Diversity in Ponderosa Pine Forest Structure Following Ecological Restoration Treatments

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ABSTRACT. We tested the effectiveness of ponderosa pine forest restoration by comparing forest restoration treatments to untreated forest and to reconstructed forest structure in 1870 (date of Euro-American settlement) using an experimental block design at the Grand Canyon-Parashant National Monument in northwestern Arizona. Forest tree density averaged more than 20 times the historical tree density, and basal area was 4 to 6 times higher in contemporary forests than in historical forests. Restoration treatments consisted of thinning young trees to emulate the forest density, tree composition, and spatial distribution in 1870, followed by prescribed burning. Following restoration treatment, tree density was significantly reduced but remained 6 times higher than historical forests. Basal area in restored forests was still 2.5 times greater than reconstructed basal area values. Ponderosa pine dominance changed little from pretreatment data across the four blocks, averaging 60% of stems and 87% of the basal area prior to treatment and 56% of stems and 85% of the basal area following treatment. Ninety-eight percent of contemporary forest trees were less than 100 yr old prior to restoration treatment; this proportion was reduced to 82% following treatment. Restoration treatment also significantly reduced canopy cover and increased total tree regeneration. However, treatment effects on forest fuels were highly variable. Litter and duff fuel layers were significantly reduced by prescribed fire but woody debris increased. Overall forest structural diversity following treatment implies that fire behavior, wildlife habitats, and other ecological attributes will vary relatively widely in the future landscape. FOR. SCI. 49(6):885–900.

Key Words: Canopy cover, fuels, *Pinus ponderosa*, presettlement, reference conditions, regeneration.

FEW LANDSCAPE-LEVEL RESTORATION PROJECTS can test the effectiveness of treatments because these projects are difficult to replicate or implement with appropriate statistical design (Hurlbert 1984, Michener 1997, Block et al. 2001). We designed a ponderosa pine restoration project with replicated experimental blocks to examine treatments with greater statistical inference. This research design provides both the size (16–40 ha units) and replication (n = 4) needed to assess the impacts of treatments on several ecosystem variables through time (Eberhardt and Thomas 1991). In this article, we specifically examined changes in forest structural attributes 1 yr following restoration treatments at Mt. Trumbull in the Grand Canyon-Parashant National Monument, northwest-

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ern Arizona, and compared these conditions to historical forest structure.

Restoration treatments are designed to emulate reference conditions, based on the range of natural (or "historical") variability (Aronson et al. 1993, Landres et al. 1999). When existing reference systems are not available, historical records, surveys, or paleoecological techniques can be used to estimate reference conditions for certain ecosystem attributes (Swetnam et al. 1999, Minnich et al. 2000). Although it is difficult to find undisturbed forests in the U.S. Southwest, reference conditions for forest structural attributes are uniquely available because of the relatively recent time of Euro-American settlement (less than 150 yr ago) combined with the low rates of woody material decay (Jenny et al. 1949). Ponderosa pine forest structural changes through time have been assessed using dendrochronological techniques across the Southwest by various authors (White 1985, Covington and Moore 1994, Fulé et al. 1997). These data and evidence from relict sites, historical records, and photographs facilitate the development of site-specific reference conditions that provide a broad range of historical variability in tree densities, tree composition, and tree age structure (Covington et al. 1994, Moore et al. 1999, Allen 2002, Fulé et al. 2002a).

Forest structural changes occurred across the arid Southwest as a result of early land use policies associated with Euro-American settlement. For example, at Mt. Trumbull, Euro-American settlers from southern Utah began to utilize the area for grazing and logging purposes in 1870 (Altschul and Fairley 1989). Fire records from fire-scarred trees suggest frequent fires crossed the landscape every 4 to 7 yr until 1870 (Fulé, unpublished data). In many areas of the Southwest, large herds of introduced livestock eliminated the fine fuels (grasses and forbs) that carried frequent surface fires (Weaver 1951, Cooper 1960, Swetnam and Baisan 1996). Along with the cessation of fire, continued grazing and removal of old-growth trees facilitated ponderosa pine tree irruptions (Cooper 1960, White 1985, Savage et al. 1996).

These ecosystem structural changes have reduced understory biodiversity and altered important ecosystem functions, such as decomposition, regeneration of trees, shrubs, and herbaceous plants, and nutrient cycling (Covington et al. 1997). In addition, changes in tree densities have altered fire regimes from frequent, low-intensity understory fires to infrequent, stand-replacing fires. The total acreage burned by wildfires in ponderosa pine forests increased exponentially from 12,000 ac per year in the 1940s to over 300,000 per year in the 1990s (Ellington 2001). Although ecological changes alone warrant experimentation with restoration treatments, the threat of uncontrollable wildfires to western communities and private property combined with the ballooning cost of fire fighting has also created a political impetus for restoration treatments (Appropriations Bill 2001).

Experimentation with re-introduction of surface fire suggests that this process alone is not enough to restore pre-1870 ecosystem function, and may further damage the remaining old-growth trees (Harrington and Sackett 1992, Sackett et al. 1996). Restoration treatments incorporating overstory tree thinning and returning fire to the land have been experimentally implemented on small scales (Covington et al. 1997, Lynch et al. 2000), but the Mt. Trumbull restoration project is the first to examine forest structure, understory re-establishment and other ecosystem responses on a landscape scale to thinning and burning treatments. Landscape-level experiments with replications, such as this study, are needed to provide land managers scientifically based land management alternatives (Eberhardt and Thomas 1991, Michner 1997).

Four experimental blocks were established in 1997. We compared forest structure as reconstructed to the time of fire exclusion (1870), to contemporary stand structure prior to any treatments (1997-1998), and finally to posttreatment stand structure (2000). To examine forest structural changes between pre-1870, contemporary, and restored forests, we asked: (1) How did forest structure, including tree density, composition, and basal area, change following fire cessation associated with the introduction of grazing and logging? (2) Did ponderosa pine restoration treatments return forest structure to pre-1870 conditions? (3) What effects did restoration treatments have on forest age structure, tree regeneration, canopy cover, and forest fuels? (4) What are the implications of forest structural diversity following treatment?

Methods

Study Site

The study site was a ponderosa pine (*Pinus ponderosa*) and Gambel oak (*Quercus gambelii*) forest near Mt. Trumbull in the Uinkaret Mountains, about 35 km north of the Grand Canyon on the Arizona Strip (northwestern Arizona north of the Colorado River). The site is managed by the Bureau of Land Management (BLM) in the Grand Canyon-Parashant National Monument. Elevation of the site ranged from 1,675 m to 2,620 m. The area received an average of 40–45 cm of rainfall per year, and contained some of the biota of the Great Basin (Welsh et al. 1993), in addition to the flora of northern Arizona (Kearney and Peebles 1951). The forest was predominantly ponderosa pine, although Gambel oak composed 15% of the overstory (Waltz and Fulé 1998). Other tree species in the area included New Mexico locust (*Robinia neomexicana*), quaking aspen (*Populus tremuloides*), pinyon (*Pinus edulis*) and Utah juniper (*Juniperus osteospermus*). The understory plant community was dominated by big sagebrush (*Artemisia tridentata*) and showed evidence of invasion by nonnative species such as cheatgrass (*Bromus tectorum*). Although over 300 herbaceous species were documented at the Mt. Trumbull site in the last 5 yr (J.D. Springer, pers. comm.), the forest floor cover prior to restoration treatments was 70% litter and duff, with only 15% of the cover represented by live plants (Waltz and Fulé 1998).

Four experimental blocks were established (Figure 1). The sites were chosen to block for landscape heterogeneity, located in areas representing the diversity in vegetation, topography and land-use history. Each block was divided into two equal units, randomly assigned to control or restoration treatment. The four experimental blocks varied in soil type, forest density, forest composition and herbaceous cover. **Block 1** (EB1) was located on shallow lava/cinder soils with low tree density. Tree composition consisted almost entirely

