53	0.99	1.52	0.14	0.57	0.30	2.05	3.46	1.01	5.51
S.E.M.	0.1	0.2	0.3	0.04	0.2	0.3	1.3	3.5	0.4	3.8
One-year post-treatment (2000)										
CONTROL										
Mean	1.13	1.28	2.42	0.29	0.94	1.81	14.39	0.07	3.04	14.46
S.E.M.	0.1	0.3	0.4	0.07	0.3	0.5	6.1	0.07	0.6	6.1
FULL										
Mean	0.66	0.48	1.14	0.34	0.97	5.44	22.08	9.52	6.75	31.59
S.E.M.	0.1	0.08	0.2	0.09	0.3	1.3	14.9	8.9	1.5	17.4
MIN										
Mean	0.67	0.71	1.39	0.25	0.43	2.00	1.48	0.28	2.68	1.76
S.E.M.	0.06	0.08	0.1	0.05	0.09	0.5	0.6	0.3	0.5	0.6
BURN										
Mean	1.01	1.01	2.02	0.27	0.73	0.57	3.96	0	1.57	3.96
S.E.M.	0.1	0.2	0.3	0.05	0.2	0.3	3.6	0	0.5	3.6

Table 11
Crown fuels

	CONTROL	FULL	MIN	BURN
Pre-settlement (1887)				
Crown bulk density (kg/m^3)	0.0338	0.0262	0.0195	0.0196
Average crown base height (m)[a]	4.88	4.88	4.88	4.88
Low quintile crown base height (m)[b]				
Crown fuel load (Mg/ha)	5.117	3.962	2.949	2.967
Stand height (m)	20.0	20.0	20.0	20.0
Pre-treatment (1997)				
Crown bulk density (kg/m^3)	0.0501	0.0531	0.0587	0.0662
Average crown base height (m)[c]	2.32	2.32	2.32	2.32
Low quintile crown base height (m)[c]	1.56	1.56	1.56	1.56
Crown fuel load (Mg/ha)	8.069	6.579	8.045	10.263
Stand height (m)	18.4	14.7	16.0	17.8
Post-treatment (2000)				
Crown bulk density (kg/m^3)	0.0498	0.0206	0.0340	0.0533
Average crown base height (m)	2.32	3.94	3.62	3.82
Low quintile crown base height (m)	1.56	2.37	1.78	2.63
Crown fuel load (Mg/ha)	8.018	2.215	4.208	7.462
Stand height (m)	18.4	14.7	16.0	17.8

[a] Estimated from average crown base height of mature trees in 1997.
[b] No basis for estimation (see text).
[c] Not measured in 1997; values assumed equal to the 2000 control measurements.

sensitive to crown base height, wind speed (or wind reduction factor), fuel moisture, and surface fuel model variables (1H fuel loading, herbaceous fuels, surface area-to-volume ratio, fuel bed depth). We held slope constant at 7% (the average slope of the experimental sites) but similar fuels on steeper slopes would exhibit higher fire intensity. The actual numerical values used for model inputs produced realistic predictions but the behavior of real fires in these stands would be affected by variability in fuels and weather, roads, meadows, surrounding forest fuels, landscape topography, and suppression activities.

Under the modeled conditions (Table 1), all treatments were susceptible to relatively intense fire behavior: flame lengths 7.2–12.3 m, surface fire rate of spread 27.5–35.6 m/min, and heat/area ranging from 12.7 to 21.0 kJ/m^2 (Table 12). The FULL treatment caused a major reduction in potential fire behavior as modeled in the Nexus software. The MIN treatment had an intermediate effect and the BURN treatment was least effective in altering fire characteristics. Crownfire behavior outputs included the torching index (wind speed required to initiate passive crownfire) and the crowning index (wind speed required to sustain active crownfire). Torching is primarily influenced by crown base height and crownfire is primarily influenced by the closely related variables of crown fuel load and crown bulk density (Table 12). Prior to treatment, passive crownfire (torching) was predicted for all sites under the modeled conditions, burning 59–83% of the crown volume, and the crowning index was only 4.4–16.6 km/h higher than the modeled 51 km/h wind speed, indicating that a small increase in wind speed could sustain active crownfire. After treatment, under identical moisture and weather conditions, all sites still supported passive crownfire but the FULL treatment had greatly reduced fire intensity: flame length declined by 80% and crown volume burned dropped by 81%. Flame lengths declined by 57 and 49% and the crown volume burned decreased by 57 and 47% in the MIN and BURN treatments, respectively. The predicted crown volume burned in FULL after treatment was only 11%, in contrast to 30% in MIN and 44% in BURN. Crowning indices increased by 96% in FULL, 48% in MIN, and only by 17% in BURN. After treatment, a wind of

