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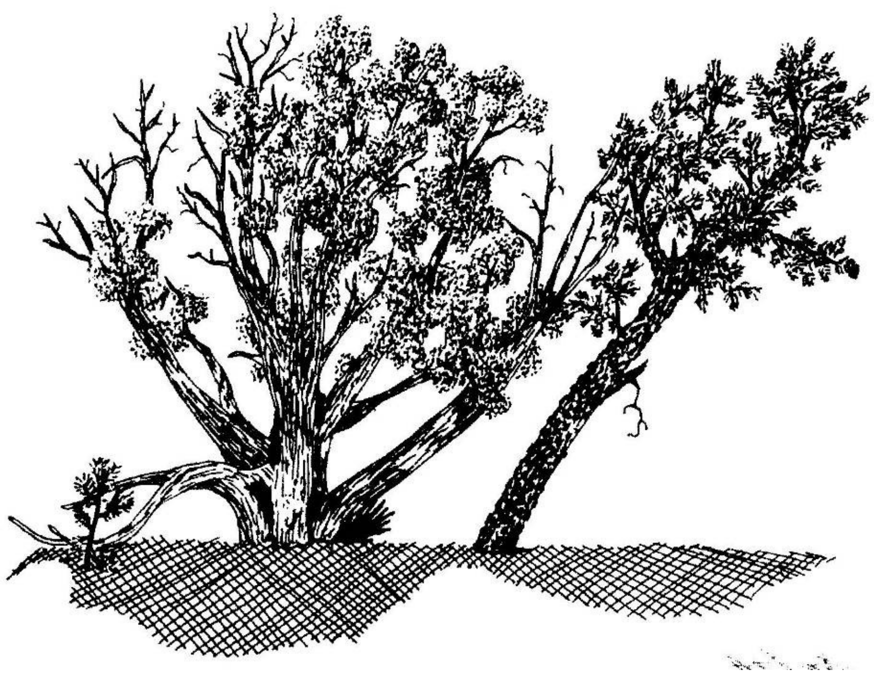

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Executive Summary

The Restoration of Ecosystem Health in Southwest Forests project was initiated in 1995 to develop the scientific basis for ecological restoration of southwestern forests and woodlands at operational, landscape scales. The majority of the work has been focused in the Greater Mount Trumbull Ecosystem within the Grand Canyon/Parashant National Monument. This innovative collaboration between Department of Interior (BLM, NPS, BIA), state (Northern Arizona University, Game and Fish, State Forestry) has resulted in one of the foremost development and application sites for designing, implementing, monitoring, and evaluating restoration-based hazardous fuel reduction and ecological restoration sites in the Intermountain West.

The work described in this report represents an expansion and enhancement of these collaborative partnerships. This report represents work completed the four project areas: pinyon-juniper restoration, pinyon-juniper herbaceous revegetation, cheatgrass abatement and monitoring, and ponderosa pine restoration .

1. **Pinyon-juniper restoration—**although pinyon-juniper woodlands form an important vegetation type on millions of acres of Federal lands, surprisingly little research and testing has been done to restore degraded, fire-susceptible habitats. BLM and ERI designed and implemented pinyon-juniper restoration and fuels management experiments in association with the Greater Grand Canyon/Parashant Ecosystem and related sites. The experimental sites include treatment and control areas, with permanent plots to measure vegetation and fuels. The range of natural variation (reference conditions) of the pinyon-juniper forest structure is being established and used to guide treatment design. This project was launched following a request by the BLM/DOI Washington, D.C. staff.
2. **Pinyon-juniper herbaceous revegetation**—experimental trials to revitalize the native plant community and stabilize soils through increased plant cover. This work was designed to monitor, implement, and evaluate alternative methods of plant community restoration, utilizing by-products of thinning treatments such as slash and mulch. The efficacy of seeding and soil amendments is being contrasted in a controlled, two-factor study. Evaluation and monitoring results will be useful for large areas of PJ woodlands on the Arizona Strip and the Colorado Plateau.
3. **Cheatgrass abatement and monitoring-** Cheatgrass is a serious symptom of poor land health. Cheatgrass out competes valuable forage and increases the risk of wildfire. This project will monitor the response of cheatgrass to different management scenarios and analyze the role of the biophysical environment to its spread. The goal is to provide management

recommendations to land managers and other stakeholders that will help avoid ongoing and future invasion by cheat grass.

4. **Ponderosa Pine Restoration and Hazardous Fuel Reduction Monitoring**- permanent monitoring plots in restoration-treated and control landscapes were re-measured in 2005, five years after the completion of thinning and burning treatments. Changes in forest structure, large tree mortality, tree growth, regeneration, and fuels were assessed.

This work explicitly serves the stated goals of the Administration, the Department of Interior, the Bureau of Land Management, the Healthy Forest Initiative, Healthy Forest Act, National Fire Plan, and Western Governor's 10-year Comprehensive Strategy by developing the scientific basis for restoration-based hazardous fuel reduction and transferring that knowledge to the various stakeholders responsible for developing treatments.

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1: A Demonstration Project for Pinyon-Juniper Ecosystem Restoration

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Abstract

To test an approach for restoring historical stand densities and increasing plant species diversity of a pinyon-juniper ecosystem, we implemented a demonstration project at two sites (CR and GP) on the Grand Canyon-Parashant National Monument in northern Arizona. Historical records indicated that livestock grazing was intensive on the sites beginning in the late 1800s and continuing through the mid 1900s. Repeat aerial photographs (1940 and 1992) indicated recent increases in stand density and encroachment of trees into formerly open areas. Age distributions indicated that the majority of pinyon trees at both sites were less than 100 years of age and juniper establishment appeared to peak in the late 1800s-early 1900s. Junipers had apparent establishment dates as early as 1700-1725. Pretreatment understory communities were sparse (< 7% total herbaceous cover) as were seedling densities in seed banks (151 seedlings per m^2 (14 seedlings per ft^2) at CR and 192 seedlings per m^2 (18 seedlings per ft^2) at GP). Before experimental treatments were implemented, a bark beetle outbreak at GP resulted in >50% pinyon mortality, which was positively related to tree size and age. The demonstration treatment consisted of thinning small trees (< 25 cm diameter at root collar (DRC)), lopping and scattering thinned trees, and seeding native understory species. Thinning and mortality reduced overstory density from 638 and 832 trees per hectare pretreatment (TPH; 258 and 337 trees per acre (TPA)) to 280 and 251 TPH (113 and 102 TPA) posttreatment at CR and GP, respectively. Post-treatment densities were similar to those suggested for the late 1800s by dendrochronological stand reconstructions. Thinning small diameter pinyon increased residual quadratic mean diameter (QMD) at CR and relative importance of juniper at both sites. Live canopy fuels were reduced by treatment at CR and by thinning plus beetle-related mortality at GP. Although thinning slash was lopped and scattered, woody surface fuels were not significantly different between treated and control units at either site. This result was likely due to the small size of thinned trees and the large interspace areas into which slash was scattered. Treatment had no immediate effects on herbaceous cover or species richness, however, both of which appear to be increasing in treated areas. Frequency of most seeded species increased by 2005. Further monitoring is necessary in order to clearly evaluate the restoration and conservation benefits of this treatment.

Introduction

Pinyon-juniper savannas, woodlands, and forests occur on approximately 12 million hectares in the southwestern United States (Springfield 1976). Although these ecosystems are not considered important from a timber production standpoint, they are valued for forage, fuelwood, wildlife habitat, watershed stabilization, recreation, aesthetics, and various woodland products (Gottfried et al. 1995). Tree density has increased since the early 1900s on many pinyon-juniper sites in the Southwest and Great Basin (Tausch et al. 1981, Jacobs and Gatewood 1999, West 1999, Romme et al. 2003). These changes appear to be linked to intensive livestock grazing, exclusion of naturally occurring fire, and climate (Burkhardt and Tisdale 1976, Young and Evans 1981, West 1999, Miller and Tausch 2001). Increases in tree density have led to reduced understory plant species abundance and diversity, increased rates of erosion, and increased susceptibility to severe wildfire. These conditions represent serious problems for conservation of naturally occurring species and ecosystem sustainability. Chaining, cabling, herbicides, and burning techniques have been popular management approaches to create open conditions and increase understory production (Arnold et al. 1964, Wright et al. 1979, Johnsen 1987, McNichols 1987, Erskine and Goodrich 1999, Fairchild 1999, Stevens 1999). In some cases, threshold effects limit recovery of understory communities even when livestock are removed and openings are created (Koniak and Everett 1982, Laycock 1991). Further, the above techniques often convert pinyon-juniper communities to grassland or shrubland types -- an approach that shows little concern for historical patterns of disturbance patterns or ecological structure.

Restoration of historical patterns of disturbance and structure may be the most effective approach for reversing loss of evolutionary habitat and sustaining species viability in ecosystems (Landres et al. 1999, Moore et al. 1999). Romme et al. (2003) identified the following three general structural conditions that represent likely natural patterns for pinyon-juniper ecosystems of the Southwest: 1) open savannas; 2) woodlands; and 3) closed forests. These represent a range of overstory densities and the authors hypothesized that differences may arise due to fire occurrence and severity (Romme et al. 2003). Natural fire patterns depend on fuel characteristics and these are driven by site parameters such as soil texture, topography, and climate. Other disturbance agents that influence structural patterns in pinyon-juniper ecosystems include drought and insects (Betancourt et al. 1993, Negrón and Wilson 2003). In addition to disturbance, open conditions can be maintained through interspecific competitive effects. Recently, interest in restoring pinyon-juniper conditions that emulate historical variability has been increasing (Jacobs and Gatewood 1999, Goodloe 1999, Stevens 1999, Brockway et al. 2002, Jacobs and Gatewood 2002). However, precise understanding of historical dynamics and function of these ecosystems is lacking. Further, natural variability across the range of the pinyon-juniper type appears to be high (Romme et al. 2003). Treatments that show promise for restoring historical stand densities and floral diversity include thinning of small trees, broadcast scattering of slash, and seeding with native plant species (Jacobs and Gatewood 1999, Brockway et al. 2002). Prescribed fire has been used to reduce surface fuel loads, although presettlement fire regimes of pinyon-juniper ecosystems remain unclear and introducing fire can have undesirable effects on understory development (Perry 1993, Jacobs and Gatewood 2002, Baker and Shinneman 2004). Restoration practices have been tested to a limited extent in degraded pinyon-juniper ecosystems of New Mexico but have not been critically evaluated across a broad range of site conditions. Apparently high levels of historical variability suggest site-specific testing of restoration approaches are needed to fully understand the range of viable management alternatives. Further, since historical reference conditions are uncertain, restoration approaches often must be based on broader goals related to improvement of ecological function (e.g., increasing natural biodiversity) and conservation of important attributes (e.g., large, old trees).

In 2002, we initiated a project to demonstrate pinyon-juniper restoration on the Grand Canyon-Parashant National Monument (Bureau of Land Management) in northern Arizona. We desired to formulate a restoration treatment based on the most complete understanding possible of historical site conditions as well as managers' desires for future conditions. The objective of our research was to evaluate the following: 1) the effectiveness of treatment in reducing stand density to levels similar to presettlement conditions and increasing understory diversity; and 2) the constraints and limitations that might hinder similar restoration attempts.

Methods

Study Sites

Two sites were identified for study on Grand Canyon-Parashant National Monument near Mount Trumbull, Arizona. The Craig Ranch site (CR) is located approximately 4.0 km (2.5 mi) north of Nixon Spring Station, at latitude 36N 26' 01" and longitude 113W 09' 40". The Goose Ponds (GP) site is located approximately 5 km (3.1 mi) northwest of Nixon Spring Station at latitude 36N 24' 46", and longitude 113W 12' 15". Elevation of the sites ranges approximately 1900-1950 m (6270-6435 ft) (Fig. 1). Precipitation averages approximately 50 cm (19.7 in) annually and falls during distinct winter and summer periods. Soils at the CR site are shallow to deep gravelly sandy loams to very cobbly clays derived from limestone, basalt, and sandstone alluvium and colluvium. Those at the GP site are shallow to very deep, very cindery loams derived from alluvial and colluvial, scoriaceous basalt and pyroclastics (USDA Soil Conservation Service 1993, 1995a, 1995b). Vegetation at the sites is classified as Great Basin Cold Temperature Woodland (Brown 1994). Overstories are all-aged mixtures of pinyon pine (Pinus edulis Engelm.) and juniper (*Juniperus osteosperma* Torr.).

Understory communities generally are sparse but common species include annual forbs: *Descurainia pinnata* and *Draba* spp.; perennial grasses: *Bouteloua gracilis*, *Bouteloua curtipendula*, and *Aristida purpurea*; perennial forbs: *Eriogonum* spp. and *Chamaesyce fendleri*; shrubs: *Quercus turbinella*, *Purshia mexicana*, and cacti: *Opuntia erinacea*.

Site characteristics suggested degradation of ecological integrity in two main forms: (1) low plant species diversity with communities dominated by dense pinyon and juniper overstories; (2) reduced soil O horizons, particularly in intercanopy openings. Examination of repeat aerial photos suggested that both sites had experienced some degree of overstory densification and tree encroachment from 1940 to 1992 (Fig. 1). Field observation of numerous dead shrub structures on the sites suggested recent change in understory community characteristics. These changes may have been in part due to intensive livestock grazing. For example, a 1961 U.S. Forest Service narrative report explained that grass cover, depleted by uncontrolled overgrazing that before 1900, had not yet recovered (before 1975 the area was under Forest Service administration) (Unpublished report, BLM District Office, St George, UT). Overuse apparently continued through the 1960s; a range inspection report from 1969 stated that all three allotments in the area were in very poor condition and grass present was almost 100% utilized each season (unpublished report, BLM District Office, St George, UT). Repeat aerial photographs indicated that a water catchment was developed at the GP site between 1940 and 1992 (Fig. 1) and historical reports indicated that a pipeline was built to provide water to livestock near the CR site. Intensive grazing likely reduced native plant species diversity and impacted soil quality but its effect on natural disturbance patterns is less clear. Preliminary observations at the two study sites revealed fire scars on live trees and small (1-several trees) patches of apparently fire-related mortality. We did not attempt to reconstruct fire history of the sites.

Design and Measurements

At each site, a 9-ha (22.2-ac) area, that was relatively homogenous in terms of overstory density, slope, and aspect, was delineated from aerial photographs and topographic maps. The areas were divided into two 4.5-ha (11.1-ac) units per site. Units were randomly assigned to receive restoration treatment or remain as a control. Six 0.04-ha (0.1 acre) circular plots were established on a 60-m (196.8 ft) grid in each unit. Plot centers were established with steel rebar and georeferenced for long-term monitoring.

In 2002, overstory, understory, and surface fuels data were recorded on each plot in order to describe pretreatment structure, composition, and response to restoration treatments. Additionally, two photopoints were established per plot to document visual changes. All live and dead trees that were presently greater than 1.37 m (4.5 ft) (or had been in the past) were numbered and measured for diameter at root collar (DRC). Total height was measured for all live and standing dead trees. Increment cores were collected at 40 cm above the ground surface from all trees greater than 20 cm DRC. Increment cores also were collected from a 20% random subsample of live trees less than 20 cm DRC. Longest and shortest crown radii were measure on all trees for which increment cores were collected. Dead tree structures (i.e., snags, logs, stumps) were tallied by condition class as described by Thomas et al. (1979) and Maser et al. (1979) for ponderosa pine.

Transects (50 m (164 ft)) to sample herbaceous plants were centered on each plot and oriented parallel to the slope. Along transects, 1-m^2 sample quadrats were located at 5-meter (16.4 ft) intervals that alternated sides of the midline (10 plots per transect). On each quadrat, cover of herbaceous (non-woody) plant species was be recorded. Transects were also be used as centerlines for 10-m (32.8 ft) wide belts on which a species list of all plants was recorded.

Tree seedlings (< 1.37 m (4.5 ft) in height) were tallied on a smaller, 100-m^2 (0.025 ac) plot nested within the larger overstory plot. Condition (live or dead) and size class (0.1-40, 41-80, and 80-137 cm) was recorded for all seedlings observed. Similarly, within these shrubs were tallied by condition and size within these smaller plots.

Surface fuels and forest floor depth were measured on 15.24-m (50 feet) planar transects (Brown 1974) established in a random direction from plot centers. Photopoints were established at North and East points on the overstory plot perimeter.

In 2002, a severe drought and bark beetle (*Ips confusus*) outbreak occurred. In 2003, before restoration treatments had been implemented, we resampled overstory structure on the plots to assess tree mortality. Signs of beetle presence (e.g, frass or pitch tubes) were noted. After thinning and spring seeding treatment components were implemented (see **Treatment**), we resampled overstory structure, regeneration, shrubs, fuels, and herbaceous understory in June 2004 and 2005.

Treatment

We based our restoration treatment on an approach described by Jacobs and Gatewood (1999, 2002) as being successful at reestablishing understory diversity in a pinyon-juniper ecosystem of New Mexico. This generally entailed thinning trees to low densities, scattering slash, and seeding with native grasses. We modified the prescription, however, in order to focus on restoring site-specific overstory density and spatial arrangement. Additionally, we desired to establish a broad array of understory life forms (i.e., grasses as well as forbs and shrubs).

The treatment we implemented on restoration units was the following: 1) thin pinyon and juniper trees less than 25 cm DRC, except for trees retained to replace presettlement evidence (i.e., dead tree structures >25 cm DRC) at a 2:1 ratio by species; 2) lop slash to 1 m (3.3 ft) or less in length and scatter material to cover bare soil; 3) seed with native plant species. Using tree increment cores, linear regression of establishment date and DRC data suggested that pinyon pine trees less than 25 cm DRC were likely to be less than 130 years of age and therefore postsettlement in origin (Establishment Date = 1977.12 - 4.25*(DRC); $R^2 = 0.57$; $P < 0.001$). Age-diameter relationships for juniper were poor ($R^2 < 0.15$) and cores did not cross-date well. Retaining two postsettlement-aged trees to replace each dead presettlement structure was used as a conservative approach to restoring historical densities and also allowed for posttreatment mortality. Selection of replacement was based on species, size, form, and proximity to structure being replaced. Thinning was completed November, 2003.