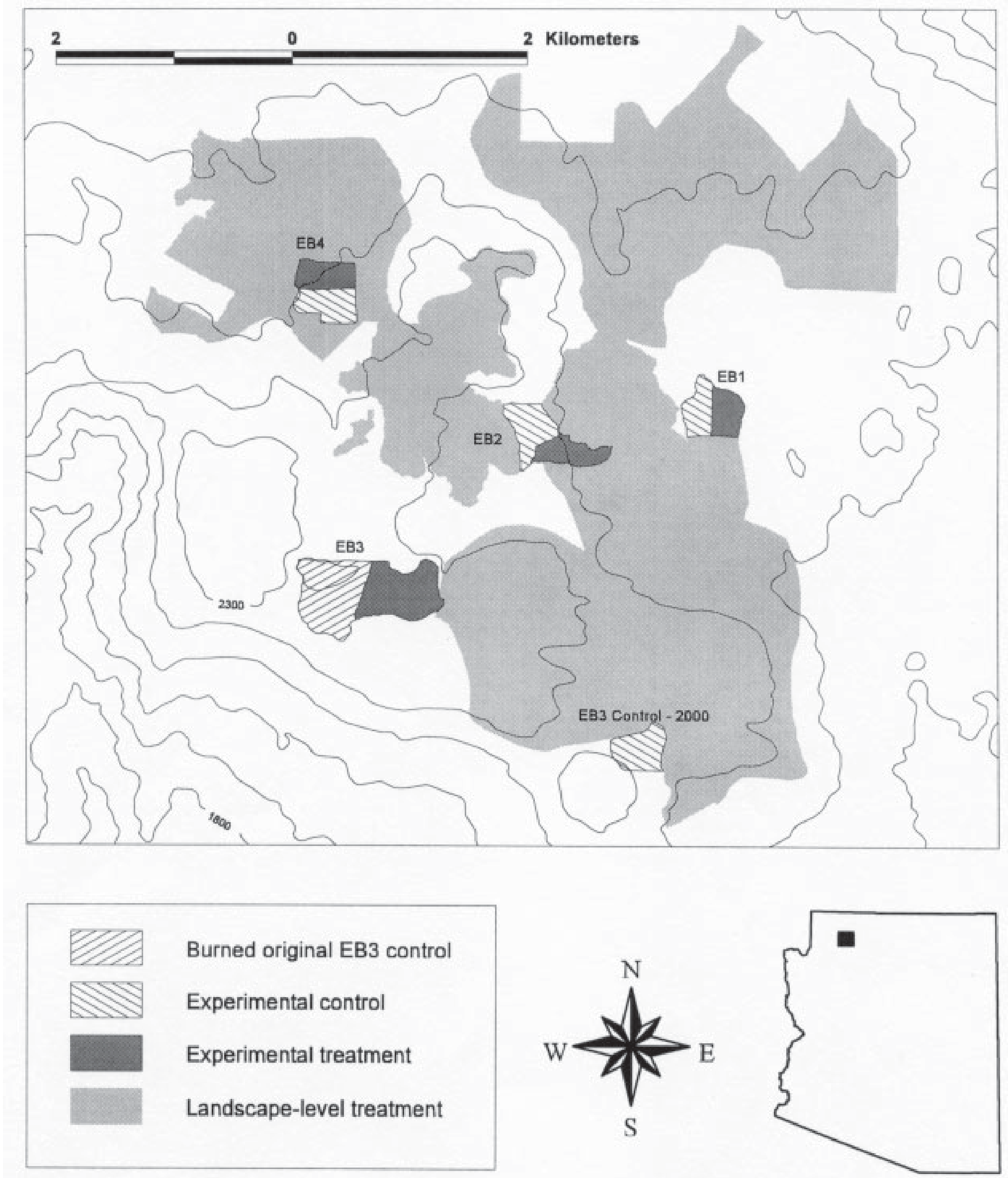

Figure 1. Map of Mt. Trumbull forest restoration project, within the Grand Canyon-Parashant National Monument. Experimental blocks (EBs) 1–4 were established in 1997–1998, adjacent to an ongoing landscape-level restoration experiment. Blocks were divided and each half randomly assigned the control or treatment unit. The 1997–1998 Control unit of EB3 was lost to wildfire in spring 2000, so EB3 Control 2000 was established May 2000 to serve as a new control for the treatment unit.

of ponderosa pine, with many living pre-1870 trees. This block had the highest shrub density, composed of primarily serviceberry and wax currant. **Block 2** (EB2) had dense Gambel oak and an abundant herbaceous understory, including sagebrush openings, on basalt soils, representing the eastern portion of the research site. At the highest elevation, **Block 3** (EB3) was dominated by large, pre-1870 ponderosa pine, but also had Gambel oak and New Mexican locust throughout the unit. This block was the largest (~ 40 ha) and was predominantly on cinder soils. **Block 4** (EB4) represented the densest stands of ponderosa pine forest at Mt. Trumbull. It consisted of dense ponderosa poles, some oak clumps, and some pinyon and juniper at its western end. This block was on basalt soils and had few living pre-1870 ponderosa pines.

Experimental Block Monitoring Plots

Experimental blocks differed in size (~16 to 40 ha) but monitoring plots were established on a consistent 7.2 ha grid in each unit. In 1997 and 1998, 20 monitoring plots in each treatment unit (total 40 plots per block) were established and sampled in all 4 experimental blocks. Plot centers were located on a 60 m grid. The standard monitoring plot was a 0.04 ha circle (diameter = 22.56 m), with a permanent iron center marker to serve as a long-term monitoring unit. Plots were corrected for slope in the field.

All trees greater than breast height (1.37 m) were measured on the entire plot (400 m^2). Each tree was tagged with an aluminum label at breast height (1.37) and the following data were recorded: diameter at breast height, crown code (dominant, codominant, intermediate, or subcanopy), damage, and condition class [(1) live; (2) declining; (3) recent snag; (4) loose bark snag; (5) clean snag; (6) snag broken above breast height; (7) snag broken below breast height; (8) downed dead tree; (9) cut stump; and (10) stump hole]. The condition class categories were derived from snag decomposition studies (Thomas et al. 1979). All living and dead trees potentially old and/or large enough to have become established prior to 1870 were identified as potentially pre-1870 trees in the field. Ponderosa pines with dbh > 37.5 cm or ponderosa of any size with yellowed bark (White 1985), as well as all oaks, junipers, and pinyon trees > 17 cm at dbh (Barger and Ffolliott 1972) were considered potentially pre-1870 trees. All living potentially pre-1870 trees and a random 10% of all postsettlement live trees were cored for determination of age and past size. Overstory canopy cover was measured every 30 cm along one 50 m transect by the point intercept method (Ganey and Block 1994).

Tree regeneration, defined as trees below breast height (1.37 m), were tallied by species, condition, and height class in a nested 100 m^2 circular plot (radius = 5.64 m). Species, condition, and height class (<40 cm; 40.1–80 cm; 80.1–137 cm) were recorded for each seedling. Fuel loadings were calculated along a 15.2 m planar transect using methods in Brown (1974) and constants from Sackett (1980).

Restoration Treatments

Thinning treatments were implemented in 1999 and 2000. The thinning design was based on the pre-1870 pattern of tree species composition and spatial arrangement (Covington et al. 1997, Mast et al. 1999). Living pre-1870 trees of all species were retained. In addition, wherever evidence of remnant pre-1870 material was encountered (snags, stumps, logs), several of the largest post-1870 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., 3 replacements per every 2 remnants) if the replacements were 40.6 cm diameter at breast height (dbh) or larger, otherwise each remnant was replaced with 3 trees. This procedure was intended to leave 150–300% more trees than in 1870, to account for the smaller biomass contributed by smaller diameter trees, potential errors in reconstructions, and to allow for un-predicted mortality due to restoration 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).

Although thinning followed the same design across blocks, operational differences resulted in treatment differences in each of the blocks. Block 1 had low tree densities (around 550 trees/ha), allowing thinning to be done entirely with hand tools (chainsaws) by a BLM crew in the spring of 1999. Felled wood was left on site, slash, litter and duff were removed from around the base of old-growth trees, and the site was burned in the fall of 1999. Blocks 2 and 3 were thinned by a commercial operator as part of a timber sale. Logs greater than 12.4 cm dbh were removed from the site as merchantable wood. Smaller trees were thinned by the BLM crew. Blocks 2 and 3 were both burned in the spring of 2000. In April 2000, a smoking stump in an adjacent prescribed burn ignited a wildfire in the control unit of EB3. The intense wildfire eliminated the unit as a valid untreated control for the block. In the summer of 2000 we initiated a new paired control (Figure 1; referred to as "EB 3 Control 2000" throughout text, tables and figures). No pretreatment data (1997–1998) were available for the control 2000, but tree densities and the herbaceous community were similar to the block 3 treatment unit prior to treatment.

Block 4 had the highest tree density (~2500 trees/ha) and accessibility difficulties that created operational problems with the treatment implementation. Trees were hand-felled, and the majority of the wood was cut and left on site. With such high tree densities, trees were not thinned to the marked prescription, but rather to a fuel loading maximum. The resulting residual stand following thinning treatments was denser than intended. A number of the remaining trees were killed during the prescribed burn in spring 2000.

Plot Remeasurement

All plot measurements were resampled the season following prescribed burn (2000). Tree tags and original maps were used to relocate and retag trees, if necessary. Tree remeasurements included tree condition, scorch, and char heights. A new condition class was established for trees consumed by fire (cond. 11) and trees previously dead (cond. 3 or above in 1998), that were unable to be relocated (cond. 12). No tree diameters were remeasured because of minimal growth changes from year to year, although negative growth was possible as a result of fire-caused bark loss. We also sampled tree regeneration, forest fuels, and canopy cover.

In EB 1, six control plots were burned when a burning snag fell across fire lines into the control area. No trees were lost to fire, but because the low intensity burn altered forest fuels and understory, these plots were excluded from posttreatment fuel loading statistical comparisons. Block 2 control also had one plot lost to fire from an adjacent unit which was also excluded from fuel analyses.

Analysis

Tree increment cores were surfaced and cross-dated (Stokes and Smiley 1968) with local tree-ring chronologies. Rings were counted on cores that could not be cross-dated, usually on young 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. The field determination of pre-1870 or post-1870 tree status was confirmed or rejected using the age data. Ten out of 1,211 trees (0.8%) that were designated as post-1870 in the field actually predated 1870.

Forest structure was reconstructed in 1870 following dendroecological methods described in detail by Fulé et al. (1997) and Mast et al. (1999). 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- and site-specific predictive regression relationships between diameter and basal area increment for pre-1870 ponderosa pine, Gambel oak, and pinyon trees. For Utah juniper, published diameter-dependent growth regression equations were applied to determine pre-1870 diameter (Barger and Ffolliott 1972). An analogous process of growth estimation was used to estimate the past diameter of the small proportion of living pre-1870 trees for which an intact increment core could not be extracted due to rot. New Mexico locust, a short-lived species, was not suitable for reconstruction. Accuracy of the dendroecological reconstruction techniques is discussed below (see Structural Changes since 1870).

The harvest date at Mt. Trumbull for large stumps (diameter ≥40 cm) was assumed to be 1885, corresponding to the historic timber cutting period in the Mt. Trumbull area (Altschul and Fairley 1989). The date for smaller stumps was assumed to be 1974, corresponding to more recent thinning under Forest Service and BLM management.