Table 12
Fire behavior outputs using the average pre-treatment fuel loads under the June 97th percentile weather conditions with 51 km/h winds and lowest quintile crown base height[a]

	CONTROL	FULL	MIN	BURN
Pre-settlement (1887)				
Fire type[b]	Surface	Surface	Surface	Surface
Crown percent burned	0	0	0	0
Rate of spread (m/min)	12.1	12.1	12.1	12.1
Heat/area (kJ/m^2)	5.7	5.7	5.7	5.7
Flame length (m)	2.0	2.0	2.0	2.0
Crownfire outputs				
Torching index (km/h)	67.0	67.0	67.0	67.0
Crowning index (km/h)	89.4	107.1	131.9	131.4
Pre-treatment (1997)				
Fire type[b]	Passive	Passive	Passive	Passive
Crown percent burned	54	59	69	83
Rate of spread (m/min)	27.5	28.9	31.6	35.6
Heat/area (kJ/m^2)	13.5	12.7	15.7	21.0
Flame length (m)	7.6	7.2	9.1	12.3
Crownfire outputs				
Torching index (km/h)	23.1	23.1	23.1	23.1
Crowning index (km/h)	67.6	64.8	60.3	55.4
Post-treatment (2000)				
Fire type[b]	Passive	Passive	Passive	Passive
Crown percent burned	53.9	11.3	29.6	43.7
Rate of spread (m/min)	27.4	15.3	20.5	24.5
Heat/area (kJ/m^2)	13.5	6.1	7.9	11.6
Flame length (m)	7.5	2.5	3.9	6.2
Crownfire outputs				
Torching index (km/h)	23.1	34.6	26.3	38.2
Crowning index (km/h)	67.8	126.9	89.1	64.7

[a] Foliar moisture content was held constant at 100%, fire behavior fuel model was 9 (hardwood/long-needled conifer litter), wind reduction factor was 0.3, and slope was 7% (study site average) for all simulations.
[b] Fire types are (1) surface, (2) passive or "torching," and (3) active crownfire.

64.7 km/h could still sustain crownfire in BURN but a wind of 126.9 km/h would be required for active crownfire in FULL.

Simulated fires in the reconstructed pre-settlement stands were projected to remain on the surface (Table 12), but this result is dependent on crown base height, a variable that is difficult to estimate for the reconstructed forest. However, crown bulk density for the reconstructed 1887 forest was low, averaging 0.0248 kg/m^3 (range 0.0195–0.0338 kg/m^3). The maximum crown bulk density in 1887, 0.0338 kg/m^3, was nearly 70% lower than the average 1997 value of 0.0570 kg/m^3 (Table 11). Accordingly, the crownfire indices projected for the pre-settlement stands were high, ranging from 89.4 to 131.9 km/h.

By 2040, 40 years after treatment, FVS simulations showed that retained trees in all four treatments increased in crown fuel load by 52% in CONTROL (12.1 Mg/ha), 88% in FULL (4.2 Mg/ha), 118% in MIN (9.2 Mg/ha), and 82% in BURN (13.6 Mg/ha).