Selection of native plant species for seeding was based on observations of local occurrence, baseline data from belt transects, and community data reported in relict site literature (Mason et al. 1967, Schmutz et al. 1967, Thatcher and Hart 1974, Madany and West 1984, Rowlands and Brain 2001). We selected four common grasses, one forb, and four shrub species (Table 1). We broadcast the seed at a rate that approximated common standards for range rehabilitation (Clary 1988). However, we chose to seed both in early spring and late summer in order accommodate germination and establishment requirements for both cool and warm season species. Using site preparation methods such as plowing or disking before seeding was not feasible. Similarly, we did not harrow or rake the restoration units after the seed was broadcast, but instead utilized thinning slash to provide cover and mulch for the seeds.

Data Analyses

Logistic regression was used to test (alpha = 0.05) for relationships between probability of beetle-related mortality and individual tree characteristics for pinyon pine at the GP site. Trees living in 2002 but found dead in 2003 were categorized as killed by beetles if signs of beetles were observed. Tree characteristics tested against mortality were DRC, total height, tree age, and basal area growth. Basal area growth for the last 5, 10, and 20 years was determined from increment core measurements. All pinyon trees measured in 2002 on GP plots were used to test DRC and height variables (n = 263). Only cored trees could be used to examine age and basal area growth variables (n = 48). Beetle-related mortality was minimal at the CR site and this site was not included in the above analysis.

Student’s t-test was used to compare (alpha = 0.05) forest structure and understory means at each of the two sites for pretreatment (2002) differences. When significant differences were found, analysis of covariance (ANCOVA) was used to test posttreatment differences with pretreatment conditions used as a covariate. When no pretreatment differences were found, t-tests were used to compare posttreament means. Overstory parameters estimated were trees per hectare (TPH), basal area (BA), and quadratic mean diameter (QMD) for live trees. Control and treated units were also compared for differences in mean herbaceous plant cover, species richness, and Shannon-Weiner's diversity index (Whittaker 1975).

Presettlement density was estimated for each site using age data from increment cores. Increment cores collected in the field were brought back to the laboratory, mounted on wooden slats, and sanded in preparation for cross-dating under binocular microscopes (Stokes and Smiley 1962). Juniper cores were often difficult to cross-date against known tree-ring chronologies. For such samples, we approximated tree age by conducting maximum (all visible rings) and minimum (only distinct rings) ring counts and averaging this value. Trees with center dates less than 1875 were considered presettlement in origin. To account for additional trees that may have died between 1875 and 2002, we included dead structures (i.e., snags, logs, and stumps) greater that 25 cm DRC in our presettlement density estimates.

Tree canopy biomass was estimated using allometric equations for pinyon (*Pinus edulis*) and juniper (*Juniperus monosperma*) provided by Grier et al. (1992). Surface fuel loading was estimated using equations provided by Brown (1974) and coefficients for ponderosa pine (*Pinus ponderosa*) fuels provided by Sackett (1980); no pinyon or juniper fuels coefficients were available in the published literature. Differences in biomass and fuels means between control and treated units were analyzed as described above for overstory characteristics.

Results

Pretreatment Conditions

Pretreatment measurements in 2002 indicated dense forest conditions at both sites (Table 2). At the CR site, juniper was dominant in the overstory in terms of number of trees (61% of TPH) and BA (83% of BA). At the GP site, juniper trees were outnumbered (29%) by pinyon but made up a greater proportion of the total basal area (73%). The majority of pinyon trees at both sites were less than 100 years of age and a notable spike in establishment occurred after 1950, particularly at the GP site (Fig 2.). Similarly, juniper at both sites appeared to have establishment peaks corresponding to the late 1800s-early 1900s (Fig. 2). At the CR site, junipers had apparent establishment dates as early as 1700-1725. At the GP site, we found no junipers that had established before 1800. Juniper seedlings (individuals less than 1.37 m (4.5 ft) in height) averaged 117-200 per ha (47-81 per ac) at the CR site and 50-117 per ha (20-47 per ac) at the GP site. Pinyon seedlings averaged 900-967 and 617-1433 per ha (364-391 and 250-580 per ac) at the CR and GP sites, respectively.

Understory vegetation was sparse at both sites (Table 3). Mean cover was less than 7% at the CR site and less than 4% at the GP site. Species richness at CR averaged about twice that of GP (Table 4). Total number of understory species observed was 26 and 10 at CR and GP, respectively.

Species occurring on 10% or more of the sampled treated and/or control quadrats at CR in 2002 included *Cordylanthus parviflorus,* (annual forb), *Aristida purpurea*, *Bouteluoa gracilis*, *Carex geophila*, *Poa fendleriana* (perennial grasses), *Arabis fendleri*, *Eriogonum umbellatum*, *Hymenopappus filifolius*, *Hymenoxys cooperi* (perennial forbs), and *Purshia mexicana* (shrub) (Appendix 1a). Only one exotic species was found; *Lactuca serriola* occurred on ~3% of the quadrats in the treated unit. At GP, understory communities were relatively depauperate with only Eriogonum umbellatum and Purshia mexicana occurring on 10% or more of the sampled quadrats (Appendix 1b). No exotic species were found. Seedling emergence from 2002 soil seed bank samples averaged 151 seedlings per m^2 (14 seedlings per ft^2) at CR and 192 seedlings per m^2 (18 seedlings per ft^2) at GP. Seed bank species richness was low; just five species emerged in total from all samples from the CR site and only eight species were found in GP samples. At both sites, seed banks were dominated by annual forbs (Brassicaceae) with traces of perennial forbs within the Asteraceae family. Only one grass species (*Muhlenburgia* spp.) emerged from samples at either site.

Woody surface fuels were minimal at both sites (Table 4). Combined 1-hour and 10-hour fuels averaged 1.2 Mg/ha (0.53 T/ac) at the CR site and 1.7 Mg/ha (0.76 T/ac) at the GP site. Total forest floor depth at both sites averaged less than 1.3 cm (0.5 in). Total live canopy fuels averaged near 7.5 Mg/ha (3.3 T/ac) at both sties (Table 3).

Bark Beetle Effects

Bark beetle-related mortality of pinyon trees in 2003 was significantly related to tree size and growth (Fig. 3). Probability increased with increasing height, DRC, and age. Mortality was less likely for trees that showed high relative BA growth over the most recent 10 years (i.e., 1991-2001) as compared to the 10 years previous to this (i.e., 1981-1991). Mean height of beetle-killed trees was 4.2 m (13.8 ft) whereas surviving trees averaged 3.3 m (10.8 ft). Mean DRC was 12.3 cm (4.8 in) and 10.0 cm (3.9 in) for beetle-killed and surviving trees, respectively. Mean age of beetle-killed trees was 89 whereas surviving trees averaged 64 years. Beetle-killed trees showed an average reduction of 15% (factor of 0.85; 1991-2001 compared to 1981-1991) in 10-year BA increment whereas surviving trees showed an average BA increase of 30% (factor of 1.3).

Treatment Effects

Implementation of the restoration thinning prescription significantly altered overstory structural characteristics at the CR site but did not affect those at the GP site, largely because of the greater impacts of beetle-related mortality (Table 2). Thinning trees smaller than 25 cm DRC, while replacing dead structures greater than 25 cm DRC, reduced the number of juniper trees by nearly one-half and the number of pinyon by more than a factor of three at CR. Thinning at GP reduced the mean number of junipers by 83 TPH (33.6 TPA) but this did not result in a significant difference between control and treated conditions (Table 2). Bark beetle-related mortality resulted in statistically similar pinyon densities between the control and treated units at GP (Table 2). Basal area was not significantly affected by thinning treatment at either site (Table 2). Diameter distributions, however, showed that dominance of small pinyon at CR was decreased by thinning (Fig. 4). This was expressed as significantly greater QMD of both juniper and pinyon in the treated unit compared with the control at CR (Table 2). At the GP site, diameter distributions were affected by both thinning and beetle-related mortality and no significant differences in QMD were found between the control and treated units (Table 2).

No significant differences in herbaceous plant cover, species richness, or diversity were detected between control and treated units at either site in June 2004 or 2005 (Table 3). Species commonly occurring on quadrats were similar to those reported for pretreatment conditions (Appendix 1a). Further, we detected 11 additional species in 2004 and 9 more still in 2005 across the treated and control units at the CR site. *Lactuca serriola* and *Bromus tectorum* were the only exotic species observed and these occurred on 5% of the quadrats (Appendix 1a). At the GP site, 30 additional species were found in 2004 and 12 more still in 2005 across the control and treated units (Appendix 1b). Seven exotic species were observed posttreatment, the most frequent (≤60%) was *Lactuca serriola*. *Bromus tectorum* also appeared to increase in frequency compared with 2002 measurements (Appendix 1b).

Seeded species consistently increased in frequency in treated areas as compared with controls. At CR, all seeded grass species increased frequency relative to the control with the exception of *Oryzopsis hymenoides* and *Bouteluoa gracilis* (Fig. 5). *O. hymenoides* was not observed on any plot. The nitrogen-fixing forb, *Lupinus argenteus*, was observed in nearly 25% of the treatment unit plots whereas it was not detected at all in the control unit. *Rhus triblobata* was the only seeded shrub that was observed; its frequency was 18% in the treated unit and 0 in the control (Fig. 5). Similar patterns were seen at the GP site. *Oryzopsis hymenoides* was not observed yet all other seeded grasses increased frequency in treated units compared with controls (Fig. 6). *Amelanchier utahensis* was the only seeded shrub to be observed (Fig. 6).

Stand densities after thinning at both the CR and GP restoration units were similar to presettlement estimates. At the CR site, the number of trees estimated to exist in 1875 was 261 TPH (102.7 TPA). Approximately 75% of these presettlement trees (live and dead) were juniper (196 TPH (79 TPA)). The total number of trees remaining after restoration thinning at CR was 280 TPH (113 TPA) and juniper comprised approximately 73%. At GP, estimated presettlement density was 104

TPH (42 TPA) and approximately 60% of these trees were juniper. There were 251 TPH (101.6) remaining after treatment at GP and juniper made up 62% of the residual number.

Thinning increased surface fuel loading at both sites, particularly for moisture-lag classes greater than 10-hours (Table 4). Differences between control and treated units, however, were not statistically significant at either site. Changes in forest floor litter and duff depths due to treatment were minimal and remained low after treatment (Table 4). Canopy biomass was significantly reduced by thinning at the CR site (Table 4). Due to beetle-related mortality that occurred in the both treated as well as the control units, no significant differences were found in live canopy biomass at GP site in 2004 (Table 4).

Discussion

Postsettlement Changes

Various lines of evidence, including historical and contemporary aerial photographs, diameter and age distributions, and dendrochronological reconstructions indicated a transition at both study sites from previously more open stand conditions existing in the late 1800s to closed conditions found at the site in 2002 before experimental restoration treatment was implemented. At the CR site, the number of trees in 2002 was more than twice the number estimated to be present in 1875. This difference was even more dramatic at the GP site where 2002 density was greater than the estimated presettlement number of trees by a factor of eight. In addition to changes in overstory density, both sites appeared to be moving toward increased importance of pinyon relative to juniper. Large junipers were present at both sites; this was reflected in greater BA in comparison to pinyon. Age data suggested that peak juniper establishment was around 1875-1900 at CR and 1875-1925 at GP whereas pinyon establishment appeared to peak around 1950 at both sites. It should be noted that precise cross-dating of juniper is difficult and these establishment dates are best considered as approximations. Pinyon seedlings outnumbered juniper by a factor of four or more at both sites.

Comparable postsettlement changes have been described on pinyon-juniper sites throughout the Southwest and Great Basin (Blackburn and Tueller 1970, West et al. 1975, Tausch et al. 1981, Jacobs and Gatewood 1999, West 1999, Romme et al. 2003, Landis and Bailey 2005). For example, on four black sagebrush (*Artemisia nova*) sites in Nevada, Blackburn and Tueller (1970) concluded that juniper (*J. osteosperma*) initially invaded open sage communities whereas pinyon became more prevalent as overstory densities increased. Age distributions indicated that juniper trees were present as early as 1725 but establishment began to dramatically increase around 1850 for juniper and 1920 for pinyon. Similarly, Tausch et al. (1981) reported that increases in tree dominance since the early 1800s on eastern Nevada and western Utah sites were driven by pinyon establishment. Factors responsible for driving these structural changes include relaxation of interspecific competition due to intensive grazing, increases in woody vegetation ("nurses" for pinyon establishment – see below), fire exclusion due to livestock grazing (removal of fine fuels) and active suppression, warmer, moister climatic patterns of the late 1800s-early 1900s, and recent increases in atmospheric CO_2 (Leopold 1924, Burkhardt and Tisdale 1976, Young and Evans 1981, West 1999, Miller and Tausch 2001, Romme et al. 2003).

Presettlement fire regimes across the range of the pinyon-juniper type may have varied greatly, from frequent, low severity regimes (10-30-year return intervals) to infrequent, high severity regimes (up to 400-year intervals) as determined by physical site conditions (Gruell 1999, Romme et al. 2003, Baker and Shinneman 2004, Floyd et al. 2004). Fire's role, however, in shaping and maintaining presettlement structural patterns in pinyon-juniper ecosystems is unclear (Baker and Shinneman 2004). Uncertainty exists for a number of reasons: primarily, tree characteristics of pinyon and juniper apparently do not allow for long-term recording of fire events (i.e., trees do not typically develop large "cat faces" with multiple fire scars before they suffer from direct or indirectly-caused mortality), and difficulties in accurate cross-dating of juniper wood does not allow for precise fire histories to be constructed (West 1999, Floyd et al. 2004). On our sites, we found no evidence of stand-replacing fire such as extensive areas with charred, dead trees or large, open areas that appeared to be recovering from a severe fire. We did observe individual charred snags and groups of 3-4 charred trees. We also observed fire injuries on live and dead trees. It is probable that periodic fire played some historical role in limiting tree establishment and maintaining more open conditions on our study sites.

Pinyon establishment is facilitated by seed-caching animals and favorable microsite conditions found beneath trees and shrubs as compared to intercanopy spaces (Chambers 2001, Pearson and Theimer 2004). Lower soil temperatures, higher moisture levels, and greater soil concentrations of nitrogen and phosphorus under shrubs and trees compared with interspaces provides suitable "nurse" environments for pinyon seedling survival. Harris et al. (2003) found that composition of vegetation on grazed mesas of southern Utah was significantly higher in forbs and shrubs than on ungrazed mesas, which had relatively more grasses. Thus, intensive grazing on our sites, beginning in the late 1800s and proceeding through the mid 1900s, may have led to increased woody vegetation and decreased grass abundance, which in turn may have facilitated increases in pinyon and juniper establishment. It is well known that herbaceous production and biomass in pinyon-juniper ecosystems decrease as overstory density increases (see Schott and Pieper 1987); extensive lateral roots of juniper (*J. osteosperma*, *J. occidentalis*) effectively "mine" resources such as moisture and nutrients from intercanopy spaces (Everett et al. 1986, Tiedemann and Klemmedson 1995). Thus, with increased tree establishment and growth, understory communities probably continued to decline on our study sites.

Bark Beetles

At the GP site, a bark beetle outbreak reduced overstory density of pinyon and this could be interpreted as a natural disturbance that counteracts recent increases in density and provides restoration benefits. Although this is true in part, at the GP site beetles preferentially attacked larger, older pinyon – elements of the historical stand conditions that are desirable to retain for conservation (e.g., wildlife habitat) and multiresource reasons. Further, high density of standing dead pinyon may represent increased fire hazard for one or two years while dead needles remain on the tree.

Similar patterns of beetle activity were found by Negrón and Wilson (2003) who reported that tree diameter (DRC) and mistletoe infestation were good predictors (72% correct classification) of beetle attack on pinyon near Flagstaff, Arizona. Further, stand density was also positively related to beetle attack. In order to reduce the probability of beetle-related mortality in pinyon-juniper woodlands, reducing pinyon stand density index (SDI; Reineke 1933) to values of 50 or less are recommended (Negrón and Wilson 2003). Thinning at the CR site reduced pinyon SDI to 81 in 2004 from 122 in 2002. Thinning and mortality reduced SDI at the GP site to 34 in 2004 from 212 in 2002. In the control units at CR and GP, SDI values in 2004 were 155 and 75, respectively.

Experimental Restoration

Restoration-based thinning prescriptions implemented at the two sites appeared to be effective in reduce stand density to levels similar to those suggested by dendrochronological reconstructions. The majority of trees removed from both sites were small pinyon (Table 2); this increased the relative importance of juniper and restored overstory composition to characteristics similar to those existing in the late 1800s. Overstory thinning treatments have been utilized in other pinyon-juniper ecosystem restoration experiments and selective tree removal using chainsaws is preferable to indiscriminate techniques such as anchor chaining or cabling that may cause substantial soil disturbance and stimulate regeneration of juniper (Jacobs and Gatewood 1999, Brockway et al. 2002). Previous restoration experiments, however, have set apparently arbitrary goals for residual structure and have not attempted to evaluate the effectiveness of treatments in restoring historical overstory patterns (e.g., Jacobs and Gatewood 1999, Brockway et al. 2002). The approach we tested, however, conserves all large trees. Clearer description of historical patterns could be provided by reconstruction models that utilize dendrochronological information, tree death date predictions (i.e., decay rates, harvesting records, insect outbreak dates, etc.), and back-growth equations.

Thinning treatment significantly reduced live canopy biomass as compared with the control unit at the CR site. This likely represents a decrease in crown fire hazard, although we did not attempt to model fire behavior. At the GP site, bark beetle-related mortality in the control unit decreased live canopy biomass and no significant differences were found between treated and control units. Real differences in crown fire potential may exist at this site since many small trees killed by beetles in the control unit will retain their needles for one or two years, which probably represents an increased hazard. Although thinning slash was lopped and scattered at both sites, we found no significant differences in woody surface fuel loads between the control and treated units. This may reflect the combined influences of thinning only small trees and large interspace areas that were targeted for slash dispersal.