This experimental design allowed us to use each block as a replicate, with a total sample size (n) of 4. We used an alpha level of 0.10 to denote significant differences because of the small sample size. Comparisons of forest structural attributes (tree density and tree basal area) between 1870 forests and contemporary forests (1998) were analyzed with a blocked 2-way ANOVA. Reconstructed forests were not independent of contemporary forests, as reconstructions were accomplished using the same forest plots that measured contemporary plots. The 2-way ANOVA used for such paired comparisons is identical to a paired t-test, but it allows additional evaluation of the variability within the blocking factor (Sokal and Rohlf 1994). To determine any pretreatment differences between control and treatment units, within year (1870 and 1998) differences between control and treatment units within blocks were examined with a 2-way ANOVA. The effects of the restoration treatment were analyzed within treatment year 2000 with a 2-way ANOVA. Finally, post-treatment (2000) forest structural variables were compared to pre-1870 forest variables to assess the effectiveness of restoration treatments with a 2-way ANOVA.

Canopy cover was analyzed for differences between control and treatment units in contemporary forests pretreatment (1997–1998) and posttreatment (2000) with a 2-way ANOVA. Tree regeneration was variable among species within blocks, due to differences in tree composition. Total tree regeneration was analyzed with a Wilcoxin signed ranks test for differences between control and treated units prior to (1997–1998) and after (2000) restoration treatment. Ponderosa pine was the only tree species occurring in all four blocks, so regeneration of this species was specifically examined for differences between control and treated forests with a Wilcoxin signed ranks test.

Fuel loadings were separated into forest floor depths by litter and duff, and woody fuel loadings by size classes. Litter and duff depths were analyzed for differences between control and treatment units before (1997–1998) and after (2000) restoration treatments with a MANOVA, followed by univariate tests. The larger size classes of woody fuel were analyzed with a MANOVA, before and after restoration treatments. The variability between blocks may have masked patterns occurring within blocks. Therefore, all data were additionally presented on a per-block basis in tables and figures, to help clarify initial patterns.

Results

Pre-1870 and Contemporary Forest Structure —Changes Through Time

Forest structure at the time of the last widespread fire (circa 1870) was relatively open (61.5 trees/ha) and was significantly less dense than contemporary forests at the time of our initial survey (~1,370 trees/ha in 1997–1998, Figure 2, 2-way ANOVA $F = 38.5$, $P < 0.001$, $n = 4$). Basal area values significantly increased 400%—600% from the reconstructed 1870 forest to the 1998 survey (Table 1, 2-way ANOVA $F = 81.2$, $P < 0.001$, $n = 4$). Pre-1870 basal area ranged from 4 to 13 m^2/ha. There were no significant differences in tree density or basal area between control and treatment units in reconstructed 1870 forest data (tree density 2-way ANOVA $F = 0.19$, $P = 0.694$; basal area 2-way ANOVA $F = 0.29$, $P = 0.630$), suggesting that the treatment units within blocks were similar prior to fire suppression.

Species composition in the 1870 forest was similar to the contemporary forest in three of the four blocks. However, in the 1870 stand of EB4, *Juniperus osteosperma* contributed one-third of the stand basal area and almost half the stems across both the control and treatment units (Tables 1 and 2). These data suggest that this block changed from a ponderosa/ juniper ecotone in 1870 to an almost pure ponderosa pine forest by 1998. The ratio of oak:pine stems stayed similar over time across the experimental blocks, as both species increased since fire exclusion. Other compositional change observed in all blocks included the general increase in the deciduous tree, *Robinia neomexicana*. This tree is an excellent sprouter and in the absence of fire can develop clumped stands (Gottfried 1980).

Restoration Treatment Effects on Forest Structure

Thinning significantly reduced contemporary tree density by an average of 77%, ranging from 68% of stems removed in EB2 to 85% of stems removed in EB3 (Figure 2, year 2000). Basal area was decreased by 48% on average, with a range of 35% removed in EB1 to 58% removed in EB3 (Table 1, year 2000). However, posttreatment tree density and basal area were still 6 times and 2.5 times significantly greater, respectively, than the pre-1870 reconstructed tree density and basal area (Figure 2, trees/ha 2-way ANOVA $F = 12.5$, $P = 0.038$; Table 1, basal area 2-way ANOVA $F = 22.0$, $P = 0.018$). The treated blocks had between 2 and 15 times as many tree stems as the reconstructed, pre-1870 stands.

Tree species composition appeared to change little from pre-1870 and 1997-98 composition proportions following restoration treatment (Figure 2), although this was not tested statistically due to high variability among blocks. The treatment removed only ponderosa pine trees. However, nonpine tree mortality did occur with the prescribed fire. Gambel oak and New Mexico locust displayed opposite trends following

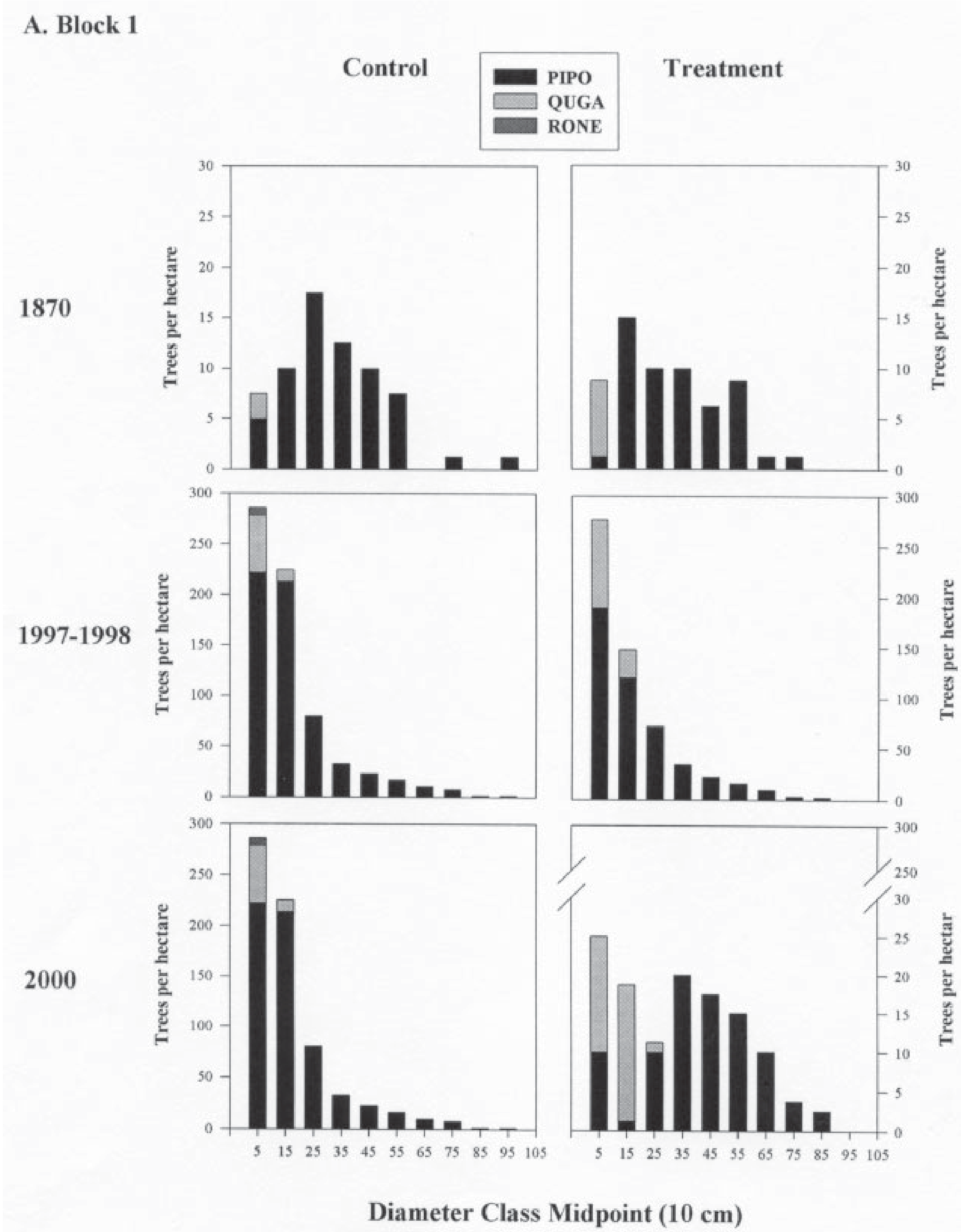

Figure 2a. Diameter distributions of overstory tree structure for each experimental block—treatment unit by time period. Trees per hectare were tallied within 10 cm diameter classes; the x-axis represents the midpoint of the diameter class. The 1870 diameter distributions were based on reconstructed forests, 1997–1998 distributions from pretreatment data, and 2000 diameter distributions are following treatment. Changes in control units from 1998 to 2000 represent naturally occurring mortality. A. EB1; B. EB2; C. EB3; D. EB4.

restoration treatments: Gambel oak showed increasing proportions, while New Mexico locust appeared to decrease following thinning and burning treatments. For example, proportions of Gambel oak ranged from 6% to 29% of the contemporary stems, averaging 18% across the four blocks in 1998. Following treatment, oak averaged 25% of stems in the treatment units. New Mexico locust shifted from 18% of stems in 1998 to 13.5% of stems in 2000. Both responses were trends only and were not tested statistically.