4. Discussion

Causal inferences are limited because the experiment was unreplicated. These circumstances are often encountered in ecosystem experiments, especially in the complex policy and funding environment of research on public lands. Comparisons are meaningful, however, because the treatment alternatives were tested against a control site and because pre-treatment measurements were taken, analogous to the BACI design of Stewart-Oaten et al. (1992). Since the treatments were not subtle, distinct effects associated with each treatment were already apparent in the first-year data. A meta-analysis approach integrating related experimental treatments across the broader region will eventually provide the most complete picture of ecological restoration effects (Arnqvist and Wooster, 1995).

None of the treatments tested here could immediately reverse long-standing ecological degradation, such as the loss of old-growth trees. Rather, they represent differing levels of management intervention aimed at initiating the process of restoring pre-degradation ecosystem characteristics. All require a long-term commitment to prevent future degradation (e.g. overharvesting, overgrazing, excessive recreational use), maintain the surface fire disturbance regime, and continue monitoring and evaluation of the sites. Furthermore, patterns seen in the first growing season following treatment will change over time. Additional mortality is expected to become evident, especially for conifers (Sackett et al., 1996). On the other hand, some trees considered dead in 2000 due to crown scorch may survive (Dieterich, 1979). Especially given the drought conditions in 2000, understory plant response was probably not indicative of future successional trends.

Forest structural conditions were restored most closely within the range of natural variability by the FULL treatment. Tree density and basal area were actually reduced to the low end of the 1887 distribution, nearly one standard deviation below the 1887 mean. Basal area reconstructed in 1887 averaged 13.0 m^2/ha on the FULL, compared to only 6.9 m^2/ha in 2000 (Table 2). Forest density in 2000 was about 150% of the reconstructed 1887 density, but current pine density of 42.5 trees/ha was lower than the historic density (60.0 trees/ha; Table 3). Since the thinning prescription called for retaining all living old-growth trees and multiple replacements for dead trees of pre-settlement origin, the low density of pines was not an intentional outcome of the FULL treatment. Rather, it resulted from the heavy mistletoe infestation encountered in potential replacement trees. The FULL site had a mean of only 35 pine seedlings/ha in 2000. Monitoring will assess whether the regeneration density proves sufficient for maintaining pine in the unit. If not, natural regeneration can be supplemented with planted seedlings of local provenance. The MIN treatment reduced basal area to 13.4 m^2/ha, very close to the historic 12.6 m^2/ha, but tree density of 683.8 trees/ha remained nearly seven standard deviations above the 1887 level (113.8 trees/ha). The mean density in 2000 was more than double the maximum density in 1887. The BURN treatment thinned about 2300 small-diameter trees/ha but the site remained far above historic levels for both basal area and density. Canopy cover in BURN was unaffected by burning (Table 4).

The understory plant community declined consistently in plant cover and species richness from 1997 to 2000 across the spectrum from the undisturbed CONTROL treatment to the mechanically harvested FULL treatment. Even in the first growing season following disturbance, any deleterious understory effects associated with burning, thinning, or equipment operation were not distinguishable against the background of drought effects. Burn severity, crown scorch, and bole char were also essentially the same across all three burned treatments.

Concern over the initial disturbance associated with thinning and burning treatments has led some to call for staged, multiple entries to accomplish restoration (e.g. Southwest Forest Alliance, 1996). In theory, a series of minimally disturbing treatments could reduce undesirable impacts associated with understory damage from machinery and heat injury from burning heavy slash fuels. This hypothesis was not supported by the present study, at least in the first post-treatment year, since no substantive differences were observed in understory response or heat effects across the range of treatments.

Simulation of 40 years of stand growth following treatment was consistent with expectations. Of all treatments, only FULL remained within the pre-settlement range of variability for crown fuel loading in 2040. Individual trees were predicted to be largest in FULL in 2040 (quadratic mean diameter, all species averaged 33.5 cm), followed by CONTROL (24.9 cm), MIN (24.6 cm), and BURN (20.6 cm). It is more difficult to estimate crown bulk density in 2040 since crown base height is unknown. If crown base height and stand height in 2040 were assumed to be 2.3 and 18.4 m, respectively (equal to the 2000 control unit values), then crown bulk density would be: CONTROL 0.0754 kg/m^3, FULL 0.0258 kg/m^3, MIN 0.0568 kg/m^3, and BURN 0.0844 kg/m^3. Again, only the FULL treatment remained within the pre-settlement range of variability for crown bulk density. Under a wildfire scenario similar to that in Table 12, therefore, all the treatments except FULL would be highly susceptible to crownfire by 2040.