Other pinyon-juniper restoration experiments have tested slash dispersal and seeding treatments to promote understory recovery (Jacobs and Gatewood 1999, Brockway et al. 2002, Stoddard et al. Ch 3. this report). Results have been variable. For example, Jacobs and Gatewood (1999) found that lopping and scattering slash into interspaces substantially increased herbaceous cover at two sites in northern New Mexico, although seeding did not significantly contribute to the increases. In contrast, Brockway et al. (2002) reported positive effects of tree removal on grass cover but no significant differences between slash removal and slash dispersal treatments at a site in central New Mexico. Stoddard et al. (Ch. 3. this report) reports finding significant increases in grass cover on plots with slash additions and experimentally sown seeds compared with control plots and those that were only seeded at two Mount Trumbull sites in northern Arizona. Thinning slash that is scattered into degraded interspaces may increase rates of seedling establishment by altering microsite conditions. Some of these changes may include dampening of soil temperature fluctuations and extremes, increasing soil moisture content, providing structure that traps seeds, and reducing erosion (Jacobs and Gatewood 1999, Hastings et al. 2003, Stoddard et al. Ch 3. this report). Causes of seeding failure may include poor germination, low soil moisture, and high rates of predation (Jacobs and Gatewood 1999). Plant establishment and growth may also be constrained by low nitrogen availability as related to microbial immobilization following a large input of carbon (i.e., thinning slash) (Stoddard et al. Ch 3. this report). Additionally, variation in response to slash and seeding treatments are likely related to preexisting plant community, soil, and seed bank characteristics. In our study, no significant differences in herbaceous parameters were found between the treatment (slash additions and seeding) and control units. However, patterns of increasing cover and species richness are encouraging. Seeding appeared to enhance populations of perennial grasses with the exception of *Oryzopsis hymenoides*. Additionally, seeding with the nitrogen-fixing forb, *Lupinus argenteus*, showed some success and may help to increase nitrogen availability in the treated units. We found only limited evidence of success with respect to seeding shrubs. Further monitoring is necessary before definite conclusions can be made regarding the effectiveness of treatment on increasing understory abundance and species richness.

Conclusions

Although there is still much that is not known regarding the historic ecosystem structure and dynamics at the two pinyon-juniper sites at Grand-Canyon Parashant Monument, the implemented treatment appeared to be effective at reducing stand density and altering overstory species composition to levels more characteristic of the late 1800s. The treatment was also an attempt to gain conditions generally desired by Bureau of Land Management staff and the public. These conditions included a productive, diverse plant community, reduced fuel hazard, and conservation of large, old pinyon and juniper trees. Results indicated that live canopy fuel levels were reduced, which likely reduced crown fire hazard. Old trees were also conserved. These additional goals represent a balance between strict-sense emulation of historical patterns and consideration of present human values. Taken as whole, therefore, this experiment can be described as "good" restoration as proposed by Higgs (1997).

Understory communities appeared to be recovering as evidenced by increasing cover, species richness, and frequency of seeded species by 2005. With any restoration project, long-term commitment is needed by all involved parties in order to properly monitor outcomes, identify transient treatment effects, and evaluate success (Michener 1997, Fulé 2003). In pinyon-juniper ecosystems, plant community development is relatively slow (Erdman 1970) and five to ten years is a likely minimum time over which to monitor treatment effects. Permanent plots established in this study will allow future measurements to be conducted and more definitive evaluations to be made concerning these restoration goals.

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Table 1. Species seeded in 2004 on pinyon-juniper restoration units at Grand Canyon-Parashant National Monument.

	Seeding Date			
	kg per unit		lb per unit	
Species	March	August	March	August
Grasses:				
Bouteloua curtipendula	6.0	6.0	13.2	13.2
Bouteloua gracilis	5.2	5.2	11.5	11.5
Elymus elymoides	2.5	2.5	5.5	5.5
Oryzopsis hymenoides	10.8	10.8	23.7	23.7
Sporobolis cryptandrus	5.1	5.1	11.2	11.2
Forbs:				
Lupinus argenteus	2.3	2.3	5.0	5.0
Shrubs:				
Amelanchier utahensis	2.3	2.3	5.0	5.0
Atriplex canescens	5.1	5.1	11.2	11.2
Ephedra viridis	1.1	1.1	2.5	2.5
Rhus trilobata	1.1	1.1	2.5	2.5
Total	41.5	41.5	91.3	91.3
Rate per hectare (kg)	9.2	9.2		
Rate per acre (lb)			8.3	8.3

Table 2. Overstory characteristics[1] at Craig Ranch (CR) and Goose Ponds (GP) demonstration sites in 2002 and 2004.

			TPH[2]			BA (m^2/ha)[5]			QMD (cm)[6]	
Site	Unit	Date	JUOS[3]	PIED[4]	Total	JUOS	PIED	Total	JUOS	PIED
CR	Control	2002	580	313	893	33.9	7.2	41.1	26.9	16.6
	Treated		387	251	638	25.9	5.1	31.0	29.4	16.2
	Control	2004	568	304	872	33.5	7.0	40.5	26.9	16.7
	Treated		206**	74**	280**	23.0	3.7	26.7	38.3*	26.8**
GP	Control	2002	144	498	642	14.5	8.3	22.8	35.1	14.7
	Treated		239	593	832	19.5	7.1	26.6	34.5	13.4
	Control	2004	144	267	411	14.5	3.1	17.6	35.1	11.5
	Treated		156	95	251	18.4	1.4	19.8	39.3	13.5

[1] Asterisks denote statistically different means for Control versus Treated conditions in 2004; * $P < 0.05$; ** $P < 0.01$
[2] Trees per hectare (divide by 2.47 for trees per acre)
[3] Juniper (*Juniperus osteosperma*)
[4] Pinyon (*Pinus edulis*)
[5] Basal area (divide by 0.2296 for ft^2 per acre)
[6] Quadratic mean diameter measured at root collar (divide by 2.54 for inches)

Table 3. Understory characteristics at Craig Ranch and Goose Ponds sites in 2002, 2004, and 2005 at Grand Canyon-Parashant National Monument. Thinning and seeding was implemented November 2003-August 2004. Sampling was conducted each year in June.

	Craig Ranch			Goose Ponds		
Community characteristic	**2002**	**2004**	**2005**	**2002**	**2004**	**2005**
Cover (%)						
Control	**6.2** ± 1.6	**5.5** ± 2.4	**5.9** ± 2.1	**2.8** ± 1.4	**4.9** ± 1.4	**9.8** ± 1.7
Treatment	**4.7** ± 0.9	**3.6** ± 1.4	**7.9** ± 1.9	**0.9** ± 0.5	**2.7** ± 0.3	**7.5** ± 0.7
Richness[1]						
Control	**18.8** ± 0.4	**24.3** ± 0.8	**28.3** ± 1.0	**8.5** ± 0.8	**27.8** ± 2.8	**27.5** ± 2.5
Treatment	**20.2** ± 1.1	**26.5** ± 1.3	**33.3** ± 1.8	**9.0** ± 0.5	**24.3** ± 1.9	**33.5** ± 2.0
Diversity[2]						
Control	**3.4**	**3.6**	**3.7**	**2.8**	**3.9**	**3.9**
Treatment	**3.4**	**3.7**	**3.9**	**2.8**	**3.8**	**4.0**

[1] Richness is number of understory species based on 0.05-ha (0.12-acre) sample belts.

[2] Shannon-Weiner's index of diversity.

Table 4. Fuels characteristics on control and treated units in 2002 and 2004 at the Craig Ranch (CR) and Goose Ponds (GP) demonstration sites at Grand Canyon-Parashant National Monument. Shown are litter and duff depths, surface fuel weights by moisture timelag class, and live canopy fuels.

			Depth (cm)		Surface Fuels (Mg/ha) [1]						Live Canopy (Mg/ha) [2]		
Site	Unit	Date	Litter	Duff	1H	10H	100H	1000HR	1000HS	Total	JUOS[3]	PIED[4]	Total
CR	Control	2002	0.4	0.4	0.6	1.5	2.9	0.0	0.0	5.0	7.93	2.97	10.9
	Treated		0.3	0.4	0.4	0.8	0.0	1.1	0.0	2.4	5.58	2.11	7.7
	Control	2004	0.2	0.4	0.7	1.2	3.8	0.0	0.0	5.7	7.82*	2.88	10.7*
	Treated		0.4	0.3	0.7	0.3	6.7	0.0	7.8	15.5	4.53	1.39	5.9
GP	Control	2002	0.1	0.8	0.7	1.6	1.4	1.8	2.3	7.8	2.88	3.71	6.6
	Treated		0.2	1.0	0.4	1.3	0.0	6.3	0.6	8.6	3.98	3.32	7.3
	Control	2004	0.2	0.7	0.8	1.9	2.4	3.0	1.8	9.9	2.88	1.51	4.4
	Treated		0.4	0.9	1.0	2.3	4.8	1.4	0.9	10.3	3.46	0.62	4.1

[1] Multiply Mg/ha by 2.4 for approximate T/ac

[2] Estimated using allometric equations provided by Grier et al. (1992). Asterisks indicate statistically different means between treated and control units at $P < 0.05$. Biomass is foliage plus fine twigs.

[3] Juniper (*Juniperus osteosperma*)

[4] Pinyon (*Pinus edulis*)

Table 4. Composition[1] (%) of total understory plant cover by functional group. Note: total plant cover was less than 10% for all years and demonstration units (see Table 3).

	Control			Treatment		
	2002	**2004**	**2005**	**2002**	**2004**	**2005**
Craig Ranch						
Native herbaceous	**43**	**34**	**52**	**57**	**39**	**43**
Exotic herbaceous	**0**	**T**	**T**	**0**	**T**	**T**
Perennial herbaceous	**23**	**34**	**42**	**24**	**38**	**40**
Annual herbaceous	**20**	**T**	**10**	**33**	**1**	**3**
Perennial graminoid[2]	**15**	**18**	**21**	**11**	**8**	**13**
Annual graminoid	**0**	**0**	**T**	**T**	**0**	**T**
Shrubs	**57**	**66**	**48**	**44**	**61**	**57**
Goose Pond						
Native herbaceous	**2**	**30**	**68**	**20**	**60**	**61**
Exotic herbaceous	**0**	**4**	**4**	**0**	**2**	**10**
Perennial herbaceous	**2**	**30**	**64**	**19**	**60**	**58**
Annual herbaceous	**T**	**4**	**8**	**1**	**2**	**13**
Perennial graminoid	**0**	**1**	**1**	**1**	**3**	**6**
Annual graminoid	**0**	**T**	**T**	**0**	**T**	**1**
Shrubs	**98**	**66**	**27**	**80**	**38**	**29**

[1] Values of “T” represent trace amounts less than 1% of total understory plant cover.

[2] “Graminoid” is grass and grass-like species, including sedges.

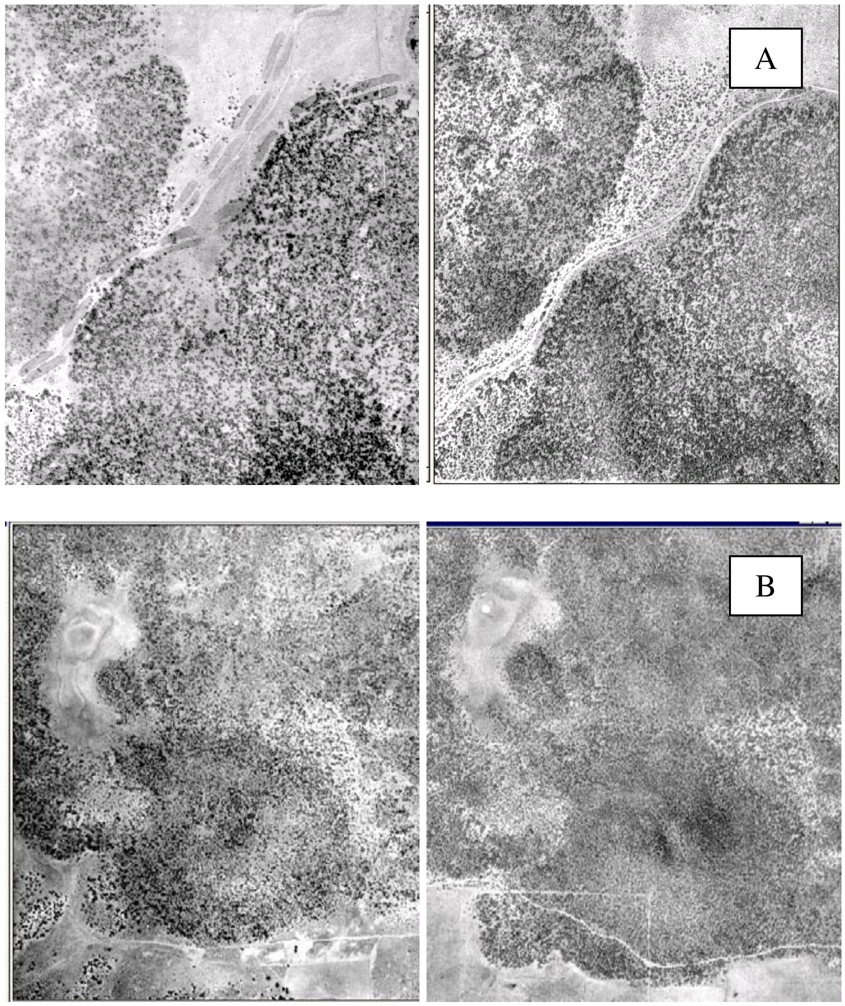
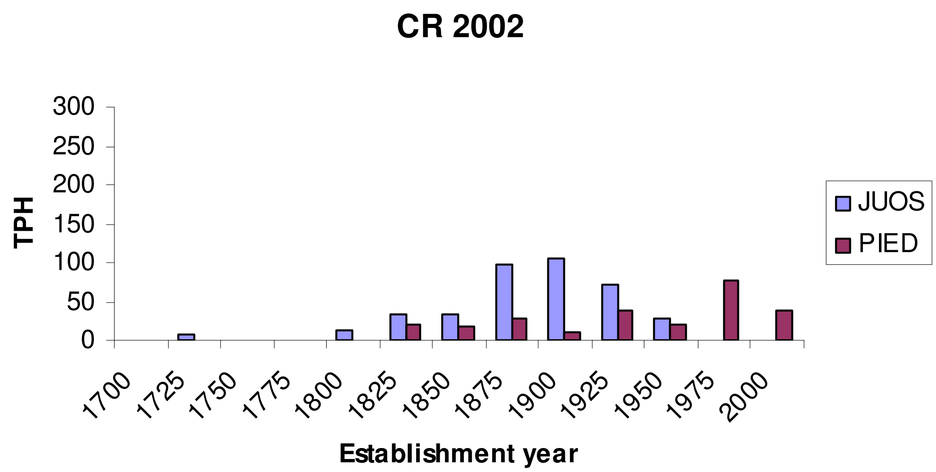


Figure 1. Repeat aerial photographs of Craig Ranch (A) and Goose Ponds (GP) study sites on Grand Canyon-Parashant National Monument. At both sites, the left photograph of the pair shows conditions in 1940 and the right photograph shows conditions in 1992. Note densification of pinyon-juniper stands and encroachment into formerly open areas. At the Goose Ponds site, development of a livestock water catchment is visible in the upper left hand corner of the 1992 photograph.



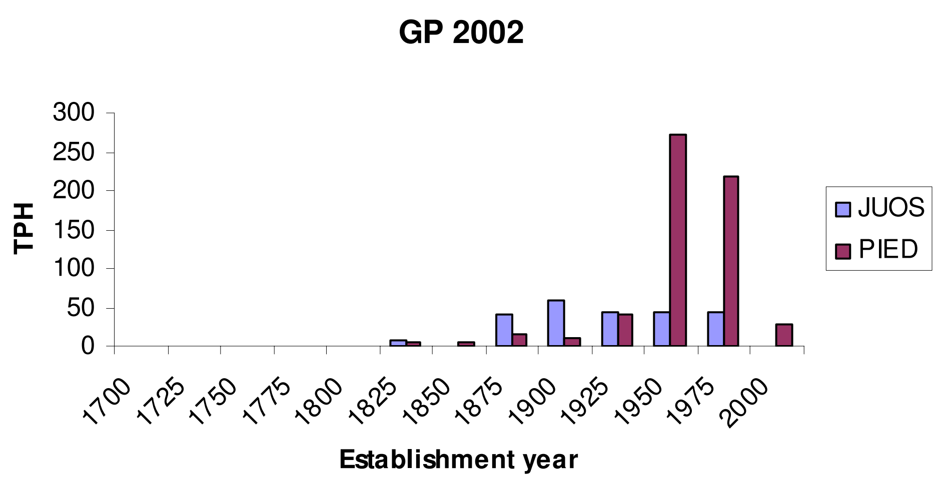


Figure 2. Age distributions for juniper (JUOS) and pinyon (PIED) trees at Craig Ranch (CR) and Goose Ponds (GP) sites in 2002 (pretreatment) at Grand Canyon-Parashant National Monument.

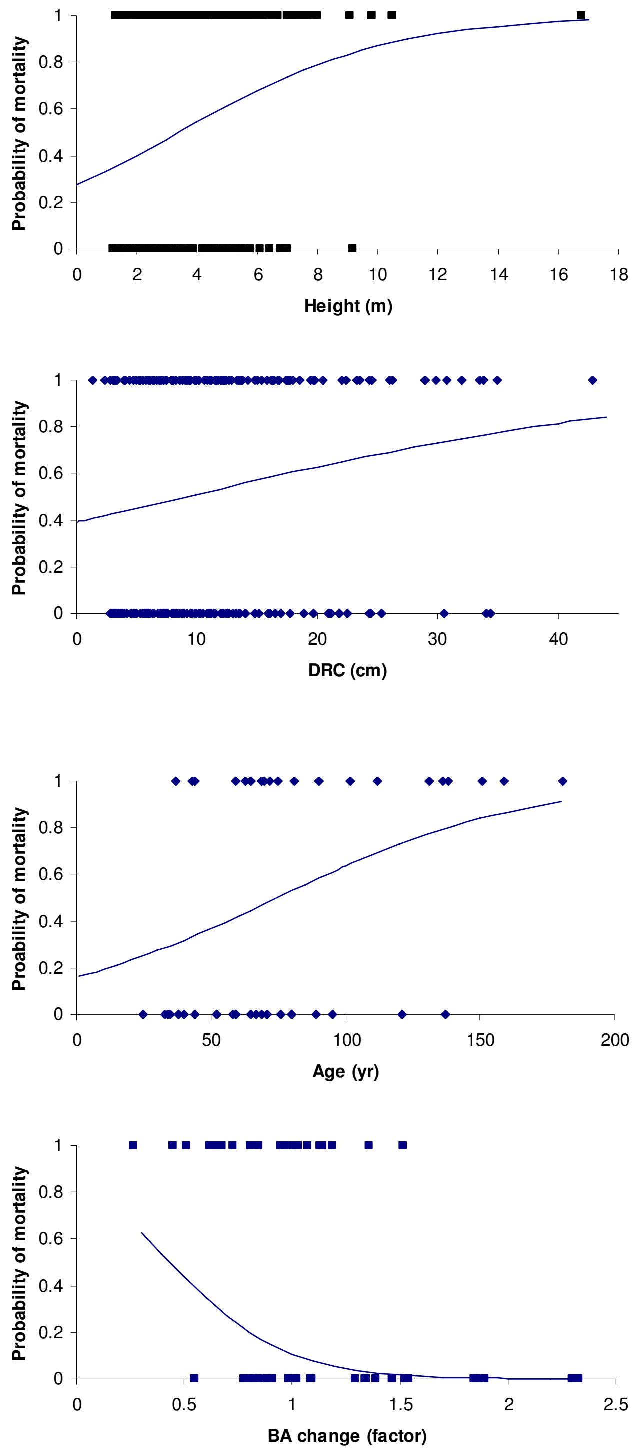


Figure 3. Relationships ($P < 0.05$) between beetle-related mortality and pinyon tree characteristics at Goose Ponds (GP) site at Grand Canyon-Parashant National Monument.