The stem density and basal area reduction posttreatment is also represented by a shift in the diameter distributions of the units following treatment (Figure 2a–d). Figure 2 depicts diameter distributions of control and treatment units in reconstructed pre-1870 stands (1870), contemporary stands prior to treatment (1998), and contemporary stands following treatment (2000). In 1870, the ponderosa pine diameter distribution resembled a broadly distributed curve with most trees in the middle of the diameter range. However, in 1998, approximately 40% of living trees were between 0 and 5 cm dbh. Almost 75% of living ponderosa pine trees were below 25 cm dbh [note scale change between reconstructed forests (1870) and contemporary forests (1998), Figure 2]. Following treatment (2000 treatment units), diameters were distributed in a manner more comparable to the reconstructed

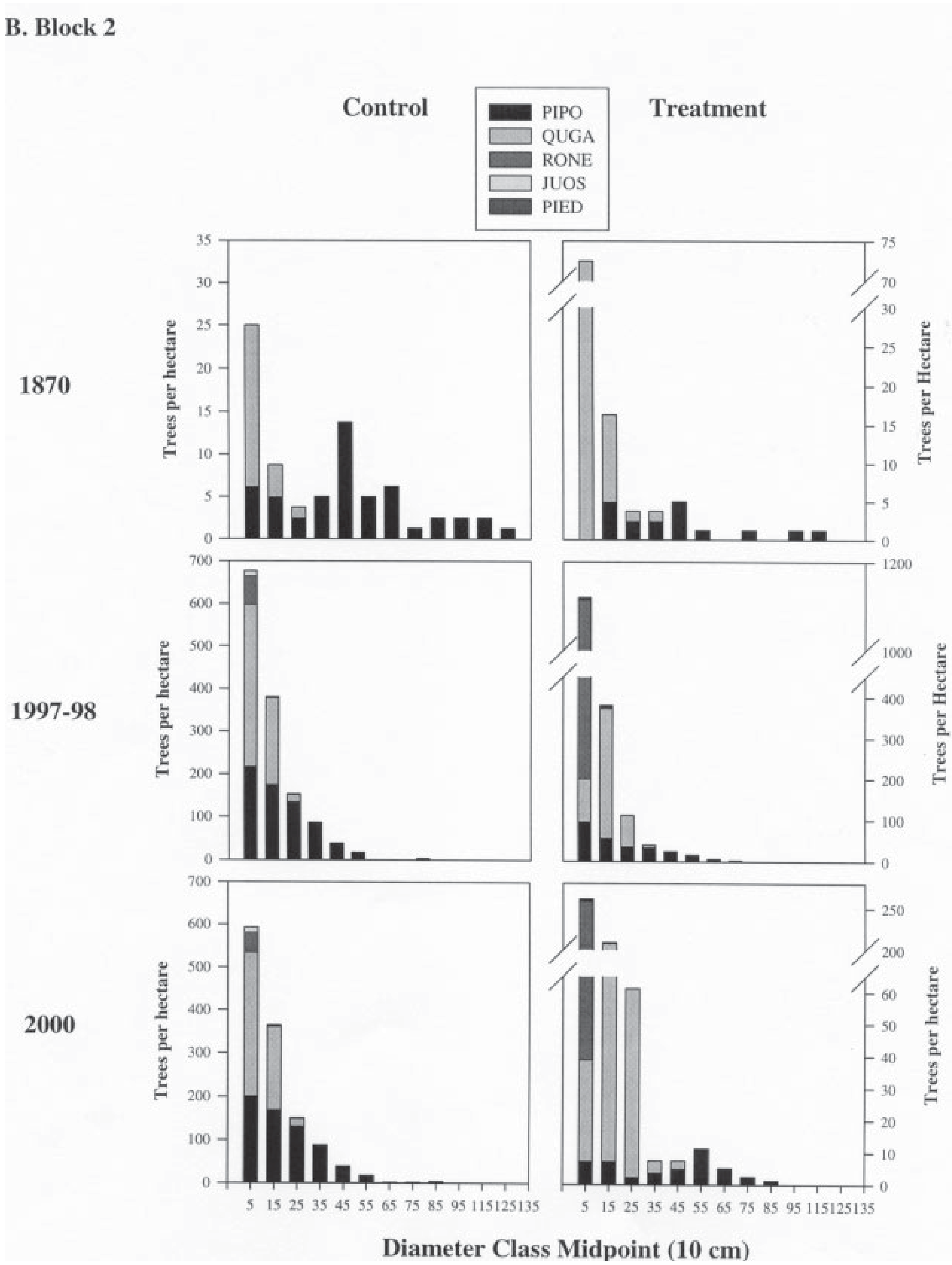

Figure 2b.

(1870) diameter distribution, although total trees per hectare was higher. Ninety percent of the ponderosa pines thinned was taken from diameter classes less than 35 cm at breast height.

Restoration Treatment Effects on Contemporary Forests

Contemporary forests age distributions for each control and treatment unit within each block are shown in Figure 3 (1997–1998). Although there was variation between blocks in age distributions (note graph scales), 94%–99% of ponderosa pine trees had established since the time of the last widespread fire, with center dates later than 1870. The ponderosa pine regeneration pulse was greatly reduced in the 1950 age class (1940–1960); a much smaller proportion of trees established in these decades. The youngest trees were oaks and locust, although a pulse of oak regeneration was also observed in the 1870 age class (1860–1880). This pulse is most likely due to the cessation of fires, which allowed the sprouting oak to reach tree stature. These age distributions are comparable to other sites in the Southwest and Mexico (Fulé et al. 1997, Mast et al. 1999).

Following treatment (2000), the age distribution was altered by the removal of large proportions of the post-1870 establishment classes (Figure 3, last column, treatment units only, note scale change). Although the posttreatment age distributions are still skewed towards younger trees, the proportion of young trees was reduced. Trees with center dates later than 1870 comprised ~70% of the trees in blocks

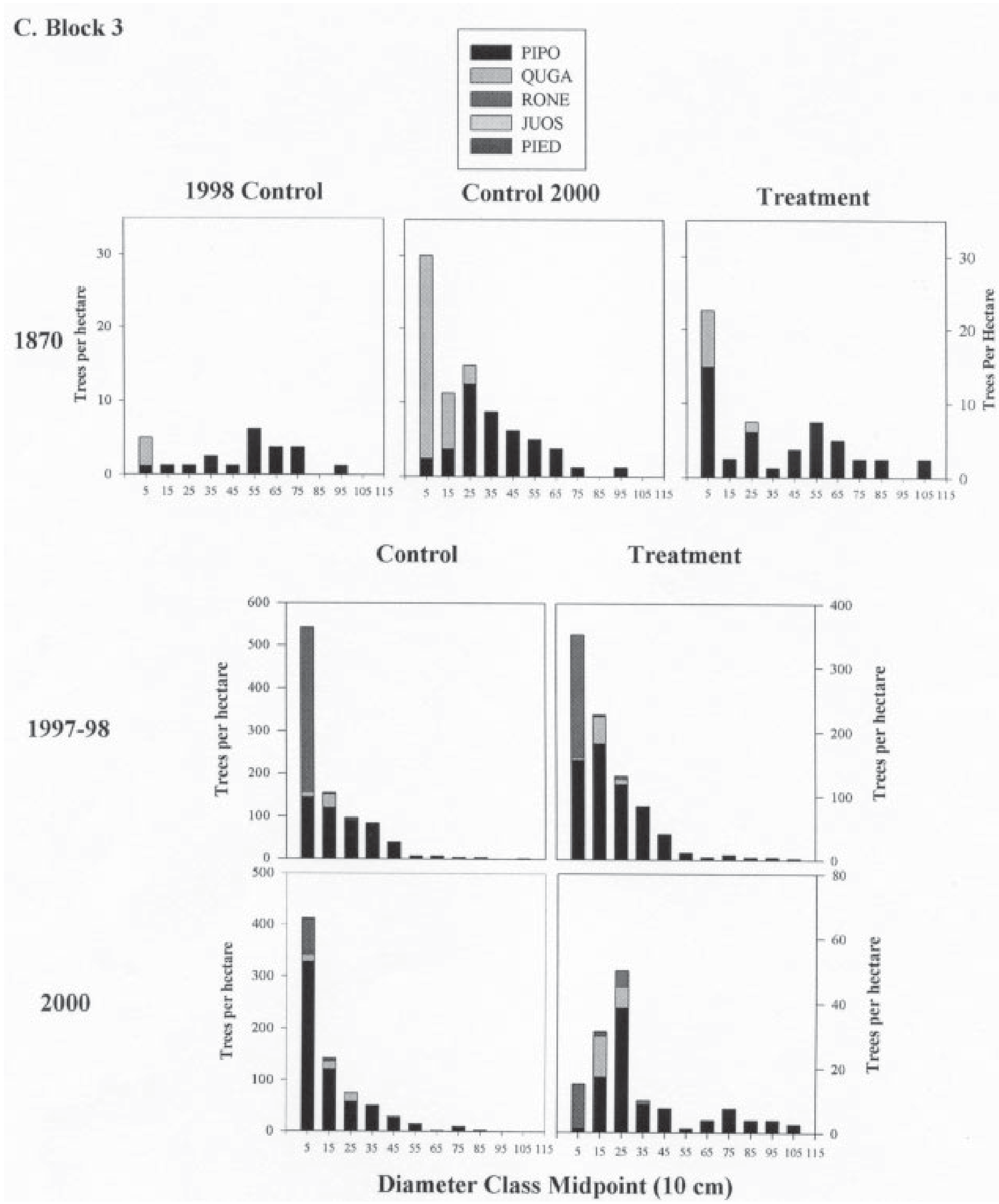

Figure 2c.

1 and 3, and 90% of the trees in block 2 due mostly to oaks. Block 4 was still predominantly composed of postsettlement trees (99%). No trees that established before 1870 were removed by the thinning treatment.

Restoration treatment reduced canopy cover from a pretreatment average of 56% to a posttreatment average of 35% following restoration treatment; canopy cover was significantly lower in treatment units than in control units (2-way ANOVA $F = 215.7$, $P < 0.001$). Posttreatment canopy cover ranged from 22% in EB3 to as high as 53% in EB4. Canopy cover is correlated with trees per hectare, but also gives an indication of tree biomass. Other studies show canopy cover to be inversely related to understory production (Moore and Deiter 1992), and lower canopy cover values may indicate future increases of herbaceous production.