Changes over time will affect the treated sites in a variety of ways. Oaks are expected to resprout quickly while pines establish in sporadic favorable years (Savage et al., 1996; Fulé et al., 1997), but repeated burning is expected to limit tree regeneration to safe sites (sensu White, 1985) in all treatments. Future growth of established trees is anticipated to respond to competition, with individual tree growth rates for pines and oaks in FULL likely to be several times higher than in BURN (Kolb et al., 1998; Onkonburi, 1999; Ffolliott et al., 2000). However, increased tree biomass in FULL will be spread over a greater canopy volume as trees grow in height. As long as new regeneration remains regulated by fire, FULL is likely to remain within the range of natural variability in basal area, density, and crown bulk density. Since MIN and BURN are already above the historic levels for these variables even immediately after treatment, they are expected to remain high since repeated surface burning kills relatively few established trees (Sackett et al., 1996).

The economic efficiency of these treatments cannot be fully assessed because of their small scale. However, it is reasonable to assume that even for a larger operation, per-hectare costs would remain highest in FULL due to the greater amount of tree marking, thinning and slash, followed by MIN, with BURN remaining relatively low. The cost ratios of the experimental treatments were approximately 17:13:1 for FULL:MIN:BURN. Costs can also be compared in terms of the relative benefits achieved. For example, if costs are calculated per kilometer increase in crownfire index, the costs per treatment are US$ 12.00/km in FULL, US$ 19.70/km in MIN, and US$ 4.70/km in BURN, a ratio of approximately 2.5:4:1, making FULL more economically favorable than MIN with respect to crownfire hazard. The BURN treatment remained lowest in cost, but the total gain in crownfire index was only 9.3 km/h, not enough to make a meaningful difference in the event of a severe wildfire. A detailed analysis of economic differences in terms of all the possible variables that may be of ecological or social importance is beyond the scope of this paper. Such an assessment would be likely to include factors such as (1) "costs" of differential fire hazard and understory recovery potential following the initial treatment, (2) "costs" of remaining outside the range of natural variability, if restoration to within RNV was a management objective, and (3) costs of future thinning, snag creation, or fuel treatment if the initial MIN or BURN treatment was deemed insufficient.

4.1. Implications for research and management

Each treatment alternative had distinct advantages and disadvantages that may influence management decisions about their broader application. The FULL treatment provided a rapid and effective initial restoration intervention to reduce crownfire hazard and create conditions expected to be suitable for recovery of understory diversity and productivity, with minimal ecological "costs" in terms of soil and understory disturbance. Rapid change to the forest has beneficial effects in terms of immediate crownfire protection, economic efficiency, maximizing the opportunity for understory response, and eliminating the need for future thinning entries and concomitant disturbance. Negative consequences of rapid change could include physiological or environmental shock to residual trees (e.g. sunscald, windthrow). Such effects were not observed in similar thinning treatments near Flagstaff, AZ for either old-growth or second-growth ponderosa pine (Kolb et al., 1998; Feeney et al., 1998). Another concern might be excessively fast alterations to habitat for some wildlife species (Wagner et al., 2000). Effects on mobile species are difficult to measure at the scale of this experiment, although a concurrent study of small mammal communities on the study site is in progress (Chambers, in press). At Mt. Trumbull, AZ, Waltz and Covington (1999) found increased butterfly abundance and diversity following treatments similar to FULL. Finally, the FULL treatment required road access and use of heavy machinery. These factors weigh against FULL in roadless areas, wilderness, or sites where machinery and noise are undesirable (e.g. noise disturbance to Mexican spotted owls during the breeding season). Depending on the level of risk from crownfire, temporary disturbance may be deemed an acceptable tradeoff for improved fire protection and economic efficiency.