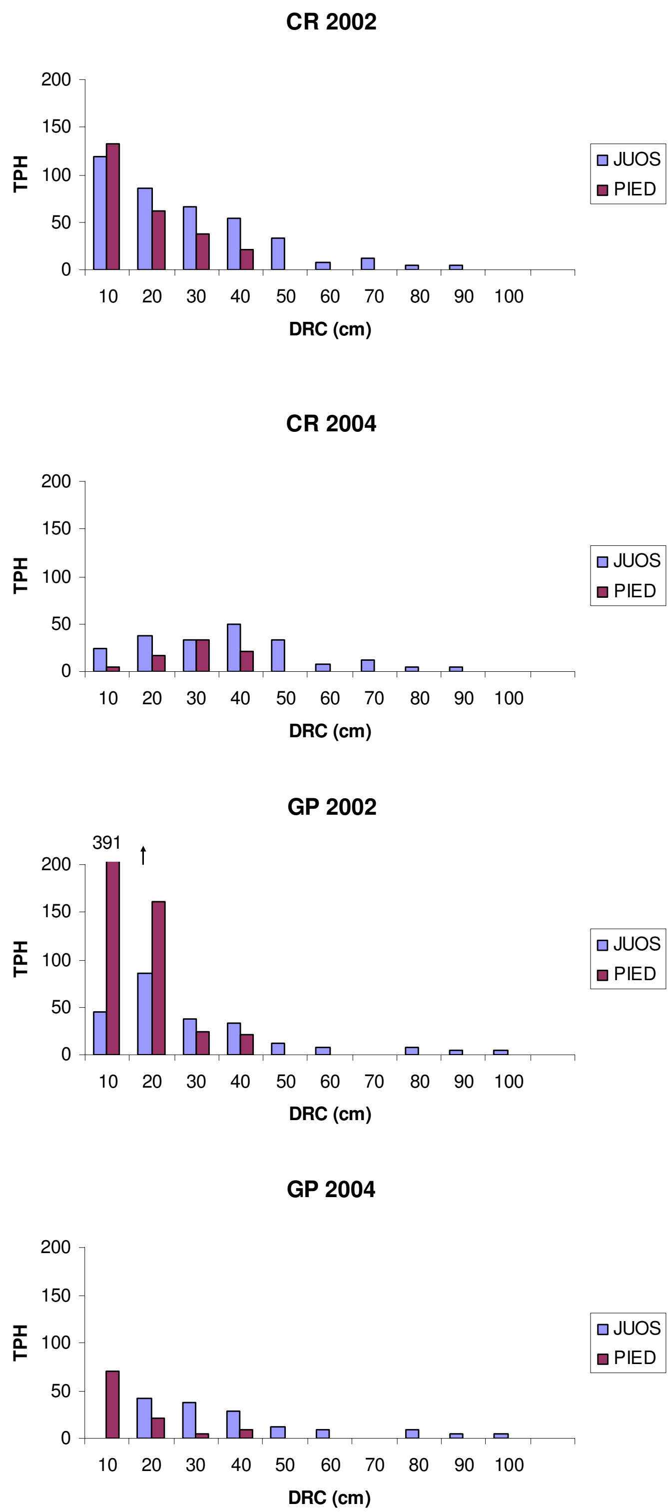


Figure 4. Diameter (DRC) distributions for treated units at Craig Ranch (CR) and Goose Pond (GP) sites on Grand Canyon-Parashant National Monument in 2002 (pretreatment) and 2004 (posttreatment).

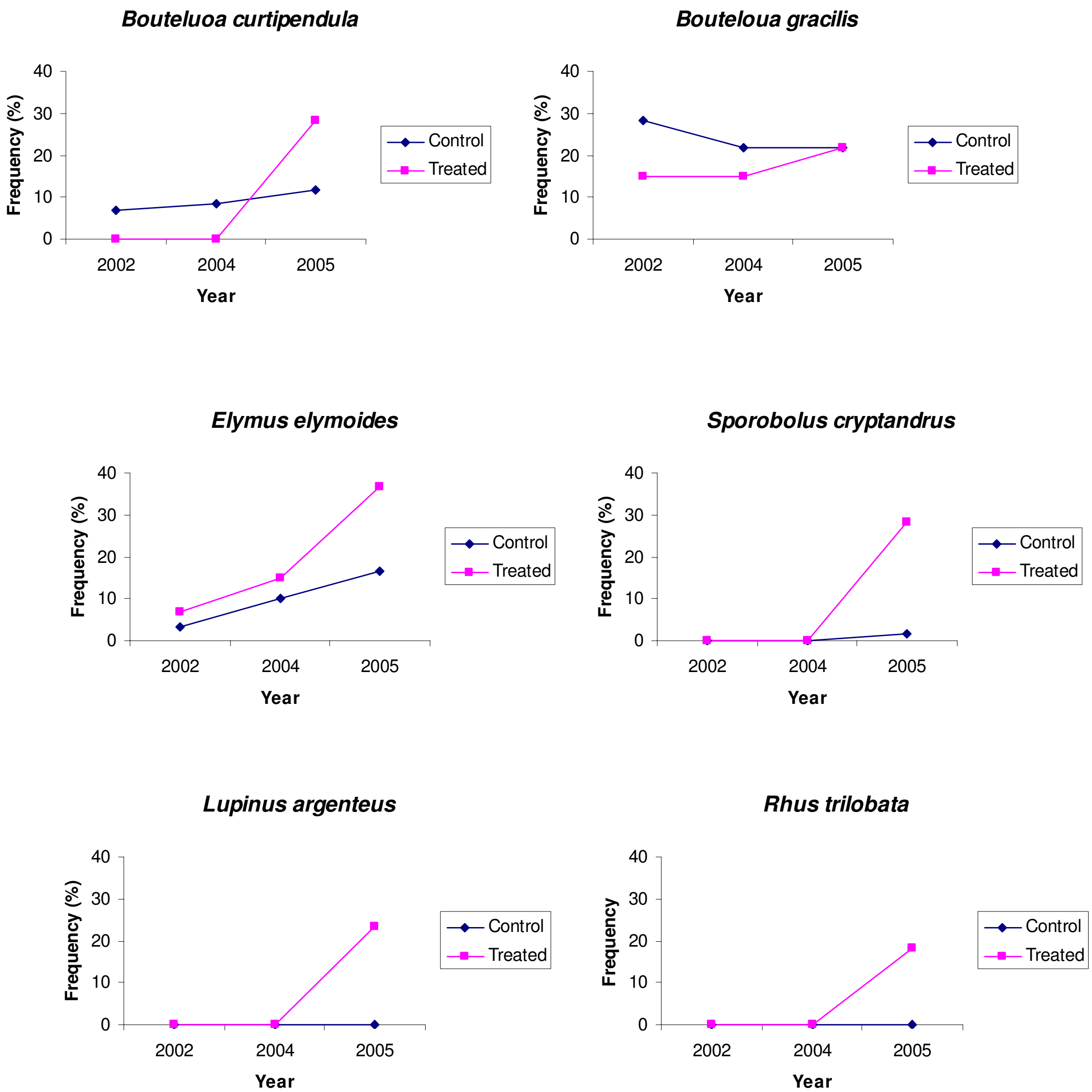


Figure 5. Mean frequency (percent occurrence; standard deviation not shown) of species seeded in the treated unit compared with the unseeded control at the Craig Ranch site. Overstory of the treated unit was thinned in 2003 and seeding was done in 2003-2004.

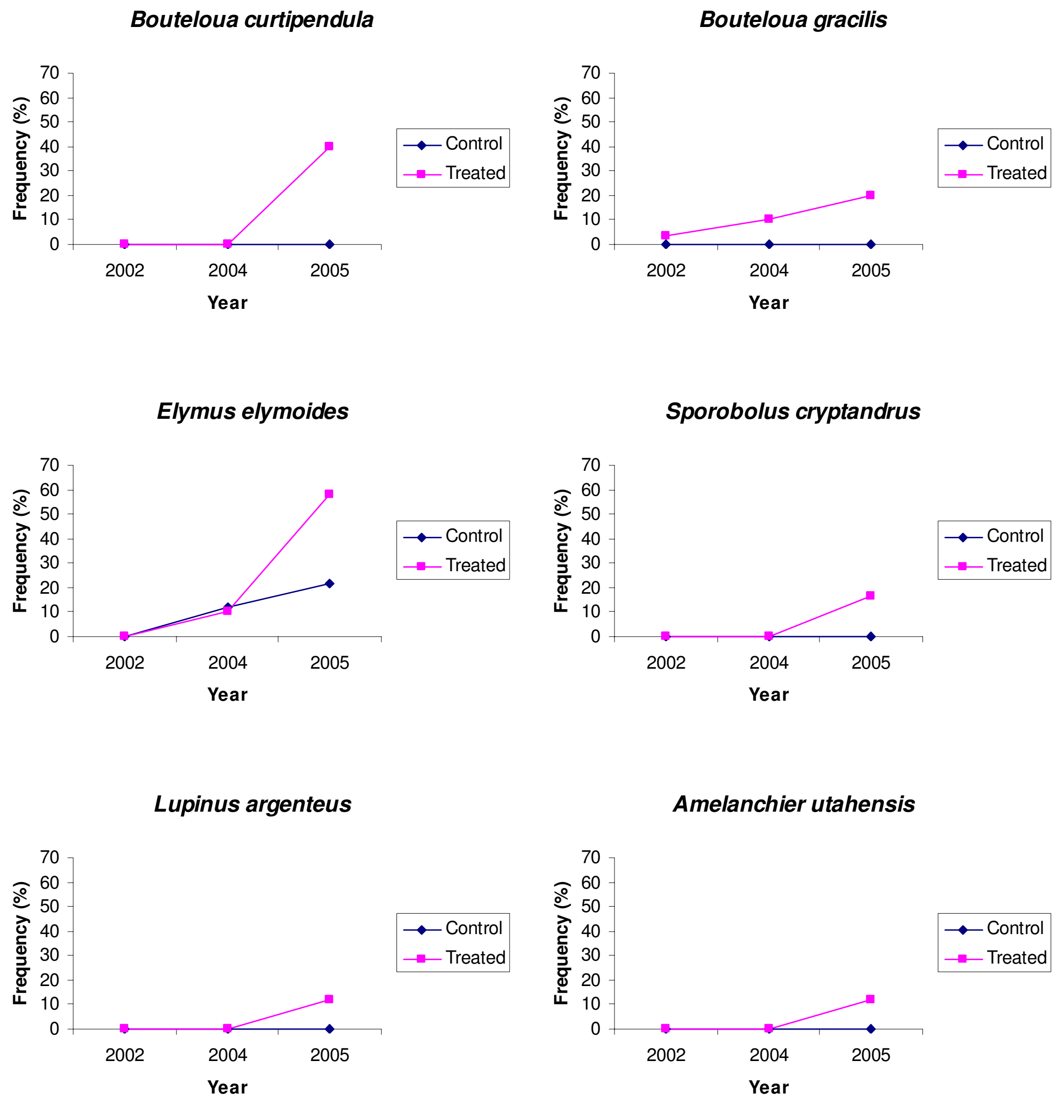


Figure 6. Mean frequency (percent occurrence; standard deviation not shown) of species seeded in the treated unit compared with the unseeded control at the Goose Ponds site. Overstory of the treated unit was thinned in 2003 and seeding was done in 2003-2004.

Appendix 1a. Mean and standard deviation (std) of understory plant species frequency (% occurrence on plots) at cover of understory plant species at Craig Ranch site.

		Treatment			Control		
		2002	2004	2005	2002	2004	2005
Functional Group	**Species**	**mean**	**mean**	**mean**	**mean**	**mean**	**mean**
Annual	*Chenopodium leptophyllum*	0	0	1.7 ± 4.1	0	0	5.0 ± 8.4
	Cordylanthus parviflorus	55.0 ± 24.3	13.3 ± 15.1	16.7 ± 18.6	66.7 ± 30.1	3.3 ± 5.2	56.7 ± 25.0
	Descurainia pinnata	0	0	3.3 ± 8.2	1.7 ± 4.1	0	0
	Descurainia sp.	0	0	0	0	0	1.7 ± 4.1
	Draba sp.	0	0	0	3.3 ± 5.2	0	0
	Linum neomexicaum	0	3.3 ± 5.2	8.3 ± 9.8	0	0	0
	Lupinus kingii	0	0	3.3 ± 5.2	0		0
	Polygonum douglasii	8.3 ± 13.3	1.7 ± 4.1	11.7 ± 18.3	0	0	0
Exotic	*Bromus tectorum*	0	0	5.0 ± 8.4	0	1.7 ± 4.1	5.0 ± 5.5
	Lactuca serriola	3.3 ± .1	0	5.0 ± 5.5	0	1.7 ± 4.1	1.7 ± 4.1
	Tragopogon dubius	0	0	0	0	0	0
Graminoid	*Aristida adsensionis*	1.7 ± 4.1	0	0	0	0	0
	Aristida purpurea	25.0 ± 33.9	20.0 ± 30.3	16.7 ± 23.3	5.0 ± 8.3	5.0 ± 8.4	10.0 ± 12.6
	*Bouteloua curtipendula**	0	0	28.3 ± 21.4	6.7 ± 8.2	8.3 ± 7.5	11.7 ± 13.3
	*Bouteloua gracilis**	15.0 ± 15.2	15.0 ± 13.4	21.7 ± 27.1	28.3 ± 14.4	21.7 ± 17.2	21.7 ± 11.7
	Carex geophila	1.7 ± 4.1	0	0	21.7 ± 29.3	15.0 ± 27.4	10.0 ± 12.6
	Elymus elymoides	6.7 ± 8.2	15.0 ± 10.5	36.7 ± 8.2	3.3 ± 8.2	10.0 ± 12.6	16.7 ± 10.3
	Koeleria macrantha	0	0	0	1.7 ± 4.1	1.7 ± 4.1	1.7 ± 4.1
	Poa fendleriana	1.7 ± 4.1	8.3 ± 9.8	10.0 ± 8.9	20.0 ± 30.3	18.3 ± 31.3	20.0 ± 22.8
	*Sporobolus cryptandrus**	0	0	28.3 ± 14.7	0	0	1.7 ± 4.1
Perennial	*Arabis fendleri*	0	8.3 ± 11.7	20.0 ± 15.5	10.0 ± 11.1	20.0 ± 17.9	41.7 ± 22.3
	Asclepias asperula	0	0	0	0	1.7 ± 4.1	3.3 ± 5.2
	Calochortus nuttallii	0	3.3 ± 5.2	5.0 ± 12.2	0	1.7 ± 4.1	0
	Chamaesyce fendleri	3.3 ± 8.2	5.0 ± 12.3	15.0 ± 18.7	0	1.7 ± 4.1	1.7 ± 4.1
	Cymopterus purpureus	0	6.7 ± 16.3	0	0	3.3 ± 5.2	6.7 ± 8.2
	Echinocereus coccineus	0	0	0	0	0	1.7 ± 4.1
	Eriogonum racemosum	0	0	0	0	1.7 ± 4.1	1.7 ± 4.1
	Eriogonum umbellatum	65.0 ± 28.8	48.3 ± 41.7	58.3 ± 35.4	35.0 ± 29.5	30.0 ± 29.7	25.0 ± 28.8
	Heliomeris multiflora	0	0	1.7 ± 4.1	0	0	0
	Hymenopappus filifolius	11.7 ± 16.0	11.7 ± 11.7	11.7 ± 13.3	6.7 ± 8.2	15.0 ± 13.8	20.0 ± 17.9
	Hymenoxys cooperi	23.3 ± 20.1	21.7 ± 11.7	23.3 ± 12.1	33.3 ± 13.7	33.3 ± 16.3	36.7 ± 16.3
	Lesquerella intermedia	0	6.7 ± 12.1	5.0 ± 5.5	1.7 ± 4.1	10.0 ± 12.6	6.7 ± 8.2
	Lotus wrightii	0	13.3 ± 17.5	11.7 ± 18.3	0	6.7 ± 10.3	3.3 ± 5.2
	*Lupinus argenteus**	0	0	23.3 ± 15.1	0	0	0
	Opuntia sp.	0	0	3.3 ± 5.2	0	0	0
	Orobanche fasciculata	0	1.7 ± 4.1	6.7 ± 10.3	0	0	1.7 ± 4.1
	Packera multilobata	0	0	0	0	3.3 ± 8.2	0
	Penstemon barbatus	0	0	0	0	5.0 ± 12.2	5.0 ± 12.2
	Penstemon linarioides	5.0 ± 8.4	0	0	3.3 ± 5.2	1.7 ± 4.1	1.7 ± 4.1

Appendix 1a (cont.). Mean and standard deviation (std) of understory plant species frequency (% occurrence on plots) at cover of understory plant species at Craig Ranch site.