Regeneration of all tree species showed no significant differences prior to treatment in contemporary forests when analyzed across all blocks (Wilcoxin signed ranks $Z = 0$, $P = 1.00$; control mean ± SE = 2807.5 ± 997.9, treatment mean ± SE = 2852.5 ± 1326.3). Following treatment, total tree regeneration was significantly higher in restored units than control (Wilcoxin signed ranks $Z = 1.83$, $P = 0.068$, control mean ± SE = 2028.8 ± 848.8, treatment mean ± SE = 2432.5 ± 1018.5). Variation within units was high when examined by block (Table 2). The only tree species with regeneration present in all four blocks was ponderosa pine, and this species had identical regeneration patterns prior to restoration treatments (Figure 4, 1997–1998 Wilcoxin signed ranks $Z = 0.37$, $P = 0.715$). However, following restoration treatments, this species showed a significant reduction in regeneration (Figure 4, 2000 Wilcoxin signed ranks $Z = -1.84$, $P = 0.066$), which was counteracted by increases in the deciduous, sprouting species (oak and locust) following restoration. Gambel oak and New Mexican locust were not equally represented in the blocks, so effects of restoration treatment could not be tested effectively for these species.

Total dead fuel loadings were high prior to treatments, ranging from a low of 16 Mg/ha in EB3 to a high of 64 Mg/ha

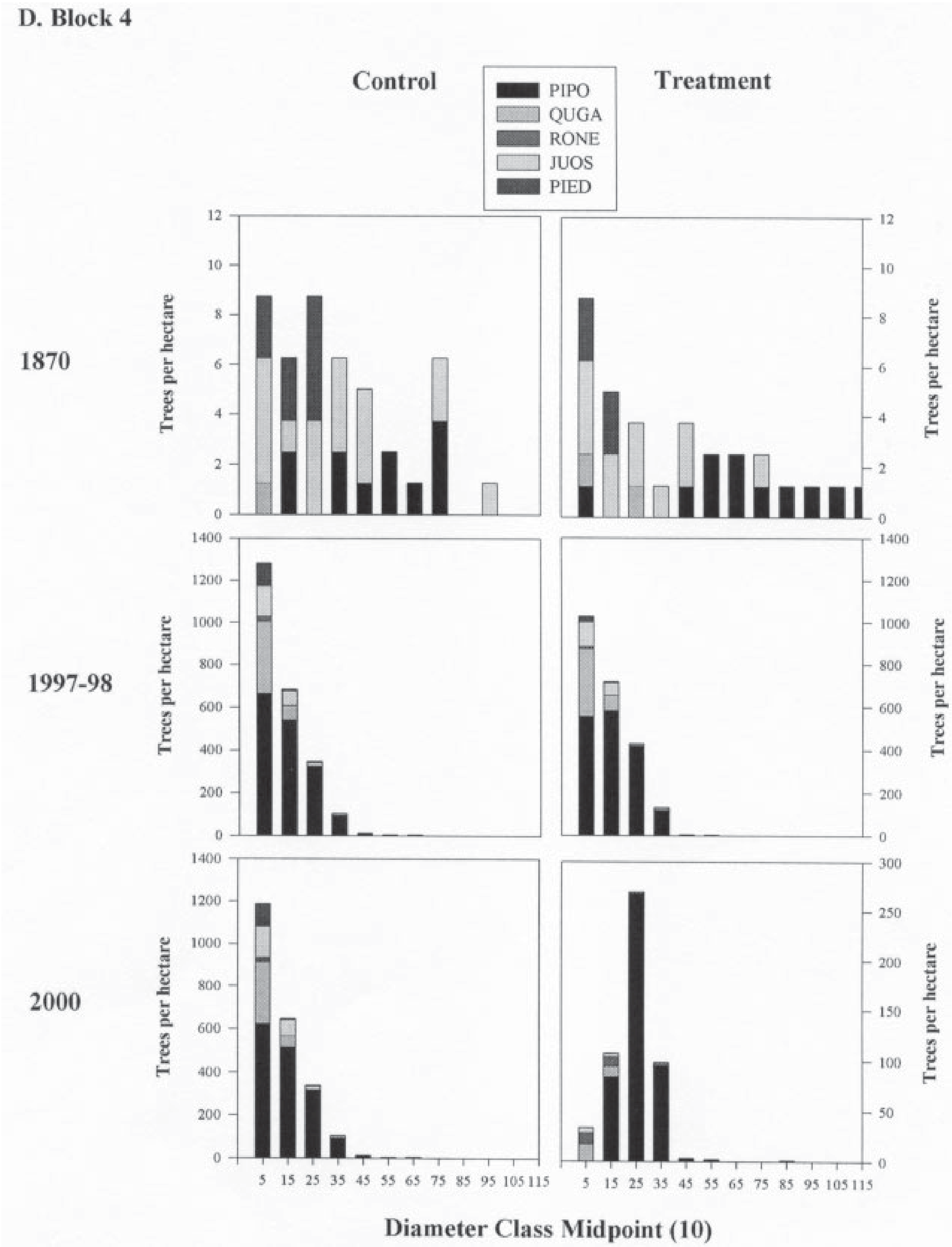

Figure 2d.

in EB2 (Table 3). Fuels averaged 34 Mg/ha prior to treatment across the entire site. Forest floor depths ranged from 2 cm to 5 cm, and no significant differences between treatment units were found prior to treatment (Figure 5A, MANOVA F = 1.54, P = 0.301). Woody debris was highly variable across the site and also showed no significant differences between control and treatment units prior to restoration treatments (Figure 5A, MANOVA F = 12.6, P = 0.075).

Significant forest fuel depth reduction occurred in treatment units across blocks in litter and duff layers (Figure 5B, MANOVA F = 9.19, P = 0.021). No statistically significant differences were found in woody debris following restoration treatments (Figure 5B, MANOVA F = 1.3, P = 0.486). However, trends toward increases in 100 hr fuels and 1000 hr sound fuels, as well as decreases in 1000 hr rotten fuels, are apparent in Figure 5B. The thinning component of restoration treatments adds sound fuel through slash, often resulting in increases in larger woody fuels, represented by 100 hr and 1000 hr fuels. The burning phase reduces fine fuels, such as litter and duff, and rotten woody fuels.

Discussion

Structural Changes since 1870

Forest structure showed significant increases in tree density and basal area since 1870 when the frequent surface fire regime was halted at the Mt. Trumbull site. On average, 1998 tree densities were 25 times higher than 1870 densities, and basal area values were approximately 2.5 times higher. Ponderosa pine densities were 13 times as high in 1998 in blocks 1–3, but increased at EB4 an

Table 1. Basal area (m^2/ha) at Mt. Trumbull study sites in 1870 (reconstructed), 1997–1998 (pretreatment), and 2000 (posttreatment), mean (in bold) and standard error in parentheses. C = control, T = treated. Species codes are derived from the first two letters of the genus and species (JUOS = *Juniperus osteosperma*, PIED = *Pinus edulis*, PIPO = *Pinus ponderosa*, QUGA = *Quercus gambellii*). Block 3 Control 2000 was established in May 2000 to replace the 1997–1998 control lost to wildfire.

Block	Date	Total	JUOS	PIED	PIPO	QUGA	RONE
1C	1870	7.0 (1.5)			7.0 (1.5)	0.004 (0.004)	
	1997-98	26.9 (2.3)		0.002 (0.002)	26.7 (2.4)	0.2 (0.2)	0.02 (0.02)
	2000	27.0 (2.3)			26.8 (2.4)	0.2 (0.2)	0.008 (0.008)
1T	1870	5.7 (0.9)			5.6 (0.9)	0.02 (0.02)	
	1997-98	23.7 (1.7)			23.2 (1.7)	0.5 (0.4)	
	2000	15.5 (1.7)			15.2 (1.7)	0.3 (0.2)	
2C	1870	13.8 (3.3)			13.7 (3.4)	0.1 (0.05)	
	1997-98	36.1 (2.8)	0.1 (0.07)		31.6 (2.7)	4.4 (0.9)	0.03 (0.03)
	2000	35.6 (2.7)	0.1 (0.07)		31.3 (2.6)	4.2 (0.9)	0.03 (0.03)
2T	1870	4.6 (1.6)			4.1 (1.6)	0.5 (0.2)	
	1997-98	28.2 (3.8)	0.005 (0.004)	0.03 (0.02)	17.5 (3.1)	10.3 (2.7)	0.3 (0.1)
	2000	14.9 (2.5)	0.002 (0.002)	0.01 (0.01)	7.5 (2.1)	7.3 (2.0)	0.06 (0.04)
3C	1870	5.7 (1.7)			5.7 (1.7)	0.002 (0.001)	
	1997-98	30.4 (3.8)	0.09 (0.09)	0.2 (0.09)	29.5 (3.9)	0.6 (0.3)	0.04 (0.02)
3T	1870	9.2 (2.1)			9.1 (2.2)	0.08 (0.06)	
	1997-98	39.7 (4.3)	0.01 (0.01)	0.04 (0.04)	38.3 (4.5)	1.0 (0.4)	0.4 (0.4)
	2000	17.0 (3.6)			16.1 (3.7)	0.4 (0.2)	0.4 (0.4)
3C 2000	1870	6.7 (1.4)			6.4 (1.4)	0.3 (0.2)	
	2000	27.0 (3.1)	0.02 (0.02)	0.01 (0.01)	25.4 (3.1)	1.4 (0.7)	0.2 (0.1)
4C	1870	6.4 (1.6)	3.1 (1.3)	0.3 (0.3)	3.0 (1.2)	0.003 (0.003)	
	1997-98	44.6 (3.0)	3.9 (1.4)	1.1 (0.9)	38.4 (3.7)	1.2 (0.4)	0.007 (0.004)
	2000	44.2 (3.0)	3.7 (1.2)	1.1 (0.9)	38.3 (3.7)	1.1 (0.4)	0.009 (0.006)
4T	1870	7.5 (2.0)	1.2 (0.7)	0.05 (0.05)	6.2 (2.0)	0.09 (0.08)	
	1997-98	50.3 (1.8)	2.3 (0.5)	0.4 (0.3)	46.3 (2.1)	1.3 (0.3)	0.003 (0.003)
	2000	26.4 (2.3)	0.8 (0.5)	0.08 (0.05)	25.3 (2.2)	0.2 (0.09)	

order of magnitude higher, with ponderosa pine more than 100 times as dense in 1998 (1700 trees/ha) than in 1870 (14 trees/ha). These changes are characteristic of many Southwest forests where dense forest stands established after interruption of the fire regime (Cooper 1960, White 1985, Covington et al. 1994, Fulé et al. 1997). Although heavily logged, some large diameter trees remained in the current stands across the site. On average, this research site had 16.5 pre-1870 era trees per hectare, ranging from a low of 2.5 trees/ha in EB4 to a high of 35 trees/ha in EB1. Studies have shown these trees are stressed under current dense conditions (Biondi 1996) and are more susceptible to disease and insect attack (Kolb et al. 1998).