Of the two agencies involved in this study, virtually all of the ponderosa pine/Gambel oak forestlands of the Kaibab National Forest are practically and legally accessible for the FULL treatment. The value of treatment by-products such as logs or chips could be used to defray costs. However, the ecosystem management mandate of the Forest Service does not necessarily center on restoration of natural habitats (Kaufmann et al., 1994). In contrast, GCNP has good road access only on the South Rim. The North Rim, where the bulk of forestlands occur, has few roads and is proposed for wilderness designation. Deriving value for treatment by-products, whether through sale or trade for treatment work, is against current policy. But the park does have a mandate to ''conserve the scenery and the natural and historic objects and the wild life therein ... by such means as will leave them unimpaired ...'' (Sellars, 1997, p. 38), a directive consistent with ecological restoration (Moore et al., 1999). Initial application of a treatment like FULL within the park could include protection of developments and restoration along roads or boundaries, creating protected buffers for dispersed non-mechanized thinning and prescribed burn blocks.

The BURN treatment killed many small-diameter trees, reduced fuels, and raised crown base height. However, basal area, canopy cover, and crownfire hazard were not greatly reduced because the majority of trees >20 cm in diameter survived. Any differences in tree mortality due to not raking accumulated fuels away from tree boles will probably take several years to become apparent (Sackett et al., 1996) or may not be significant (Kaufmann, 2000). Excessive rapidity of forest change is not a concern for BURN. Instead, the fundamental issue is that burning alone may be insufficient to restore forests conditions to the natural range of variability. Process modeling of long-term forest change under repeated burning has led to varying results, depending in large part on model assumptions about fire behavior and fire-caused mortality. For example, Miller and Urban (2000) suggested that relatively intense prescribed fires could restore forest structure over several centuries in the Sierra Nevada, while a model applied by Covington et al. (in press) indicated that an Arizona forest remained dense indefinitely under a prescribed fire regime. The present study cannot resolve the question, but it does show that the BURN treatment site remained dense and vulnerable to crownfire after initial burning. Since the canopy cover and large tree structures were relatively unchanged, the potential for density-dependent growth declines (Biondi, 1996) and mortality (Mast et al., 1999) of old-growth trees will probably continue and prospects for future understory recovery appear limited. Continued application of the BURN treatment on both park and forest lands appears likely.

The fact that the MIN treatment was intermediate to FULL and BURN in almost every respect suggests that the treatments fall along a continuum of forest conditions rather than representing three qualitatively different environments. The implication of continuity is that the properties of *other* intermediate treatments, not tested in this experiment, could reasonably be predicted based on the effects observed here. This hypothesis awaits testing at this and other sites over time. While the MIN site remained outside the range of natural variability in tree structure, substantial gains in reducing canopy closure and crownfire hazard were achieved. Since the thinning was centered on old-growth trees, the fire protection may be disproportionately valuable in terms of allowing old trees to survive wildfire or intense prescribed burning. The MIN treatment could be carried out with machinery but does not require mechanized equipment, making it more useful in the park's unroaded lands and forests managed as proposed wilderness.

Extending from this initial experiment, we recommend that Kaibab National Forest and GCNP consider applying treatments similar to FULL and MIN, with appropriate site-specific modifications, to additional sites. A first step would be implementation of the companion experimental blocks within the park. Another useful approach would be to apply either the FULL or MIN treatments in sites currently being thinned to reduce fire hazards around developed areas of the park. Over time, variants of these treatments or similar ones being tested in southwestern forests (Covington et al., 1999; McIver et al., 2001) could be combined to create landscape-scale burn blocks buffered by restored edges, within which fire could be applied consistently and safely. For example, the concept of defensible shaded fuelbreaks accompanied by area treatments outlined by Agee et al. (2000) could be applied in the park by using a FULL-type treatment around developments and along roads and borders, with MIN- and BURN-type treatments applied to larger adjacent areas. On the forest, large-scale thinning could be pursued more rapidly and economically, protecting resources on both sides averaged 33.5 cm of the boundary.

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