			Treatment			Control	
Functional Group	Species	2002 mean	2004 mean	2005 mean	2002 mean	2004 mean	2005 mean
Perennial	*Penstemon thompsoniae*	0	18.3 ± 21.4	23.3 ± 20.7	0	11.7 ± 14.7	16.7 ± 15.1
	Phlox longifolia	0	5.0 ± 5.5	1.7 ± 4.1	0	1.7 ± 4.1	3.3 ± 8.2
	Psoralidium tenuiflorum	0	25.0 ± 28.8	35.0 ± 33.9	0	8.3 ± 9.8	10.0 ± 12.2
	Senecio flaccidus	0	1.7 ± 4.1	0	0	0	0
	Sphaeralcea parvifolia	0	1.7 ± 4.1	3.3 ± 5.2	0	0	0
Shrub	*Artemisia tridentata*	3.3 ± 5.2	3.3 ± 3.2	5.0 ± 12.2	1.7 ± 4.1	0	0
	Ericameria nauseosa	3.3 ± 5.2	0	0	0	0	0
	Gutierrezia sarothrae	3.3 ± 8.2	8.3 ± 7.5	8.3 ± 9.8	5.7 ± 5.3	6.7 ± 10.3	5.0 ± 8.4
	Opuntia erinacea	1.7 ± 4.1	0	0	0	0	0
	Purshia mexiccana	6.7 ± 12.1	10.0 ± 20.0	13.3 ± 19.7	18.3 ± 21.4	18.3 ± 21.4	15.0 ± 17.6
	*Rhus trilobata**	0	0	0	0	0	
	Quercus turbinella	1.7 ± 4.1	5.0 ± 12.2	8.3 ± 16.0	3.3 ± 8.2	8.3 ± 20.4	5.0 ± 12.2
	Yucca baccata	1.7 ± 4.1	3.3 ± 5.2	3.3 ± 5.2	0	0	0

Appendix 1b. Mean and standard deviation (std) of understory plant species frequency (% occurrence on plots) at Goose Ponds site.

			Treatment			Control	
Functional Group	Species	2002 mean	2004 mean	2005 mean	2002 mean	2004 mean	2005 mean
Annual	*Argemone muntia*	0	1.7 ± 4.1	1.7 ± 4.1	0	0	0
	Chamaesyce serpyllifolia	0	0	3.3 ± 5.2	0	0	0
	Chenopodium album	0	0	0	1.7 ± 4.1	10.0 ± 12.6	0
	Chenopodium leptophyllum	0	3.3 ± 8.2	5.0 ± 5.5	0	1.7 ± 4.1	18.3 ± 19.4
	Conyza canadensis	0	0	0	0	0	8.3 ± 9.8
	Descurainia incana	0	0	0	0	0	0
	Descurainia sp.	0	0	13.3 ± 17.5	0	0	23.3 ± 18.6
	Gayophytum diffusum	0	0	1.7 ± 4.1	0	0	1.7 ± 4.1
	Gilia ophthalmoides	0	3.3 ± 5.2	3.3 ± 5.2	0	6.7 ± 12.1	28.3 ± 20.4
	Lepidium densiflorum	0	0	1.7 ± 4.1	0	0	1.7 ± 4.1
	Lepidium sp.	0	0	0	0	0	1.7 ± 4.1
	Lupinus kingii	0	1.7 ± 4.1	5.0 ± 5.5	0	1.7 ± 4.1	0
	Nicotiana attenuata	0	3.3 ± 8.2	0	0	1.7 ± 4.1	1.7 ± 4.1
	Phlox gracilis	0	0	1.7 ± 4.1	0	3.3 ± 8.2	0
	Polygonum douglasii	1.7 ± 4.1	1.7 ± 4.1	6.7 ± 8.2	0	0	3.3 ± 5.2
	Solanum triflorum	0	0	±	0	0	1.7 ± 4.1
Exotic	*Bromus rubens*	0	0	0	0	0	1.7 ± 4.1
	Bromus tectorum	0	1.7 ± 4.1	5.0 ± 8.4	0	6.7 ± 8.2	11.7 ± 7.5
	Convolvulus arvensis	0	1.7 ± 4.1	0	0	0	0
	Lactuca serriola	0	10.0 ± 11.0	60.0 ± 36.3	0	21.7 ± 28.6	38.3 ± 22.3

Appendix 1b (cont.). Mean and standard deviation (std) of understory plant species frequency (% occurrence on plots) at Goose Ponds site.

		Treatment			Control		
		2002	2004	2005	2002	2004	2005
Functional Group	**Species**	**mean**	**mean**	**mean**	**mean**	**mean**	**mean**
Exotic	*Salsola tragus*	0	0	0	0	1.7 ± 4.1	0
	Sisymbrium altissimum	0	0	0	0	1.7 ± 4.1	0
	Tragopogon dubius	0	1.7 ± 4.1	1.7 ± 4.1	0	0	0
	Verbascum thapsus	0	0	0	0	1.7 ± 4.1	1.7 ± 4.1
Graminoid	*Bouteloua curtipendula**	0	0	40.0 ± 12.6	0	0	0
	*Bouteloua gracilis**	3.3 ± 5.2	10.0 ± 15.5	20.0 ± 11.0	0	0	0
	*Elymus elymoides**	0	10.0 ± 6.3	58.3 ± 20.4	0	11.7 ± 13.3	21.7 ± 17.2
	Hordeum jubatum	0	0	1.7 ± 4.1	0	1.7 ± 4.1	1.7 ± 4.1
	Poa fendleriana	0	0	6.7 ± 5.2	0	0	1.7 ± 4.1
	*Sporobolus cryptandrus**	0	0	16.7 ± 22.5	0	0	0
Perennial	*Agoseris glauca*	0	0	1.7 ± 4.1	0	0	0
	Arabis fendleri	0	30.0 ± 16.7	55.0 ± 26.6	0	15.0 ± 13.8	36.7 ± 33.3
	Chaenactis douglasii	0	28.3 ± 24.0	58.3 ± 39.7	0	25.0 ± 30.8	56.7 ± 31.4
	Conopholis alpina	0	0	0	0	0	1.7 ± 4.1
	Convolvulus equitans	0	0	3.3 ± 5.2	0	0	0
	Erigeron divergens	0	0	0	0	3.3 ± 5.2	0
	Eriogonum racemosum	0	0	0	0	1.7 ± 4.1	0
	Eriogonum umbellatum	20.0 ± 19.0	15.0 ± 12.2	18.3 ± 18.3	5.0 ± 8.4	5.0 ± 8.3	8.3 ± 16.0
	Frasera albomarginata	0	0	1.7 ± 4.1	0	0	3.3 ± 8.2
	Fritillaria atropurpurea	0	0	0	0	0	1.7 ± 4.1
	Hymenopappus filifolius	3.3 ± 8.2	0	0	3.3 ± 5.2	0	0
	Lepidium montanum	0	0	0	0	1.7 ± 4.1	0
	Lotus utahensis	0	5.0 ± 5.5	0	0	1.7 ± 4.1	0
	Lotus wrightii	0	1.7 ± 4.1	6.7 ± 10.3	0	3.3 ± 5.2	0
	*Lupinus argenteus**	0	0	11.7 ± 13.3	0	0	0
	Lupinus sp.	0	0	3.3 ± 8.2	0	0	0
	Marrubium vulgare	0	0	0	0	1.7 ± 4.1	1.7 ± 4.1
	Mirablis multiflora	0	0	0	0	0	1.7 ± 4.1
	Mirablis oxybaphoides	0	0	0	0	1.7 ± 4.1	0
	Opuntia sp	0	1.7 ± 4.1	3.3 ± 5.2	0	6.7 ± 10.3	1.7 ± 4.1
	Opuntia whipplei	0	0	0	0	0	3.3 ± 5.2
	Orobanche fasiculata	0	0	1.7 ± 4.1	0	0	0
	Packera multilobata	0	1.7 ± 4.1	0	0	0	0
	Penstemon linarioides	1.7 ± 4.1	1.7 ± 4.1	3.3 ± 5.2	0	0	0
	Penstemon palmeri	0	1.7 ± 4.1	1.7 ± 4.1	1.7 ± 4.1	15.0 ± 20.1	30.0 ± 36.9
	Phaseolus angustissimus	1.7 ± 4.1	20.0 ± 21.0	31.7 ± 23.2	3.3 ± 5.2	31.7 ± 21.4	46.7 ± 27.3
	Phlox longifolia	0	1.7 ± 4.1	5.0 ± 5.5	0	0	0
	Psoralidium tenuiflorum	0	1.7 ± 4.1	1.7 ± 4.1	0	0	0
	Solidago sp	0	0	0	0	3.3 ± 8.2	0
	Sphaeralcea parvifolia	0	11.7 ± 14.8	10.0 ± 12.6	0	16.7 ± 16.3	30.0 ± 36.9
	Townsendia exscapa	0	0	0	0	1.7 ± 4.1	0
Shrub	*Amelanchier utahensis**	0	0	3.3 ± 8.2	0	0	0
	Artemisia tridentata	2.0 ± 4.5	3.3 ± 5.2	3.2 ± 5.2	5.0 ± 5.4	15.0 ± 12.2	25.0 ± 20.7
	Purshia mexiccana	16.7 ± 12.1	15.0 ± 13.8	20.0 ± 12.6	20.0 ± 11.0	20.0 ± 16.7	20.0 ± 16.7
	Quercus turbinella	0	0	1.7 ± 4.1	0	0	0
	Ribes cereum	0	0	0	0	0	1.7 ± 4.1

2. Ecological Fuels Management in a Pinyon-Juniper Woodland of Northern Arizona

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Abstract

The following is a progress report describing a study of ecological responses to fuels management treatments in a pinyon-juniper ecosystem of northern Arizona. The project was initiated in spring 2004, pretreatment data have been collected, and implementation of fuels management treatments (i.e., thinning and prescribed fire) is underway.

Introduction

Across large areas of the western United States, woodlands dominated by pinyon (esp. *Pinus edulis, P. monophylla*) and juniper (esp. *Juniperus osteosperma, J. monosperma*) have expanded their range and increased in tree density since the late 1800s (Gottfried et al. 1995, West 1999). Intensive cattle grazing and climatic factors are thought to be the primary causes of structural changes although exclusion of naturally occurring fires and abandonment by native peoples are also considered important on some sites (Gottfried et al. 1995). Dense overstory conditions and low abundance and diversity of understory flora as a result of past land management decisions are issues of concern for conservation and fire management. Presently, efforts are underway to improve ecosystem health in these woodlands and reduce potential for severe wildfire. Actions to achieve these goals include thinning small-diameter trees to reduce overstory density, lopping and scattering slash to improve soil and microsite conditions, seeding to reintroduce native plant diversity, and use of prescribed fire to reduce fuel hazards (Jacobs and Gatewood 1999, Brockway et al. 2002, Huffman et al. Ch. 1 this report).

Natural fire patterns in pinyon-juniper woodlands are uncertain (Romme et al. 2003, Baker and Shinneman 2004). Although surface fire is an important management tool for reducing downed woody fuel hazards, its historical role in maintaining ecological integrity of pinyon-juniper ecosystems is poorly understood. Prescribed fire in pinyon-juniper woodlands may produce ecologically beneficial outcomes where plant populations are vigorous and composed of fire-resilient species. Under these circumstances, prescribed burning treatments may encourage plant growth and reproduction through mineralization of nutrients and rejuvenation of decadent resprouting species (Everett and Sharrow 1983). However, where communities are dominated by fire sensitive species, or where competitive effects have reduced plant vigor, prescribed fire may negatively impact populations (Perry 1993). Although previous research has examined ecological responses to prescribed fire and live fuel reduction in pinyon-juniper woodlands, few studies have designed side-by-side comparisons of these treatments in factorial combination.

The purpose of this experiment is to study the effects of pinyon-juniper fuel reduction treatments on various ecosystem characteristics such as understory vegetation, overstory structure, and fuels. We are testing individual components (i.e., thinning and prescribed fire) of a fuel reduction project implemented by the Tusayan Ranger District of the Kaibab National Forest. Specific objectives of the study are to do the following: (1) Test effects of overstory thinning and prescribed fire on ecosystem attributes including overstory structure and composition, tree growth, understory plant community abundance and diversity (species richness and evenness), and fuel load dynamics.

Methods

Study Site

The study are is on the Tusayan Ranger District of the Kaibab National Forest in northern Arizona (Fig. 1). The site comprises sections 2, 3, and 4, T30N, R2E, Gila-Salt River baseline and meridian (latitude 36º01'24" to 35º59'43", longitude 112º11'55" to 112º07'38"). Average annual precipitation is about 430 mm, typically falling as snow in late winter and rain in late summer. Average minimum temperature is 0º C and maximum is 17º C. Soils are fine to coarse textured and derived from limestone parent material.

Vegetation at the site is classified as Great Basin Conifer Woodland (Brown 1994). Overstory composition is mainly of pinyon (*P. edulis*) and juniper (*J. osteosperma*) although scattered individuals and stands of ponderosa pine (*Pinus ponderosa)* and Gambel oak (*Quercus gambelii*) also are present. Understory communities are comprised mainly of shrubs such as cliffrose (*Purshia mexicana*), big sagebrush (*Artemisia tridentata*), and mountain mahogany (*Cercocarpus montanus*), few grasses such as blue grama (*Bouteloua gracilis*) and Muhlenbergia (*Muhlenbergia* spp.), and forbs such as buckwheat (*Eriogonum* spp.) and gilia (*Ipomopsis* spp.).

Field Measurements

Methods for the establishment and measurement of permanent monitoring plots follow procedures currently in use by ERI on other ecological restoration projects (see Waltz et al. 2001) with certain modifications. These procedures are used for pretreatment and posttreatment sampling. Additional measurements including crown scorch and charring on pinyon and juniper trees as well as fire severity measurements will be added for post-burn sampling. A completely randomized block design with fully factorial treatment combinations of overstory thinning and prescribed fire are being used to assess the ecological effects of fuels management treatments on pinyon-juniper woodlands at the study site. Four unique treatments are being generated from categorical levels of the factors "overstory thinning" and "prescribed fire". The study area is subdivided into six experimental blocks (70-200 ha each (173-494 ac)), delineated on the bases of stand condition and relative species composition. Blocks and a systematic grid of 1-hectare experimental units (200-m (656-ft) spacing) were overlaid on a study area map using a GIS (ArcView 3.3). Experimental units were randomly selected within blocks and treatments were randomly assigned to experimental units. The result will be a balanced randomized block design with two treatment replicates per block.

Centered on each experimental unit, one 0.04-ha (0.1 ac) circular sample plot was established. Plot centers were established with steel rebar and geo-referenced for long-term monitoring. On these plots, overstory, understory, and fuels data were recorded in order to describe pretreatment structure, composition, and response to restoration treatments. Additionally, photo-points were established to document visual changes. On sample plots, all live trees greater than breast height (1.37 m (4.5 ft)) on plots were tallied by species, measured for total height and tagged for remeasurement. Pinyon and juniper stems were measured for diameter at root collar (DRC) and crown radius. Pinyon and juniper trees greater than 25 cm DRC were presumed to be greater than 130 years of age. For these trees, diameter of the largest stem at 40 cm above the ground (DSH) was measured, and increment cores were collected. Increment cores were also collected from a 20% random subsample of trees less than 25 cm (9.8 in) DRC. Ponderosa pine (*Pinus ponderosa*) trees greater than 37.5 cm (14.8 in) diameter at breast height were also cored for age and measured for crown radius. Gambel oak (*Quercus gambelii*) smaller than 10 cm DBH were tallied by condition class (live, standing dead, dead and down, stump) and size class (0.1-5.0 cm (0.04-2.0 in), 5.1-10.0 cm (2.0-3.9 in)). Oak that were larger than 10 cm DBH were tagged and measured for DBH and total height. Dead tree structures (i.e., snags, logs, stumps) were tagged and tallied by condition class as described by Thomas et al. (1979) and Maser et al. (1979) for ponderosa pine and measured for DRC (pinyon and juniper) or DSH (ponderosa pine). Woody surface fuels were tallied and forest floor depth (litter and duff) was measured on 15.24-m (50-ft) transects established in a random direction from plot centers.

Tree seedlings (< 1.37 m in height) and shrubs were tallied on a 100-m^2 (1076-ft^2) circular plot nested within the larger overstory plot. Shrubs were tallied by height and condition classes. Understory sampling transects, 50 meters (164 ft) in length, were oriented parallel to slope and centered on the circular sample plots described above. Transect endpoints were established with steel rebar for long-term monitoring. Along transects on alternating sides, 1-m^2 (10.8 ft^2) quadrats were placed at 5-meter (16.4-ft) intervals (10 quadrats per transect). On each quadrat, cover of herbaceous (non-woody) plant species was recorded. Transects were also used as centerlines for 10-m (32.8 ft) wide sampling belts. On belts, a species list of all plants was recorded.

Two photo-points per plot were established at north and east points on the overstory plot perimeter. Photographs were taken toward plot center with the horizon located in the lower one-third of the field of view. Dry-erase boards were used to document plot number and date of the photos.

Treatments

Overstory thinning prescription will be the same among all one-hectare experimental units selected for thinning (n = 24). This prescription will represent a blend of prescriptions developed by US Forest Service staff for the Topeka project. Tree marking will be done by combined efforts of the Ecological Restoration Institute and Tusayan Ranger District. Thinning will be done by hand using chainsaws. Thinned trees will be lopped to 61 cm (2 feet), and thinning slash will be scattered. Larger fuels may be piled to reduce extreme fuel loads where necessary. Specific prescription details are as follows:

1. All pinyon trees up to 25.4 cm (10 inches) DRC will be thinned.
2. All junipers up to 30.5 cm (12 inches) DRC will be thinned.
3. All ponderosa pine up to 22.9 cm (9 inches) DBH (diameter at breast height; 1.37 m above ground) will be thinned with the exception of yellow-barked individuals, which will be retained.
4. All oak are retained.

Prescribed fire treatments will follow recommendations set forth by staff at the Tusayan Ranger District and Kaibab National Forest. Optimally, all experimental units selected for prescribed fire treatments (n = 24) should be burned during the same year and same season. Protection of experimental units selected for "no-burn" treatments will be accomplished using fuel breaks (fire lines) and burn-out techniques. Fire lines, 12 inches in width, will be dug to mineral soil and surround no-burn experimental units. These will be established and maintained using the combined efforts of Tusayan Ranger District, Kaibab National Forest, and Ecological Restoration Institute staff. Pre-burning fuels near fire lines prior to implementation of prescribed fire within Topeka units also will help protect the one-hectare experimental units.