Forest composition showed minor alteration in deciduous tree proportions, but did not change noticeably in 3 of 4 blocks, which were consistently dominated by ponderosa pine. However, EB4 saw compositional changes from a pinyon-juniper-ponderosa pine ecotone in 1870 to an almost pure ponderosa pine stand in 1998. Forest ecotones are the first systems to show compositional changes as the processes that determine species distributions (i.e., climate change, fire) are disrupted (Allen and Breshears 1998, Weltzin and McPherson 2000). Many higher elevation forest studies have found compositional changes, with encroachment and density increases by Douglas-fir and white fir (Heinlein 1996, Allen 2002, Fulé et al. 2002a). This study suggests that comparable changes may be found at the lower elevation ecotones.

The forest reconstruction methodology depended on successfully finding and correctly identifying pre-1870 tree evidence such as stumps, logs, and snags. The likelihood of success in northern Arizona is relatively high given the absence of fire combined with the arid environment to limit decomposition and slow decomposition rates of conifer wood (Fulé et al. 1997). Mast et al. (1999) showed that old wood could be correctly classified in the field. Huffman et al. (2001) tested reconstruction procedures on the oldest historical forest plots in the Southwest, established 1909–1914, finding a missed tree error rate less than 3 trees/ha (approximately 11%) for trees with dbh > 10.2 cm in northern Arizona. The very smallest trees are the most likely to decay rapidly. As an example of the upper boundary of possible error in the present study, assume that all the trees <15 cm dbh present today were to be added to the 1870 forest. The tree density would rise by an average 1,121 trees/ha, changing our assessment of diameter distribution. But the basal area would increase by only 3.7 m^2/ha, accounting for only about 13% of the rise in basal area from 1870 to present. Intersecting lines

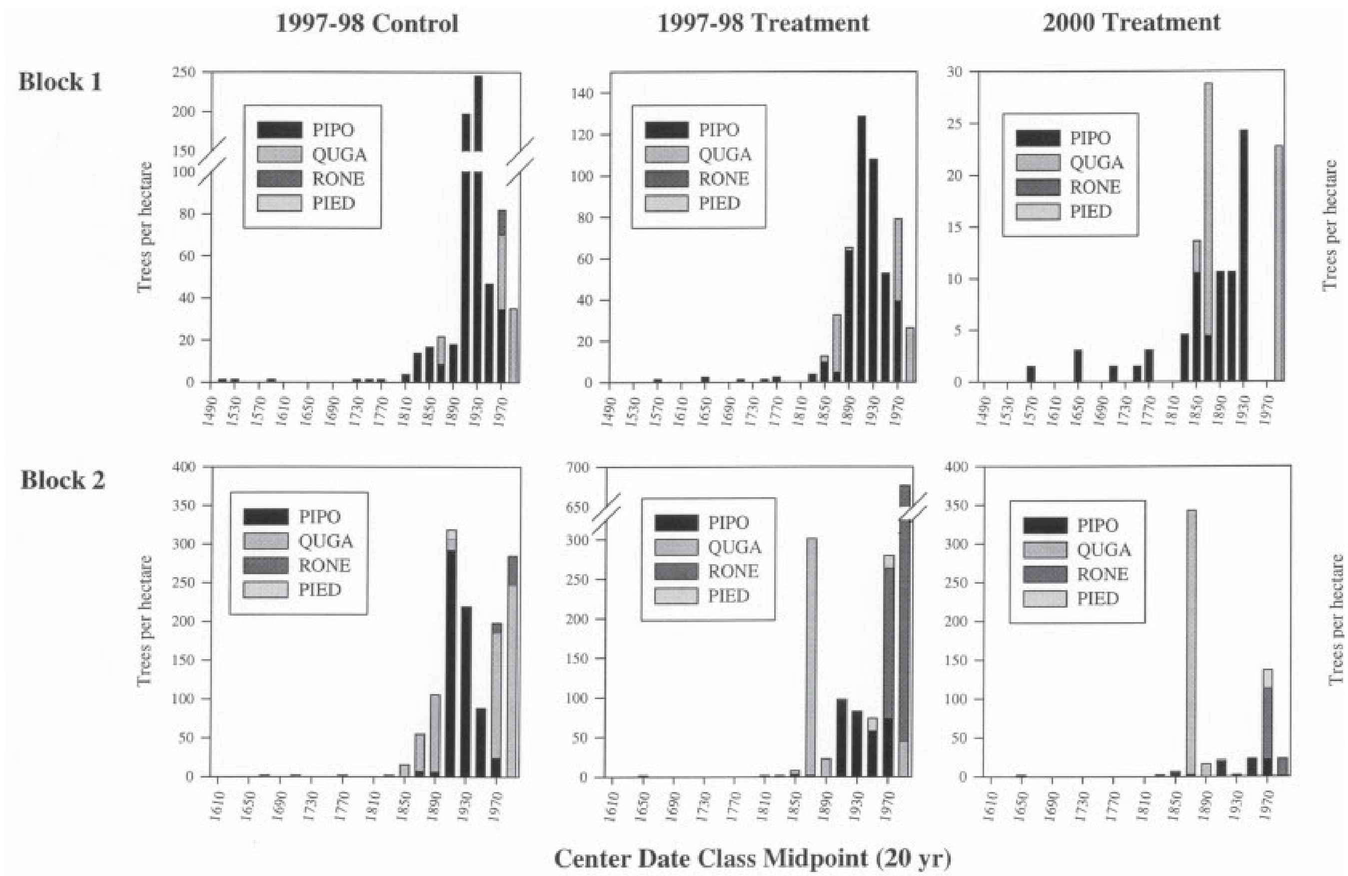

Figure 3a. Age distributions in control and treatment units of each block prior to treatment (1997–1998), and in the treatment units following treatment (2000 Treatment). Trees per hectare were tallied within 20 yr classes; the x-axis represents the midpoint of center date class. Data is presented by block.

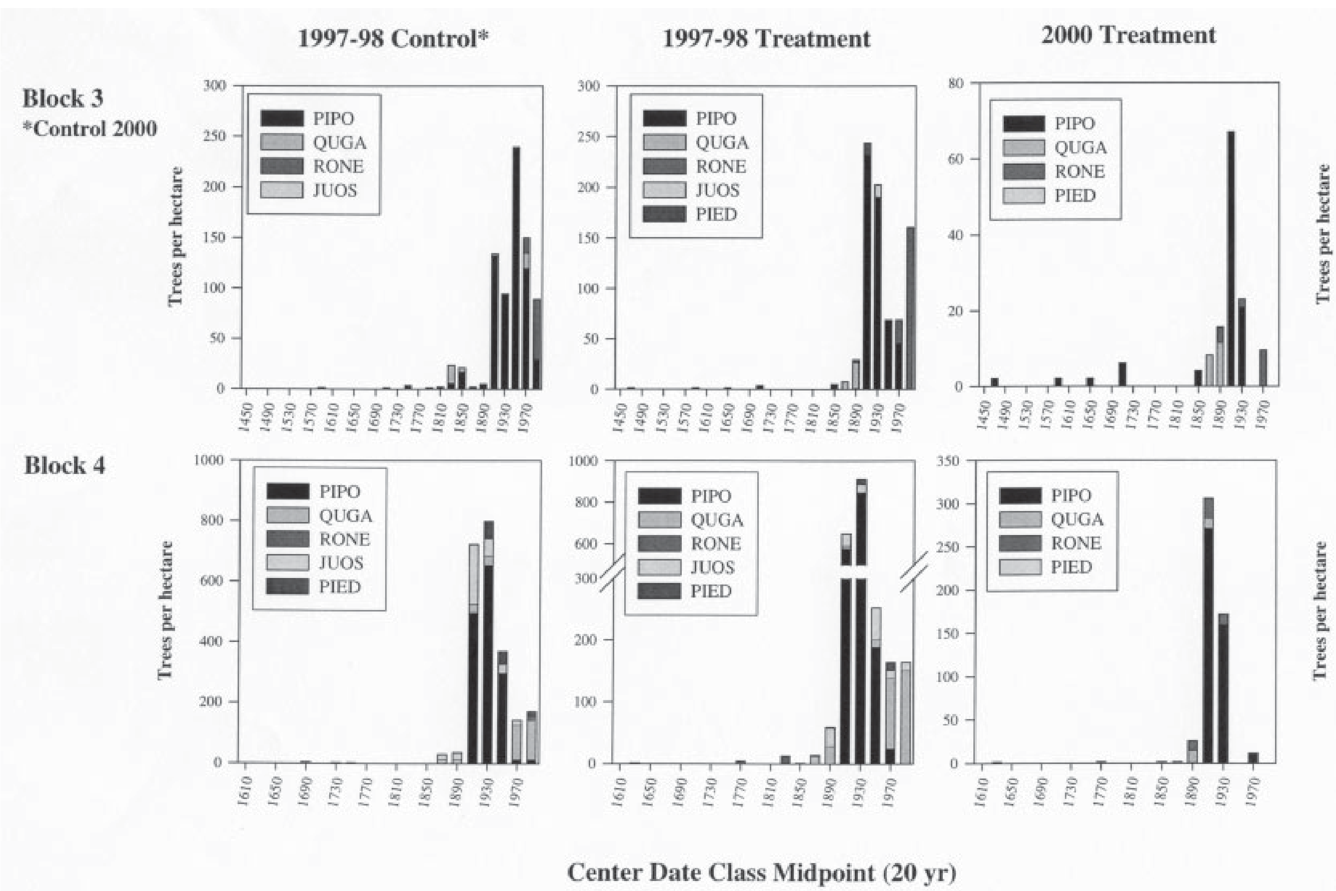

Figure 3b.