Data Analyses

Data will be analyzed using two-way analysis of variance (ANOVA) and ANOVA for repeated measures. Parameters tested for treatment effects will include means of overstory and understory structure, growth, composition, and fuel loadings. For count data and those that do not adhere to assumptions for classical tests, nonparametric analyses may be used. Results of these analyses will be shared with U.S. Forest service staff and other interested groups. Further, ERI-Forest Service collaborations will be appropriately acknowledged in any data presentations or manuscripts developed from this study.

Schedule

Outlined below is a tentative schedule for sampling and treatment of the experimental units at the Topeka site.

Activity	**Responsibility**	**Date**
Transect layout and pretreatment sampling	ERI staff	*Done*; May – July 2004
Treatment marking	ERI and Forest Service staff	*Done*; July – August 2004
Thinning	Forest Service staff or USFS administered contract	*In progress*; July 2004 – July 2005
Prescribed Burning	Forest Service staff or USFS administered contract	TBA
Posttreatment (full) sampling	ERI field staff	TBA

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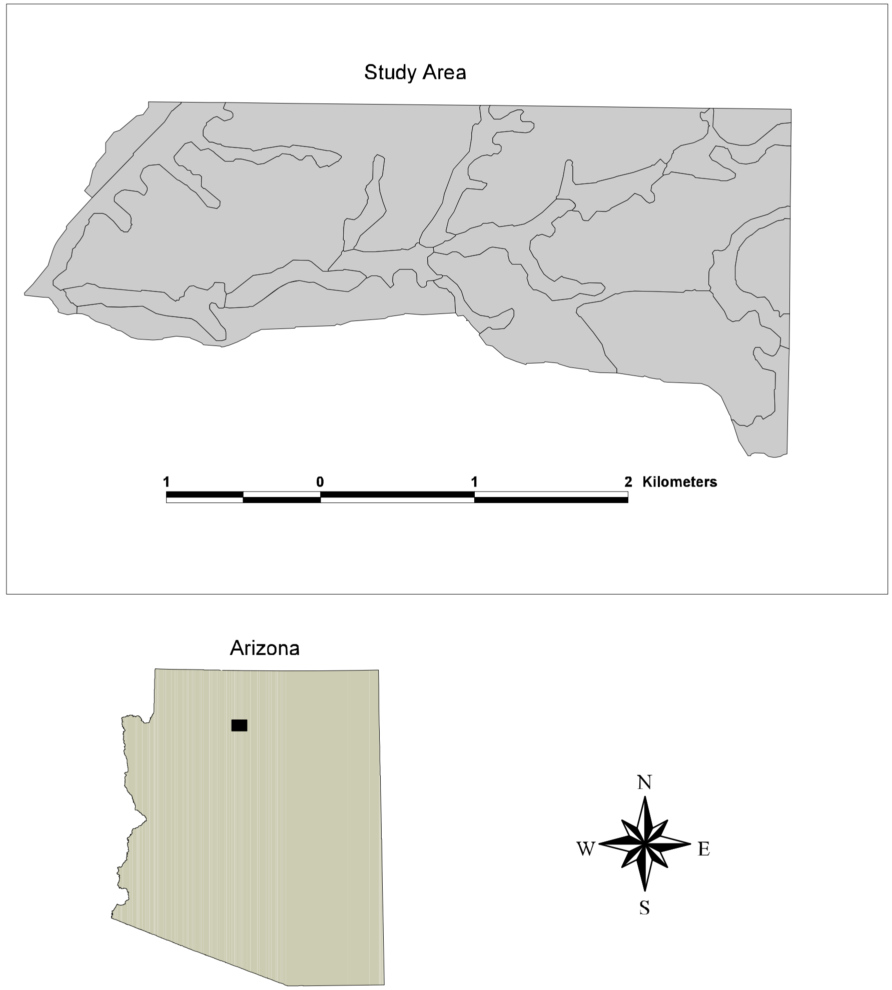

Figure 1. Map of fuels management study area on the Tusayan Ranger District of the Kaibab National Forest in Arizona.

3. Does slash help retain soil and increase grass cover in a pinyon-juniper woodland?

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Abstract

We established an experiment within the interspaces of pinyon (*Pinus edulis*) – juniper (*Juniperous osteosperma*) tree canopies to examine the effects of slash treatments on soil stability, soil chemistry, soil biota and herbaceous emergence. The study site is in the Grand Canyon-Parashant National Monument, at Mt. Trumbull, Arizona. Our goal was to create favorable microsites or "islands" of elevated soil fertility for graminoid seed establishment. In 2003, we selected thirty ≈ 0.04ha (0.1 ac) interspaces across cinder and sedimentary soil types. The four treatments consisted of a control (no treatment), seed, slash, and slash/seed, each measured on 1-m^2 plots. Mean sediment loss within two years equaled 2.9 cm (1.1in) in the control vs. 1.1 cm (.4in) in the slash treatment. Slash treatments also significantly increased litter cover and decreased soil exposure. Soil moisture increased slightly within slash treatments compared to non slash treatments. Available $N0_3$ decreased significantly within slash treatments, possibly indicating increased microbial activity. Arbuscular mycorrhizal fungi and microbial carbon biomass increase significantly within slash treatments compared to non-slash treatments. Graminoid cover increased over 200% within slash and seed treatments compared to seed-only treatments. Our data shows that slash treatments do create favorable microsites for herbaceous emergence, therefore contributing to the recovery of degraded microsites.

Introduction

Pinyon-juniper savannas, woodlands and forest cover, 24 million ha throughout the western United States (West 1984). Approximately 40% or 9.6 million ha of the total pinyon-juniper lands occurs within Arizona and New Mexico (Powell et al. 1994). Today's pinyon-juniper woodlands are composed of a series of patch and matrix assemblages where the woodlands are the dominating matrix and embedded within the intercanopy spaces are the species that make up the depleted understory patches (Taush 1999). Tree density has increased considerably in pinyon-juniper woodlands since the late 1800's (Tausch et al. 1981, West 1999). The advance of juniper trees into adjacent intercanopy spaces has effected the structure, composition and key functional processes within pinyon-juniper ecosystems. This shift in pinyon-juniper community structure is a cumulative result that seem to be linked to climatic shifts favoring the establishment of pinyon and juniper trees, heavy livestock grazing pressures and possibly the exclusion of fires that removed young, fire-sensitive trees (Tausch 1999, Jacob and Gatewood 1999, West 1998).

Pinyon-juniper communities have been steadily increasing since the early Holocene (Nowak et al. 1994, Miller and Wigand 1994), but it was not until the late 1800's, after European settlers arrived did pinyon-juniper populations explode (Taush 1999). Intensive grazing pressures have reduced the abundance and richness of many understory species (Taush and West 1995, Tausch et al. 1981). This reduction of understory species has allowed woody species to encroach into intercanopy spaces (Miller and Wigand 1994). Increases in woody cover have contributed significantly to the depletion of soil moisture and available nutrients within the soil (Breshears et al. 1997, Schlesinger 1990). With the removal of understory species, sufficient ground cover to stabilize soil resources has been reduced. These exposed soils are then susceptible to sheet erosion, reduced infiltration, and the inability to form soil aggregates (Wood et al. 1987). Wood et al. (1987) determined total herbaceous ground cover to be the single most important variable in decreasing sediment loss. In addition, soil seed banks are depleted with the loss of surface soils, thereby compounding the problem (Jacobs and Gatewood 1999).

Numerous studies have shown, as juniper trees are removed from a site, herbaceous vegetation increases significantly, suggesting that juniper limits understory vegetation by controlling soil resources (Davenport et al. 1996, Miller and Wigand 1994, Everett and Sharrow 1985). These results are only possible if sufficient native understory species pools exist, both above ground and in the seed bank. In many cases pinyon-juniper sites are in an advance state of degradation where understory species have been depleted for many years and the seedbank is depauperate, resulting in a decreased likelihood of natural recovery following tree reduction. Unless these woodlands are artificially seeded, natural recovery may be slow resulting in an increased time span until the desired future conditions are reached. However seeding alone within these degraded woodlands is not the answer, due to the susceptibility of erosion.

Dispersing slash into the intercanopy spaces of pinyon-juniper woodlands is a type of canopy utilization that has been shown to abate soil loss. (Brockway et al. 2002, Jacobs and Gatewood 1999). Hastings et al. (2003) demonstrated that sediment yields had significantly decreased were slash treatments had been applied. Four years after slash treatments were applied, a four-fold increase in herbaceous cover was observed. Brockway et al. (2002) concluded that scattering slash across harvested sites would promote herbaceous growth by fostering a microsite that stablizes the soil surface. Jacobs and Gatewood (1999) found a seven fold increase in herbaceous cover within the second year following slash treatments. These results occurred throughout a variety of degraded pinyon-juniper communities within New Mexico. In these cases, understory response to slash treatments occurred naturally, that is seeding was not necessary in order to achieve these results. Jacobs and Gatewood (1999) found no significant increases in total grass cover resulting from a seeding treatment.

We hypothesized that for native herbaceous species (primarily grasses) to recover within our study site, they needed to be artificially introduced (Huffman et al. Ch 1. this report). We were specifically interested in graminoid species because root biomass associated with grasses are likely to help in soil stabilization (West 1999). Seedling emergence heavily depends on soil water availability and the ability of the soil surface to increase soil water potential (Chambers 2000). Microsites that can provide wind barriers that aid in trapping and retaining seeds, and improve soil water potential will most likely increase seed germination and seedling emergence (Chambers 2000, Harper et al. 1965). Due to escalating seed costs and the depletion of soil resources, more effective means of creating ideal microsite conditions are needed to stimulate seed germination and establishment. Seed germination and establishment are directly correlated with the number of seeds in favorable microsite seedbeds, rather than the total number of available seeds (Harper 1977, Harper et al. 1965). By creating suitable microsites or “islands” of elevated soil fertility for herbaceous species, we may contribute to the recovery of degraded pinyon-juniper communities.

The specific objectives of this study were to: (1) determine the effects of slash treatments on soil stabilization; (2) determine the effects of slash amendments on soil properties; (3) determine the effects of slash treatments on seedling emergence, specifically on grass species emergence. We hypothesized that slash amendments would result in decreased soil loss, thereby retaining essential soil nutrients and moisture. In addition, increased levels of soil moisture and nutrients, resulting from slash amendments could provide more optimal conditions for microbial activity that could eventually create a suitable condition for seedling emergence and eventually establishment.

Methods

Study Site

This research was conducted from August 2003 through 2005, at two sites located within the Grand Canyon-Parashant National Monument near Mount Trumbull, Arizona. The Craig Ranch site (CR), is approximately 4 kilometers (2.5 mi.) north of Nixon Spring Station; at latitude 36N 26' 01" and longitude 113W 09' 40". Soils at CR are shallow to deep gravelly sandy loam to cobbly clays derived from limestone, basalt and sandstone. The Goose Ponds (GP) site is located approximately 5 km (3.1 mi.) northwest of Nixon Spring Station at latitude 36N 24' 46", and longitude 113W 12' 15".

Soils at GP are shallow to deep cindery loams that are derived from basalt and cinder. The sites range in elevation from approximately 1900-1950 m (6270-6435 ft) above sea level. This region of Arizona has a semiarid climate where average January minimum temperature is -5.1˚C (41.2˚F) and average July maximum temperature is 31.0˚C (87.8˚F). Mean annual precipitation is 429mm (16.9in), though there is considerable annual variation (Fig. 1). Precipitation patterns follow a bi-modal distribution, including monsoon rains in July and August with snowfall in the winter months followed by a distinct dry period in May and June. All climate data are from a 13-year average (1992-2005) based on information recorded at the Nixon Flats Remote Automated Weather Station (RAWS) site at Mt. Trumbull, AZ (1980m (6700ft)). If data were not available from the Nixon Flats site, then data were supplemented by data from the Mt. Logan RAWS site. Vegetation at the sites is classified as Great Basin Cold Temperature Woodlands (Brown 1994). Dense mixed-aged pinyon pine (*Pinus edulis*) and juniper (*Juniperus osteosperma*) dominate the two sites. Herbaceous communities are sparse but common species consist of perennial grasses: *Bouteloua curtipendula*, *Bouteloua gracilis*, *Aristida purpurea*; perennial forbs: *Chamaesyce albomarginata*, *Eriogonum corymbosum* and *Psoralidium tenuiflorum*; shrubs: *Purshia mexicana*.

Huffman et al. Ch.1. (this report) characterized the two sites as ecologically degraded in respect to: low plant species diversity and abundance (cover <7 %); reduced soil O horizons within intercanopy spaces; and overly dense overstory conditions. Aerial photos from 1940 to 1992 show the advance of trees and densification of woodland conditions. This change may be directly correlated with intensive grazing pressures within the area (Huffman et al. Ch 1. this report). Narrative rangeland inspection reports indicate that the area had been severely overgrazed before the 1900's and continued to be overgrazed throughout the 1960's (unpublished report, BLM District Office, St. George, UT). It is therefore likely that intensive grazing pressure throughout the late 1800's to mid 1900's had a direct impact on the current denuded understory condition.

Experimental Design

This research was restricted to areas between pinyon and juniper trees (interspaces). The study design is a 2 x 2 full factorial design with two levels of seeding (no seed, seed) and two levels of slash treatment (no slash, slash). The two sites were selected to represent two distinct soil types (cinder and sandy-loam) and 15 replicates were chosen within each site. At each interspace, four $1m^2$(1m x1m) permanent plots were established. The plots within each interspace were spaced 2m apart from each other to minimize between plot influences. Plots were also located at least 3m from the edge of a tree crown to eliminate influences created by the accumulation of organic material underneath tree canopies. Each plot was randomly assigned one of four treatments within each interspace: (1) control; (2) seed; (3) slash addition without seed addition; (4) slash addition with seed addition. Treatments were established in early August 2003 to coincide with monsoonal rains. Seeded treatments consisted of a mixture of four native grass species (Table 1). Selection of native graminoid seeds was based on local occurrence, baseline data from previous local studies and herbaceous community data reports from nearby relict sites (Huffman et al Ch 1. this report). Seeds were purchased from the nearest possible seed supplier: Arizona Native Plant and Seed in Flagstaff, Arizona. Seeds were sown at a rate of 9.72g (.3oz) per $1m^2$ plot. All seeded species were sown at the same rate. Weight of slash (limbs and tops) added was 9.1kg (20lbs, wet weight). Slash diameters were approximately 8cm (3in) or less in diameter and 1m (3ft) or less in length. Only juniper trees were evenly distributed across slash treatments.

Sediment Yield methodology

Sediment loss was estimated through erosion bridge measurements (Shakesby 1993, Brockway et al. 2002). Within each plot, two permanent stakes 1m (3.3ft) apart was cemented approximately 5cm (2in) above the ground. A piece of angle iron was place on top of the two stakes. Three fixed points, equally space within the length of the angle iron were identified to measure the distance from the bridge (angle iron) to the soil surface. A 15cm square with a built-in level was used to precisely measure this distance. The three fixed points were then averaged at the plot level. Data on sediment loss were summarized for each plot and analyzed for treatment effect as the difference in sediment change from the previous year.

Sampling Methodology

Each of the 120 1m x 1m permanent understory plots was surveyed for vegetation and sediment loss during August 2003 (pretreatment), August 2004 (post treatment) and August 2005 (post treatment). Herbaceous species were identified and counted within each plot. In addition, foliar cover for each species, soil particle size and O_i layer (litter) were determined using the ocular estimate method to the nearest 0.1%. Plant nomenclature followed the USDA Plants Database (USDA, NRC 2004). Soil particles were categorized into three size groups, less than 2mm, between 2mm and 40mm, and greater than 40mm. Soil particles data was only collected in 2003, prior to treatment. Data on vegetation and substrate variables were summarized for each plot and analyzed for treatment and year effects.

Soil Chemical Field and Laboratory Methodology

Three soil samples were randomly collected within each $1m^2$ plot in August 2003 and 2004. The three soils samples from each plot were then combined (homogenized) together to be analyzed for soil abiotic and arbuscular mycorrhizae fungi potential. Soils were collected to a depth of 10cm (3.9in) using a 4.2cm (1.7in) diameter soil core. To avoid seasonal variation, all soils were collected in August 2003 and 2004, concurrently with vegetation sampling. Soil samples from each plot were analyzed for pH, % total N, % organic C, % soil moisture, and NO_3-N and NH_4-N. To conduct all chemical analyses, soils were passed through a 2mm sieve. Soil pH was determined in 1:1 slurry using a pH meter. Total N and organic C were determined using a FLASH EA 1112 Elemental analyzer. Percent soil moisture was determined gravimetrically. NO_3-N and NH_4-N were determined by KCl extraction of freshly collected soil by automated colorimetry using a Technicon autoanalyzer (Parkinson and Allen 1975). NO_3-N and NH_4-N analyses were only conducted on posttreatment soil samples (collected in August 2004).

Arbuscular Mycorrhizael Fungi and Laboratory Methodology

Bait-plant bioassays were used to quantify the relative amounts of infective propagules of arbuscular mycorrhizael fungi (AMF) within our soil samples. To avoid seasonal variation, all soils were collected in August 2003 and 2004, concurrently with vegetation sampling. Soil samples were divided for soil chemical analysis and AMF Bait-plant bioassay can detect all types of viable mycorrhizal fungal propagules and therefore is considered a more accurate method for quantifying total density of AMF propagules than direct counts of sporocarps, spores, or colonized root lengths (Brundrett & Abbott 1994). For each plot, soils were collected to a depth of 10cm (3.94in) and within 24 hrs placed into 4cm (1.57in) diameter by 20cm (7.87in) deep Conetainers (Stuewe and Sons, Inc., Corvallis, OR, U.S.A.). Zea mays L. (corn) was used as the bait-plant to determine the amount of infective AMF propagules. Corn was used as the host plant, because it grows fast, uniformly and is mycotropic with many AMF species. Corn seeds were germinated, planted into freshly collected soils, and then harvested after 5 weeks or before roots became root bound within the conetainers. Corn roots were then cut into 2.5cm (.98in) segments and randomly subsampled. Subsamples were cleared using a KOH solution and stained in .5% Shaeffer Ink (Vierheilin, et al. 1998). Segments of the root length containing AMF structures were quantified by grindline intersect method using a dissecting microscope (Giovannetti & Mosse 1980).

Microbial Biomass Carbon and Laboratory Methodology

Samples for microbial biomass carbon (C) was collected only within control and slash treatments in August 2004. All mineral soils for microbial biomass C were collected separately form other soil attributes at a depth of 10cm (3.9in), in August 2004. Soil microbial C was determined using the chloroform ($CHCl_3$) fumigation-extraction method (Brookes et al., 1985; Vance et al., 1987; Haubensak et al., 2002). Approximately 30g (1.06oz) of sieved, field-moist soil were extracted with 100mL (3.38 fluid oz) of 0.5 M K_2SO_4 and 30g (1.06oz) of mineral soil was also placed inside a dessicator with a beaker containing 30mL (1.01 fluid oz) of $CHCl_3$. The dessicator was repeatedly evacuated to boil the $CHCl_3$ and then left under vacuum for 5 days (Haubensak et al., 2002). After 5 days, the $CHCl_3$ was removed from the soil by repeated evacuations and then the soil subsamples were immediately extracted with 100mL (3.38oz) of 0.5 M K_2SO_4. Extracts were mechanically shaken for one hour, filtered with Whatman #1 filters (pre-leached with deionized water), and frozen until analysis. Organic C concentrations in unfumigated and fumigated extracts were determined by ultraviolet-enhanced persulfate oxidation using a Dohrmann DC-80 Carbon Analyzer with infrared detection (Tekmar-Dohrmann, Cincinnati, OH, USA). Microbial C was calculated by subtracting organic C in the unfumigated extracts from organic C in the fumigated extracts and dividing by a k_{EC} of 0.39 (Sparling et al., 1990).

Statistical Analyses

All data for dependent variables were summarized as means and variance for each of the four treatments within two different sites (n=15 for each site). We used analysis of variance (ANOVA) to determine treatment and year effects on each dependent variable at a alpha =0.05. The Shapiro-Wilk test was used to test data for normality, and Levene's test was used to test for homoscedasticity of the variance. Prior to analysis, NO_3-N data and AMP data were square root transformed to meet the normality assumption. Tukey's Honestly Significant Difference (HSD) test was used to make multiple comparisons of all treatment means following a significant ANOVA result. No tests were applicable for repeated measure data that does not met ANOVA assumptions. Therefore Kruska-Wallis nonparametric tests were used to compare differences between treatments within individual years for dependent variables that did not meet the normality assumption. Significance for nonparametric tests was based on a family wide alpha =0.05, and individual significant treatment contrasts were adjusted based on sequential Bonferonni adjustments. All analyses were performed using SAS JMP software (SAS Institute 2002).

Results

Litter Cover, Soil Exposure and Soil particle Size

Prior to treatment, no differences were evident in litter cover and soil exposure across both sites in 2003 (Table 2). Bare soil composed 99% of the interspaces and litter cover was less than 0.4%. One year after treatments, litter cover within slash treatments significantly increased compared to non-slash treatments across both sites (Table 2). Conversely, the percentage of bare soil significantly decreased within slash treatments compared to non-slash treatments (Table 2). Soil particle composition was different between sites (Fig 2). Soils were relatively finer in particle size at the CP site compared to the GP site. GP soils were primarily composed of cobble (2mm-40mm) size soil particles.

Sediment Yields

Associated with this increase in litter coverage and decline in exposed soil surfaces within slash treatments, there was a significant decrease in sediment loss when compared to the non-slash treatments, at both sites (Fig. 3). Two year mean sediment loss among slash treatments at the CR site was 10.0mm (.4in) compared to 28.5mm (1.1in) mean soil loss among non-slash treatments. Mean sediment loss within two years among slash treatments at the GP site was 12.0mm (.5in) and 30.3mm (1.2in) within non-slash treatments. In two years, average sediment loss among non-slash treatments was 276% greater than slash treatments. Considerable year to year variation did occur when assessing sediment loss. When comparing sediment loss from 2003-2004 verse 2004-2005 there was a significant reduction in sediment movement. Average sediment loss in 2003-2004 across both sites was 13.3mm (.5in) and 22.2mm (.9in) within slash treatment and non-slash treatments. In 2004-2005, there was accumulations in sediment within slash treatments, where as sediment continue to be loss within non-slash treatments. Average sediment gain across both sites was 2.2mm (.1in) within slash treatment and average sediment loss within non-slash treatments was 7.2mm (.3in).

Soil Properties

In August 2003, prior to treatments, there were no significant differences in pH, total N, organic C, and percent soil moisture between plots within each site (Table 3). No pretreatment data was collected for NO_3-N and NH_4-N. We assumed that because no pretreatment differences existed for total N, that there would be no differences for NO_3-N and NH_4-N. Soil pH for both CR and GP sites were slightly alkaline and relatively very little organic C and total N were present within the soil. Average soil moisture for the CR site was 3.02 % and 2.24 % within the GP site.

There were no significant changes in soil pH, organic C and total N as a result of treatment effects (Table 3). There were no significant differences between treatments on either site for soil moisture though increasing trends were observed within slash treatments compared to non-slash treatments (Fig. 4). There were also no significant differences in NH_4-N between treatments (Fig. 5). However, at both sites, NO_3-N was significantly lower in slash treatments than in non-slash treatments (Fig. 5).

Soil Biota

Micorrhizal fungi colonized all bait-plants within the 2003 and 2004 sampling periods. Prior to treatments, no statistically significant differences in relative root colonization with arbuscular mycorrhizal fungi potential (AMP) were detected between plots, at CR ($p>.8226$) and GP ($p>.6583$). On average AMP within CR and GP were 11.26 % and 11.67 % before treatment. Following treatments AMP was significantly greater in slash treatments compared to non-slash treatments across both sites (Fig. 6). There was over two times more root colonization with AM fungi in slash treatments verse non-slash treatments at both CR and GP sites. No pretreatment data were collected for microbial C. Samples were only taken from control and slash only plots in 2004. Microbial C differed significantly between control plots and slash only plots at both CR and GP sites (Fig. 7).

Seeded Species Response

No statistically significant differences in species cover or density were detected between plots, prior to treatment at either site. Total understory vegetation was sparse at both sites (<3%). Three out of the four species to be seeded were found within the CR site prior to treatments in Aug. 2003. The combined cover of the graminoid species to be seeded at the CR site averaged 0.03% and the number of individuals was less than 1 plant /m^2 prior to treatment. Within the GP site, no seeded species were detected within plots prior to treatment, though three out of the four species were observed within the area. Following seed and slash treatments, the foliar cover of seeded species was significantly greater in seed and slash plots compared to the control plots and seed only plots (Fig. 8). At the CR site, seeded species cover for control plots increased slightly, whereas seeded plots showed a 7-fold increase and seed and slash plots increased 13-folds. Only seed and slash plots significantly increased in seeded species cover within one growing season ($p<0.0001$). Two years after treatments were applied; seeded species cover continued to increase within seed and slash treatments. A two fold increase in cover between 2004 and 2005 was detected within seed and slash treatments, where as seed only treatments showed a slight reduction in aerial cover. Slash only treatments showed marginal increases in seeded species cover in 2005 where as aerial cover within control treatments stayed relatively the same. The number of seeded species emergence per m^2 also increased significantly in response to the combination of seed and slash treatments (Fig. 9). All treatments except slash only decreased in seeded species densities when comparing 2004 verse 2005. The species encountered during cover and density measurements were *Elymus elymoides*, *Bouteloua curtipendula* and *Aristida purpurea*. Not all seeded species established equally well (Table 4 and 5). *Elymus elymoides* had the most seedling emergence, whereas *Achnatherum hymenoides* was seeded but was not identified on post treatment measurements.

Treatment effects on seeded species cover and density at the GP site yielded similar results to the CR site. Mean comparisons for both cover and density were significantly different for individual treatment contrasts (Fig. 8 and 9). Seeded species foliar cover increased from no seeded species detected in 2003 to an average of 3.8 % within seed and slash treatments and less than a 0.1% increase in the seed only treatment, one year following the application of treatments. Emergence increased from no individuals detected in 2003 to an average of 46.6 individuals/m^2 on seed and slash treatment plots and 2 individuals/m^2 on seed only plots in 2004. No seeded species were detected within control plots in 2004 at the GP site. Almost a two fold reduction of individuals/m^2 was detected in seed and slash plots when comparing 2004 verse 2005, although relatively little change was detected in aerial cover. Slash treatments increase from no cover in 2004 to an average of .8% cover in 2005. Not all species that were seeded within treatments emerged after two growing season (Table 4 and 5). Once again, *Achnatherum hymenoides* was not evident during the post treatment measurement.

Discussion

Litter Cover, Soil Exposure and Sediment Yield

Scattering slash across plots significantly increased litter cover, which resulted in a decrease in soil exposure across both sites. One year after applying slash treatments, juniper needles began to decompose and new soil aggregates started to form around the juniper needle cast. Litter decomposition rates are highly correlated with the amount of moisture and the litter quality (Murphy et al. 1998). Murphy et al. (1998) found high concentrations of nitrogen and low carbon to nitrogen ratio within juniper litter. These variables are used to describe litter quality. High nutrient concentrations can promote increased rates of microbial respiration, which ultimately drives organic decomposition (Berg et al. 1982b). Therefore, these newly formed soil aggregates could be explained by the increase in microbial activity, and the protection of litter cover guarding against the impact of rain drops which allows soil particles to bind together.

Residual coarse debris also seemed to play an important part in slowing sediment runoff from a plot by creating debris dams. This was observed by the pooling of fine needles against the coarser woody material. Both increases in litter cover and the role of coarse residual debris had a significant impact on sediment loss within plots. Sediment yield responses after two years were on average 273% less within slash treatments compared to control plots across both sites. However, mean sediment loss of 10-12 mm (0.4-0.5 in.) was still being lost where slash had been scattered across interspaces. Although there were statistical significant difference in sediment loss between slash treatments and control treatments ($p<0.0001$), high levels of sediment loss were still being observed. We found that sediment loss seem to be highly variable from year to year. In 2004, there was over four times more sediment loss in non-slash plots compared to 2005. Also in 2004, slash treatments loss on average 13.3 mm of soil where as in 2005 there was an average accumulation of soil within slash treatments. Brockway et al. (2002) also observed a significantly higher increase in litter cover after a variety of slash treatments, but found no correlation of increase litter cover or decrease sediment loss at the intercanopy space scale. They also found no significant difference in sediment loss when comparing slash treatments to controls, but did observe trends of decreasing soil loss. Wood et al. 1987 explained that large bare, connected interspaces have a much lower infiltration rate than neighboring tree mounds therefore becoming major pathways for sediment runoff. This could explain the excess amount of soil loss, even after implementing slash treatments. In contrast, Hasting et al. (2003) found obvious decreases in sediment yields following slash treatments at the watershed scale. Sediment yields in control sites were less than 3 mg/ha compared to .1 mg/ha within treatment sites. Hasting et al. (2003) accounts rainfall erosivity, the ability of rainfall to erode soil, to be the single best variable for predicting sediment loss. More intense rainstorms will produce more surface runoff. Wilcox and Breshears (1995), found that slope had the greatest impact on site erodibility and established that soil runoff in juniper woodlands to be scale dependent; the larger the scale, generally the less erosion. For example, in a hillslope scale study, Wilcox (1994) measured erosion within a variety of small plots. He concluded that there was substantial movement of sediment on the hillside, but much of the sediment did not actually leave the site. And Gifford (1995) points out that if there is substantial sediment production it will likely be redeposit in to riparian area and result in a net benefit to the whole ecosystem.

In our study, slopes within interspaces (n = 30) were approximately less than 5%. The scale at which we measured the rate of soil runoff was at the intercanopy level and we observed a substantial amount of sediment movement both within controls and after slash treatments. If the top 15 cm (5.9 in) of the soil profile is where the majority of the live biota, essential nutrients and viable seed source are stored, then 1.1 cm (0.5 in) of soil movement should be considered an alarming rate. Soil movement within pinyon-juniper woodlands can be highly problematic there for progressively degrading site productivity (Baker et al. 1995). If seeding is to be an option for recovering herbaceous communities within degraded pinyon-juniper woodlands, then soil movement should be reduced at the herbaceous community scale.

Soil Properties

Soil physical and chemical properties exhibited no significant changes one year after treatments were applied though increasing trends were noticeable for organic C, and soil moisture. Organic C showed only slight increases due to slash treatments, but these small differences can be important in facilitating microbial activity and soil structure. As carbon is filtered into the soil this can greatly affect the distribution and structure of pores within the soil that permits the storage and movement of water (Perry et al. 1987). Soil moisture was continuously higher in slash treatments compared to non-slash treatments. The slight increase in moisture was mostly like due to the significant increases in litter cover. Although soil moisture differences were not statistically different, slight differences can affect seed germination and seedling establishment in a system that experiences limited annual precipitation (Lauenroth et al. 1987). No differences were apparent in NH_4-N. However NO_3-N decrease significantly within slash treatments. This decrease can be explained by the immobilization of NO_3-N by increasing microbial activity (Schlesinger 1997). The stability of an ecosystem is determined by plant diversity and species composition while the functions that drive ecosystem stability are the soil resources (Baker et al. 1995).

Soil biota

Significant increases were detected for arbuscular mycorrhizal fungi and microbial biomass C within slash treatments. No significant correlations were made with decreasing amounts of NO_3-N or NH_4-N and increasing amounts of microbial biomass and mycorrizal fungi. Though, it is highly probable that slash treatments increased microbial quantities by increasing soil moisture and essential nutrients. These increases greatly benefit long term soil stability and understory establishment. Mycorrhizal fungi are a major contributing factor to the maintenance of plant biodiversity and to ecosystem function of semi-arid and arid ecosystems, since these environment are often low in nutrient availability to plants (Reeves et al. 1979). It has been suggested that the recovery of disturbed ecosystems may depend upon the establishment of mycorrhizal fungi (Reeves et al. 1979, Allen and Allen 1980, Perry et al. 1987). The availability of these soil nutrients is in large part controlled by the below ground biota, which regulates the mineralization and immobilization of these nutrients (Schlesinger 1997). Soil microbial biomass can be define as the living component of the soil that is primary responsible for litter decomposition, nutrient cycling and energy flow (Wardle 1992).

Seeded Species Response

Seeding response to slash treatments was obvious when compared to control or seed only treatments, 24 months following treatments. Our results indicate that the addition of both seed and slash generated the highest percent cover and the most seedling emergence, whereas direct seeding had relatively very little effect on graminoid emergence and establishment. On average, graminoid cover in our seed and slash plots increased 1.6% and 3.8 % cover, depending on the site. Individual graminoid emergence on average increased from less than 1 individual to 6 and 27 individuals, once again being site specific. Conversely, plots that were only seeded averaged less than 0.2 % cover and less than 2 individuals per m^2. Seeded species accounted for 19% of the total cover and 49% of the total density within seed and slash plots, which substantially affected the overall net increase in graminoid cover, and density. Seeded species clearly responded best when slash was scattered on the plot. Slash only treatments also seem to be effective in capturing wind blown graminoid seeds. Mulch treatments can qualitatively affect the microenvironment that seedling emergence depend on (Chambers 2000). Treatments that can increase water availability and retain more seeds will have the highest probability of seedling emergence (Harper et al. 1965, Chamber 2000)

Different soil types that characterize the two sites seemed to also be an important determinant in seedling emergence. Although site to site comparisons were not tested statistically, general observations were noticed. The coarser soils, derived primary from cinders and basalts (GP site), exhibited seedling emergence 4 orders of magnitude higher than the finer soils (CR site) derived from limestone and sandstone in response to seeded and slash treatments. In addition, the robustness of each individual seedling was far less within the finer soils compared to the cinder soils. This resulted in an average of 3.8% cover in the cinder soil site and 1.6% in seeded species cover within the finer textured site across seed and slash plots. Soil surface characteristics can determine the quantity of seeds trapped and retained within the soils (Chambers 2000). Soils with larger particle sizes usually have higher seed entrapment and retention than smaller particle size soils (Chambers et al. 1991).

Conclusions

The conditions of these particular pinyon-juniper sites are in an advance state of degradation. Our results illustrate that intercanopy spaces are experiencing accelerated erosion thereby promoting the rapid export of soil resources. Soil resources that are essential in fostering soil aggregates and promoting understory vegetation are being lost. The lack of understory cover within intercanopy spaces is not present to aid in the reduction of sediment lost. If these processes continue, we can expect intercanopy spaces to be continued to be encroached upon by tree species with a resulting loss of biodiversity.

Results from our study indicate that slash treatments yield less sediment loss than non-slash treatments, which can aid in the retention of essential soil resources. Seeding alone within intercanopy spaces had no significant increase in the development of understory communities. Slash treatments alone also had no effects on understory development. Seed and slash treatments together significantly promoted regeneration of herbaceous cover and abundance within intercanopy areas. From these results, we recommend land mangers first reduce the lost of sediment within intercanopy spaces, possibly by utilizating material from woody species. Slash treatments should be considered a temporary solution in aiding the recovery of these degraded ecosystems. The establishment of understory vegetation should be considered for the long-term recovery of these degraded ecosystems.

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Table 1. Seeded species and seeding rate applied to seeded treatments.

Functional group	PS Pathway	Seeded Species	Common Name	Seeding rate (g/m^2)
Graminoid	C4	*Aristida purpurea*	purple threeawn	2.43
	C4	*Achnatherum hymenoides*	Indian ricegrass	2.43
	C4	*Bouteloua curtipendula*	blue grama	2.43
	C3	*Elymus elymoides*	western bottle-brush grass	2.43
			total	**9.72**

Table 2. Average litter cover and soil exposure. Different letters within rows indicates significant difference between treatments at α = 0.05. Standard mean error in parenthesis (n=15 within each site).