Table 2. Mean tree regeneration (trees below 1.37 m in height) in trees/ha across the four blocks by species and height class.

Block	Species	Pretreatment (1997–1998) Total	0–40	40–80	80–137	Postreatment (2000) Total	0–40	40–80	80–137
			..........(cm)..........				..........(cm)..........		
1C	PIPO	75	45	10	20	45	35	5	5
	QUGA	25	15	5	5	95	75	15	5
	RONE	25	5	15	5	35	10	20	5
1T	PIPO	170	10	70	90	20	10	10	0
	QUGA	215	150	65	0	235	130	5	0
	PIED	5	5	0	0	0	0	0	0
2C	PIPO	55	10	35	10	40	15	15	10
	QUGA	4240	2510	1260	470	3240	2275	705	260
	RONE	110	60	45	5	370	215	110	45
	JUOS	10	0	10	0	15	5	0	10
	PIED	20	10	5	5	25	15	5	5
2T	PIPO	85	330	35	20	15	0	5	10
	QUGA	3125	2055	870	200	3480	3405	70	5
	RONE	1080	305	490	285	1420	1015	255	150
	PIED	5	5	0	0	0	0	0	0
3C	PIPO	25	10	5	10	95	35	45	15
	QUGA	15	15	0	0	710	690	20	0
	RONE	2410	745	1090	575	225	75	85	65
	PIED	5	0	5	0	10	5	5	0
3T	PIPO	30	20	5	5	0	0	0	0
	QUGA	100	100	0	0	110	105	5	0
	RONE	685	255	295	135	1300	1015	265	20
	PIED	5	5	0	0	5	5	0	0
4C	PIPO	175	170	0	5	10	5	5	0
	QUGA	3835	2825	720	290	2965	2070	705	190
	RONE	70	50	10	10	55	15	25	15
	JUOS	100	75	15	10	40	30	0	10
	PIED	35	25	10	0	70	40	30	0
4T	PIPO	45	5	30	10)	0	0	0	0
	QUGA	5675	3430	1760	485	3140	3105	35	0
	RONE	15	0	5	10	5	0	5	0
	JUOS	65	30	35	0	0	0	0	0
	PIED	25	25	0	0	0	0	0	0

NOTES: C = control, T = treated. * block 3 Control 2000 was established in May 2000 to replace the 1997–1998 control lost to wildfire, 2000 control values are from the replacement unit.

of evidence from historical sources such as old forest surveys (e.g., Woolsey 1911) are also commonly used to confirm or contradict reconstruction results. Old historical survey data were not available for the Trumbull site, but paired landscape depictions from 1870 and 1995 published by Moore et al. (1999) support our finding of great increases in tree density, especially at EB4 which is in the center of each picture.

Effectiveness of Restoration Treatments

Assessing the effectiveness of an ecological restoration project is complex, involving interrelated elements of ecosystem structure and function (Zedler and Callaway 1999, Block et al. 2001) over a specified time period (Jackson et al. 1995) with respect to specific ecological goals (Landres et al. 1999) and the social context (Higgs 1997). A comprehensive assessment of the Trumbull experiment will require data from numerous studies over an extended period. Here, we limit our discussion to short-term forest structural changes, but we recognize explicitly that forest structure is only one of many important factors.

In the first year after treatment, restoration thinning and burning was successful in reducing tree density while maintaining larger and older trees (Figure 3). However, although forest density was significantly reduced, posttreatment stands remained significantly more dense than 1870 reconstructed stands. There are several implications to leaving higher numbers of trees in the restoration treatments. First, excess tree density provides a buffer for the underestimation of 1870 tree density. In this case, tree densities in restored units were between 100% and 1000% (in EB4) higher than reconstructed pre-1870 tree densities, providing additional trees well above the 11% reconstruction error rate found in Huffman et al.'s study (2001). Second, excess trees also protect against unanticipated mortality during the treatment process itself. Mortality could be associated with factors such as damage

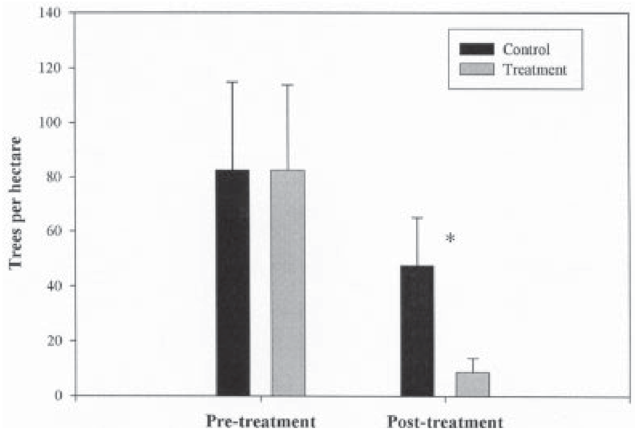

Figure 4. Ponderosa pine (*Pinus ponderosa*) tree regeneration (trees below 1.37 m in height) prior to (1997–1998) and following restoration treatment (2000), in control and treatment units (n = 4). Asterisks (*) denote significant differences. No significant differences were found in 1997–1998 (Wilcoxin signed ranks Z = 0.37, P > 0.10), ponderosa pine regeneration per hectare was significantly reduced in treatment units following restoration treatment (Wilcoxin signed ranks Z = –1.84, P < 0.10).

from falling trees, wind throw, sunscald, insect herbivory, or burning. For example, observations in 2001 showed increased mortality of old-growth trees in EB1 in the second year following restoration treatment, but not in the control (Fulé et al 2002b). This unit is located on lava soils, which may contribute to differential old-growth response to fires (Swezy and Agee 1991). Currently, all prescribed fire on lava soils has been suspended, pending further research. Third, since posttreatment tree densities and basal area are higher than those conditions in 1870, fire hazard probably remains higher than in 1870. A comparison of simulated crownfires by Fulé et al. (2001) showed a substantial reduction in potential crownfire behavior in all four treated units, but the reduction was linked to the level of thinning intensity. Finally, if excess trees are found to survive well in the first decades following thinning and burning, managers should consider additional treatments to maintain open forest conditions consistent with the range of natural variability. These treatments might range from traditional cutting and removal of trees to killing trees on-site with a variety of methods (e.g., girdling, piling fuels under target trees for lethal fire, or attracting bark beetles with aggregation pheromones).

Re-introduction of surface fire was a central objective of the restoration treatment. However, the initial prescribed burn was not comparable to pre-1870 fires as the fuel structures consisted of thinning residues and accumulated forest floor fuels. The initial burn consumed much of the highly flammable slash, including needles and small twigs. However, boles from small-diameter trees remained on the ground following treatment; they may require several fires over a decade or longer until they are mostly consumed. Fuels left after thinning were highly variable, but some patterns emerged when fuel loadings were examined by categories. For example, following the prescribed burn, forest floor fuels (litter and duff) had significant declines, but large-diameter woody debris showed increasing trends, similar to the increase observed by Lynch et al. (2000). Although there were fuel compositional changes, the total fuel loadings did not show significant changes across the four blocks with restoration treatment.

These data indicate that several goals were accomplished in the first year following treatment. Tree density reduction reduced the threat of crownfire (Fulé et al. 2001), and much

Table 3. Forest floor depth and woody debris biomass at Mt. Trumbull study sites before and after restoration treatment, mean (in bold) and standard error in parentheses. The statistics presented are the mean (standard error). Woody fuels are classified by moisture timelag class (1H = 1 hour timelag, etc.). Block 3 Control 2000 was established in May 2000 to replace the 1997–1998 control lost to wildfire, 2000 control values are from the replacement unit.

Block	Year	Litter (cm)	Duff (cm)	1H	10H	100H	1000H sound	1000H rotten	Total wood
				(Mg/ha)					
1C	1997–98	**0.73** (0.08)	**4.40** (0.66)	**0.70** (0.29)	**1.89** (0.38)	**3.30** (1.05)	**18.1** (10.8)	**4.06** (3.30)	**28.1** (10.9)
	2000	**1.22** (0.17)	**3.03** (0.49)	**0.17** (0.04)	**1.01** (0.26)	**2.86** (1.24)	**4.62** (4.46)	**0.41** (0.28)	**9.01** (5.04)
1T	1997–98	**0.48** (0.05)	**1.70** (0.23)	**0.22** (0.08)	**0.90** (0.14)	**1.58** (0.84)	**48.9** (28.0)	**12.6** (8.18)	**64.2** (28.2)
	2000	**0.32** (0.04)	**1.67** (0.17)	**0.23** (0.07)	**1.06** (0.20)	**6.47** (1.31)	**11.4** (4.07)	**12.3** (5.97)	**32.9** (6.88)
2C	1997–98	**0.75** (0.06)	**2.61** (0.25)	**0.22** (0.07)	**1.82** (0.60)	**2.87** (0.75)	**14.4** (8.51)	**13.8** (9.92)	**33.1** (12.2)
	2000	**2.23** (0.23)	**2.32** (0.34)	**0.23** (0.09)	**1.31** (0.32)	**2.26** (0.56)	**12.5** (5.87)	**7.05** (3.20)	**23.4** (6.24)
2T	1997–98	**1.17** (0.16)	**2.62** (0.47)	**0.23** (0.06)	**1.41** (0.25)	**2.43** (0.67)	**42.9** (40.2)	**8.28** (7.33)	**55.2** (47.4)
	2000	**0.90** (0.17)	**0.78** (0.13)	**0.20** (0.06)	**1.05** (0.26)	**4.44** (0.92)	**17.6** (9.22)	**0.23** (0.16)	**23.5** (9.26)
3C	1997–98	**0.83** (0.10)	**4.17** (0.38)	**0.33** (0.08)	**2.33** (0.39)	**2.15** (0.85)	**13.9** (9.30)	**9.20** (5.78)	**27.9** (14.3)
3C 2000	2000	**1.92** (0.28)	**3.87** (0.57)	**0.27** (0.06)	**0.89** (0.19)	**1.58** (0.49)	**3.86** (2.21)	**14.4** (9.44)	**21.0** (11.6)
3T	1997–98	**0.88** (0.12)	**4.06** (0.50)	**0.14** (0.05)	**1.49** (0.43)	**1.58** (0.49)	**4.04** (1.62)	**9.25** (5.62)	**16.5** (5.91)
	2000	**0.47** (0.06)	**1.05** (0.14)	**0.11** (0.03)	**1.25** (0.29)	**3.88** (0.87)	**15.0** (3.73)		**20.3** (4.01)
4C	1997–98	**0.82** (0.10)	**3.74** (0.38)	**0.44** (0.14)	**2.64** (0.62)	**3.72** (1.62)	**10.3** (4.69)	**4.68** (3.78)	**21.7** (7.21)
	2000	**2.21** (0.21)	**3.39** (0.34)	**0.45** (0.17)	**1.19** (0.31)	**6.72** (3.49)	**4.19** (2.05)	**20.1** (15.1)	**32.6** (15.8)
4T	1997–98	**1.15** (0.10)	**3.44** (0.26)	**0.40** (0.11)	**2.40** (0.56)	**3.58** (1.17)	**21.1** (11.6)	**1.79** (1.79)	**29.2** (12.4)
	2000	**1.45** (0.31)	**2.11** (0.23)	**0.31** (0.09)	**1.65** (0.48)	**9.30** (1.71)	**32.6** (6.67)	**1.55** (1.11)	**45.4** (6.90)

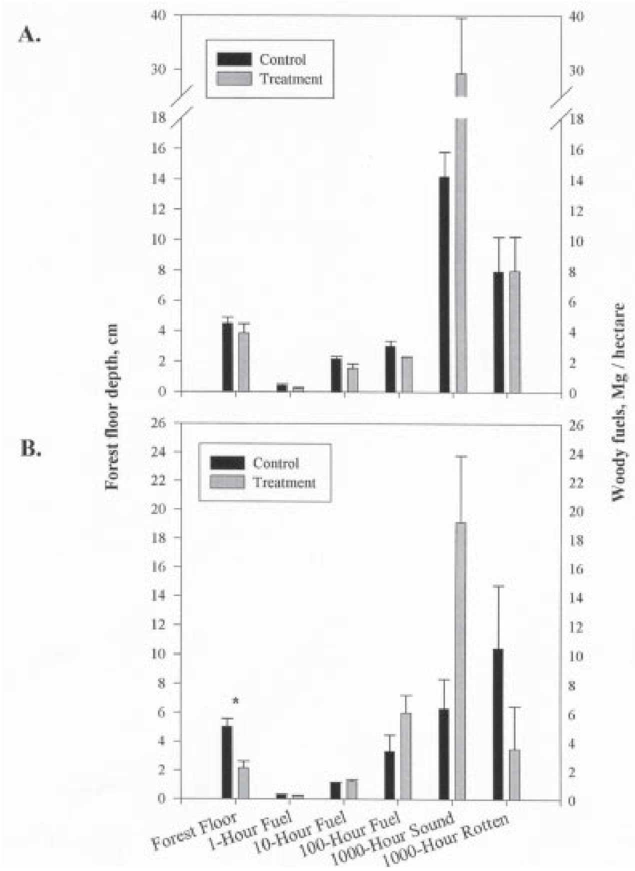

Figure 5. Forest floor depths (cm), and forest woody fuel loadings (Mg/ ha) in control and treatment units. Asterisks (*) denote significant differences. (A) 1997–1998 pretreatment forest floor depth and loading showed no significant differences between control and treatment units. (B) Following restoration treatment, 2000 forest floor depth was significantly reduced in treatment units (MANOVA $F = 9.2$, $P < 0.10$). No significant changes were recorded in woody fuel loadings following treatments.

of the accumulated fine fuel was removed. Tree composition was not significantly altered by the treatments and retained characteristics of pre-1870 forests. Diameter distributions, while still skewed relative to pre-1870 distributions, showed reduction in the proportions of small-diameter trees. Several attributes that were monitored in the restoration treatments may not show effects for several years. For example, tree regeneration was variable but showed significant increases in the first year following treatment, even with declines in ponderosa pine regeneration. Responses 5 and 10 yr following treatment and successive prescribed fires remain to be tested. These responses will probably continue to vary between conifer and deciduous species because of their different seeding/sprouting regeneration pathways. Fuel loadings will continue to alter with recurring fire and tree mortality through time.

Diversity in Forest Structure

A central theme encountered across all aspects of the experiment was the high variability of the posttreatment forest, despite the application of a "uniform" thinning prescription. In part, the variability is by design, because the thinning was designed to use direct, site-specific evidence from stumps, snags, and logs to closely emulate the 1870 conditions of species composition, density, and spatial pattern [see stem maps of forest pattern in Moore et al. (1999)]. Beyond the intentional variability, however, each treatment unit also differed in factors such as soil texture, elevation, slope, aspect, thinning methodology, removal of large logs, and burning effects. The end result was high diversity in forest structure between treatment units, a conclusion that could only be drawn from a replicated experiment.

Changes in forest structure had affected not only the mean density but its variability as well. The 1870 forest structure was relatively uniform, ranging from 26.3 to 106.3 trees/ha per unit (mean 61.3 trees/ha) and 4.6 to 13.8 m^2/ha (mean 7.4 m^2/ha) (Figure 2, Table 1), meaning that across the landscape represented by these experimental blocks the maximum differences in stand density and basal area were 80 trees/ha and 9.2 m^2/ha, respectively. By 1997–1998, the maximum differences grew to 1,881 trees/ha and 26.6 m^2/ha.

Where tree thinning was fully carried out, in blocks 1 to 3, residual diversity in forest structure was closely related to pretreatment forest structure. Blocks 1 and 3 were dominated by ponderosa pine (98% and 95% of pretreatment basal area, respectively). Forest structure after treatment was only 204–286% more dense, and basal area was 189–202% higher than the 1870 density. Block 2, with much Gambel oak, was 535% higher in tree density and 272% higher in basal area than the 1870 forest. However, the numerous Gambel oak stems contributed only 0.3 m^2/ha, <2% of the posttreatment basal area. In block 4, where high tree density of ponderosa pine prevented complete thinning, forest density was 1,489% higher and basal area was 352% higher than the 1870 forest. It is premature at this point to estimate the proportions of the Mt. Trumbull landscape in which posttreatment structure will be similar to each of the experimental blocks. But their deliberate selection as representative sites suggests that the landscape range of structural diversity will not be less than that observed on the blocks.

Posttreatment diversity has many implications for forest managers and stakeholders, a few of which are explored here. First, although canopy fuels were reduced in all treatments, predicted fire behavior varied widely from 0 to 49% canopy burning (Fulé et al. 2001). Crownfire initiation was most influenced by low crown base height, so the dense oak stems in block 2 were the most vulnerable. The situation was reversed in block 4, where high crown base heights were predicted to keep fires on the surface, but high crown bulk density could support active crownfire at windspeeds as low as 36 km/hr. Future fuels may also differ: substantially more herbaceous growth would be expected in blocks 1–3 (basal area 14.9–17.0 m^2/ha) than in block 4 (26.4 m^2/ha) (Moore and Dieter 1992). When the larger Mt. Trumbull landscape has been treated, managers should expect fire behavior to vary relatively widely in the treated areas as well as in the interspersed untreated patches.

Structural diversity at Mt. Trumbull influences wildlife habitat by affecting diverse attributes such as nectar production [butterflies: Waltz and Covington (2001)], microclimate [butterflies: Meyer and Sisk (2001), Meyer et al. (2001)], fledgling success and parasitism [western bluebird: Germaine and Germaine (2002)], and downed woody material (small mammals: Chambers, in press). Most of these research studies were set up to contrast "treated" and "untreated" forest, but this dichotomy is probably unsuitable in light of high variability. Furthermore, extensive edge habitat may be expected to exist after treatment between dense, shrubby pine-oak forest (e.g., EB 2), open patches (e.g., EB 1), and relatively dense pine forest (e.g., EB 4).

These replicated experiments are the first in ponderosa pine restoration at this scale, providing ecological data that are likely to affect management decisions in ponderosa pine forests. Restoration treatments did not re-create the pre-1870 forest or immediately reverse long-term degradation resulting from logging and grazing. Old growth trees were limited at the Mt. Trumbull site due to early 20th century logging, and younger trees will not contribute to forest structure in similar ways for decades or even centuries. However, this treatment is a potential management option that can restore important characteristics of the pre-1870 forest, such as more open stands, which in turn may help prevent damage from uncharacteristically severe fires. Maintenance of prescribed burning and regulation of incompatible activities such as excessive forage consumption represent a vital long-term commitment to ecosystem restoration. Future monitoring is necessary to understand the effects of restoration treatments combined with uses such as recreation or livestock grazing.

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