	Craig Ranch			
	Control	**Seed**	**Slash**	**Slash + Seed**
Litter Cover %				
2003	0.45 (0.40)	0.27 (0.27)	0.28 (0.14)	0.05 (0.04)
2004	2.34 (1.026)**a**	1.17 (.68)**a**	85.97 (2.50)**b**	89.07 (1.64)**b**
Exposed Soil %				
2003	99.55 (0.40)	99.73 (0.27)	99.72 (0.14)	99.95 (0.04)
2004	97.67 (1.03)**c**	98.90 (.69)**c**	14.03 (2.50)**d**	11.60 (1.79)**d**
	Goose Ponds			
	Control	**Seed**	**Slash**	**Slash + Seed**
Litter Cover %				
2003	0.20 (0.13)	0.19 (0.13)	0.40 (0.22)	0.81 (0.37)
2004	1.00 (0.50)**a**	0.60 (0.27)**a**	89.89 (2.37)**b**	93.6 (0.85)**b**
Exposed Soil %				
2003	99.80 (0.13)	99.81 (0.13)	99.60 (0.22)	99.19 (0.37)
2004	99.00 (.50)**a**	99.40 (.27)**a**	10.11 (2.37)**b**	6.4 (0.85)**b**

Table 3. Soil chemistry response to treatments. No significant difference between treatments at α = .05. Standard mean error in parenthesis (n=15 within each site).

	Craig Ranch				**Goose Ponds**			
	Control	**Seed**	**Slash**	**Slash + Seed**	**Control**	**Seed**	**Slash**	**Slash + Seed**
Organic C								
2003	**0.84** (.05)	**0.80** (.05)	**0.80** (.04)	**0.75** (.04)	**1.08** (.09)	**0.92** (.07)	**1.06** (.07)	**1.05** (.06)
2004	**0.65** (.05)	**0.70** (.05)	**0.78** (.05)	**0.80** (.05)	**0.92** (.09)	**0.91** (.09)	**1.08** (.07)	**1.07** (.06)
Total N								
2003	**0.08** (.003)	**0.08** (.002)	**0.08** (.002)	**0.07** (.003)	**0.07** (.007)	**0.06** (.004)	**0.07** (.005)	**0.12** (.05)
2004	**0.06** (.004)	**0.06** (.002)	**0.06** (.003)	**0.07** (.003)	**0.08** (.007)	**0.07** (.006)	**0.08** (.005)	**0.09** (.004)
C:N								
2003	**10.60** (.25)	**10.56** (.36)	**10.53** (.22)	**10.35** (.22)	**14.60** (.29)	**14.32** (.29)	**14.54** (.43)	**14.48** (.40)
2004	**10.83** (.44)	**10.88** (.45)	**10.98** (.35)	**11.61** (.33)	**11.96** (.35)	**11.95** (.36)	**12.60** (.25)	**12.34** (.24)
pH								
2003	**7.45** (.10)	**7.50** (.13)	**7.60** (.13)	**7.60** (.09)	**7.93** (.08)	**7.83** (.06)	**7.93** (.08)	**7.85** (.05)
2004	**7.44** (.21)	**7.29** (.26)	**7.51** (.20)	**7.70** (.20)	**7.51** (.08)	**7.39** (.14)	**7.74** (.14)	**7.85** (.03)

Table 4. Average density (#/ plants/m2) and cover of all seeded species for Craig Ranch Site (12 month after seeding).

Species	Photosynthetic Pathway	Avg. Density (m^2) 2003	Avg. Density (m^2) 2004	Avg. Density (m^2) 2005	Avg. Cover (%) 2003	Avg. Cover (%) 2004	Avg. Cover (%) 2005
Elymus elymoides	C3						
Control		0	**.20** (.20)	0	0	**.01** (.01)	0
Seed		0	**1.73** (1.46)	**.02** (.02)	0	**.13** (.13)	**.07** (.07)
Slash		0	**.40** (.24)	**1.80** (.83)	0	**.06** (.05)	**.26** (.13)
Seed + Slash		0	**7.27** (1.74)	**4.90** (1.02)	0	**.86** (.30)	**1.10** (.29)
Bouteloua curtipendila	C4						
Control		0	0	0	0	0	0
Seed		0	**.27** (.21)	**.53** (.31)	0	**.12** (.08)	**.20** (.12)
Slash		0	0	0	0	0	0
Seed + Slash		0	0	**.93** (.27)	0	0	**.52** (.21)
Artistida purpurea	C4						
Control		**.13** (.13)	**.13** (.13)	**.13** (.13)	**.02** (.02)	**.05** (.05)	**.06** (.06)
Seed		**.20** (.20)	**.14** (.14)	**.07** (.07)	**.09** (.09)	**.03** (.03)	**.01** (.01)
Slash		**.07**	**.07** (.07)	**.07**	**.02**	**.02**	**.02**

		(.07)		(.07)	(.02)	(.02)	(.02)
Seed + Slash		0	0	0	0	0	0
Achnatherum hymenoides	C3	0	0	0	0	0	0

Table 5. Average density (#/ plants/m2) and cover of all seeded species for Goose Pond Site (12 month after seeding).

Species	Photosynthetic Pathway	Avg. Density (m^2) 2003	2004	2005	Avg. Cover (%) 2003	2004	2005
Slash		0	0	0	**.02** (.02)	**.02** (.02)	**.02** (.02)
Seed + Slash		0	0	**.13** (.09)	0	0	**.01** (.01)
Elymus elymoides	C3						
Achnatherum hymenoides	C4	0	0	0	0	0	0
Control		0	0	**.07** (.07)	0	0	**.01** (.01)
Seed		0	**1.67** (.68)	**3.2** (1.46)	0	**.07** (.03)	**.11** (.03)
Slash		0	0	**7.87** (1.25)	0	0	**.84** (.17)
Seed + Slash		0	**46.6** (6.72)	**24.5** (3.41)	0	**3.88** (.62)	**2.83** (.60)
Bouteloua curtipendila	C4						
Control		0	0	0	0	0	0
Seed		0	**.07** (.07)	**.07** (.07)	0	**.01**(.01)	**.02** (.02)
Slash		0	0	0	0	0	0
Seed + Slash		0	0	**2.0** (.76)	0	0	**.67** (.31)
Artistida purpurea	C3						
Control		0	0	0	0	0	0
Seed		0	**.27** (.21)	**.07** (.07)	0	**.01** (.01)	**.01** (.01)

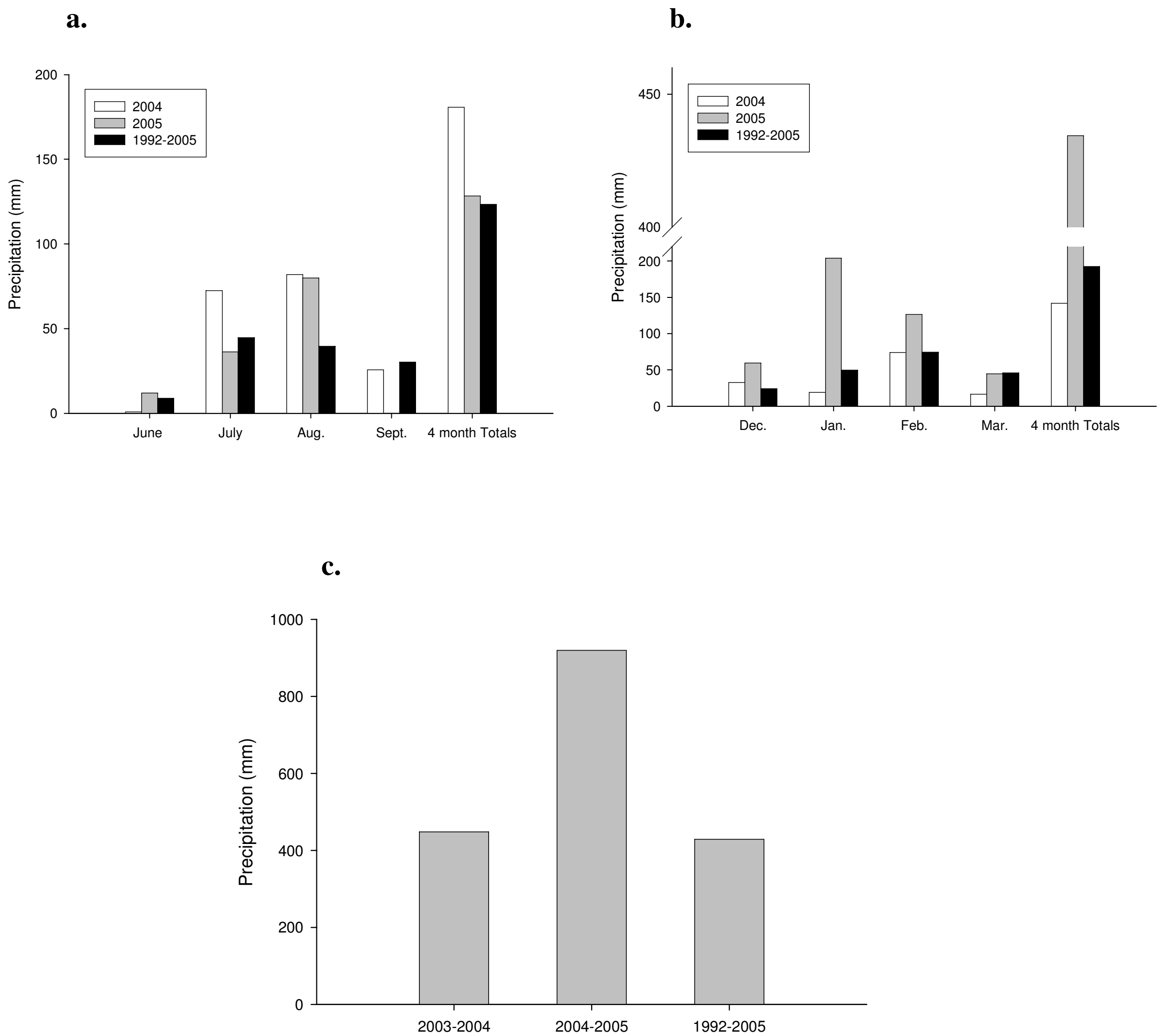

Figure 1. Monthly precipitation in 2004 and 2005 versus 13-year average. Summer (June-September) averages are shown in **a.** and winter (December-March) averages are shown in **b.** Annual precipitation (2003-2005, August –June) verse 13-year average (1992-2005) shown in **c**

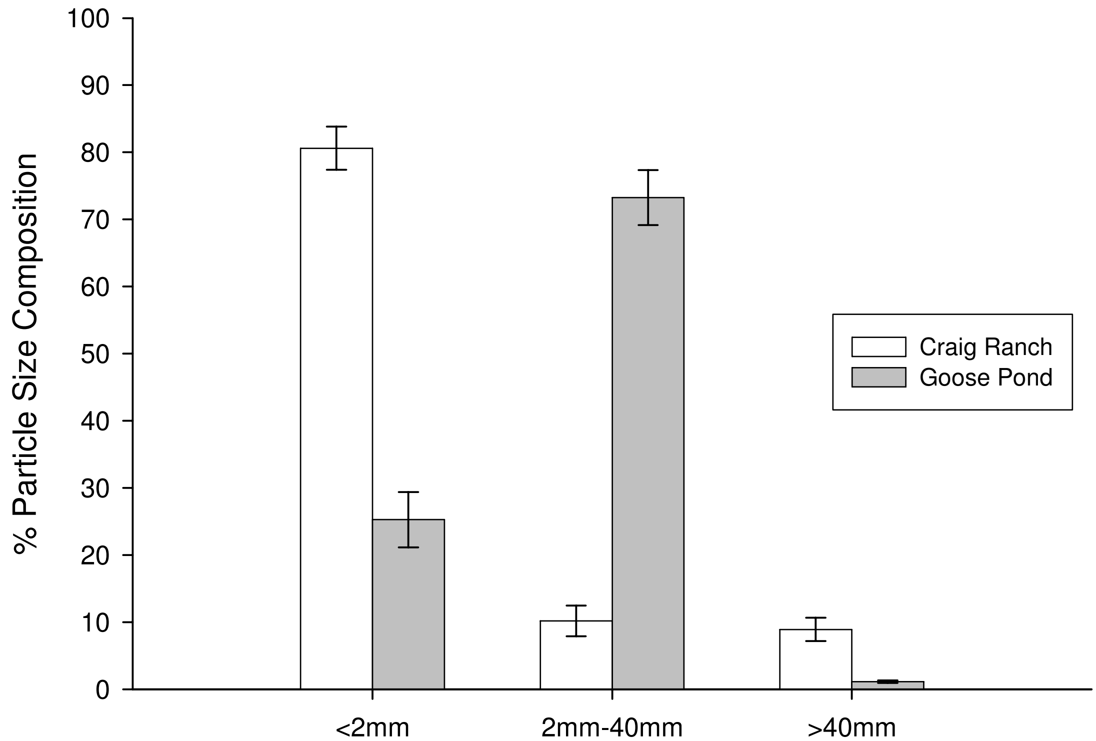

Figure 2. Average percent cover of soil particle size within the two sites. Bars represent 1 standard error (n = 60)

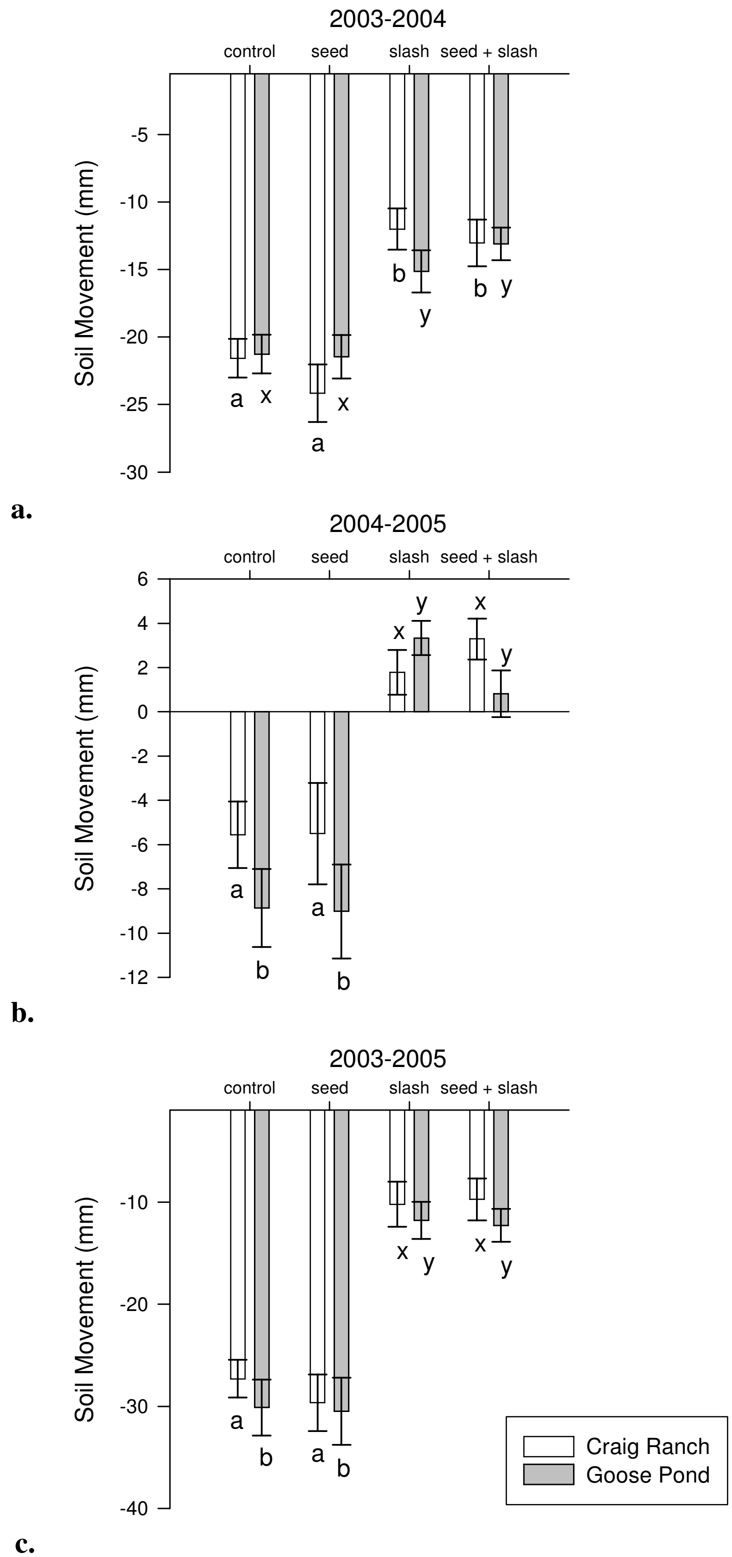

Figure 3. Average sediment loss form August 2003 to August 2005. Different letters indicate significant difference between treatments at $\alpha = 0.05$. Bars represent 1 standard error (n=15 for each site).

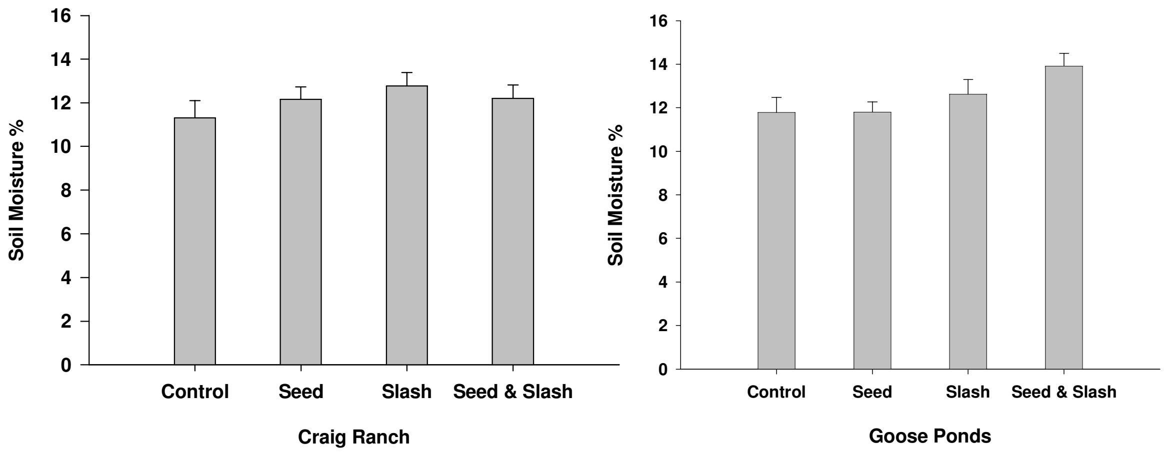

Figure 4. Average percent soil moisture within treatments for August 2004. No significant difference between treatments at $\alpha = 0.05$. Bars represent 1 standard error (n=15 for each site).

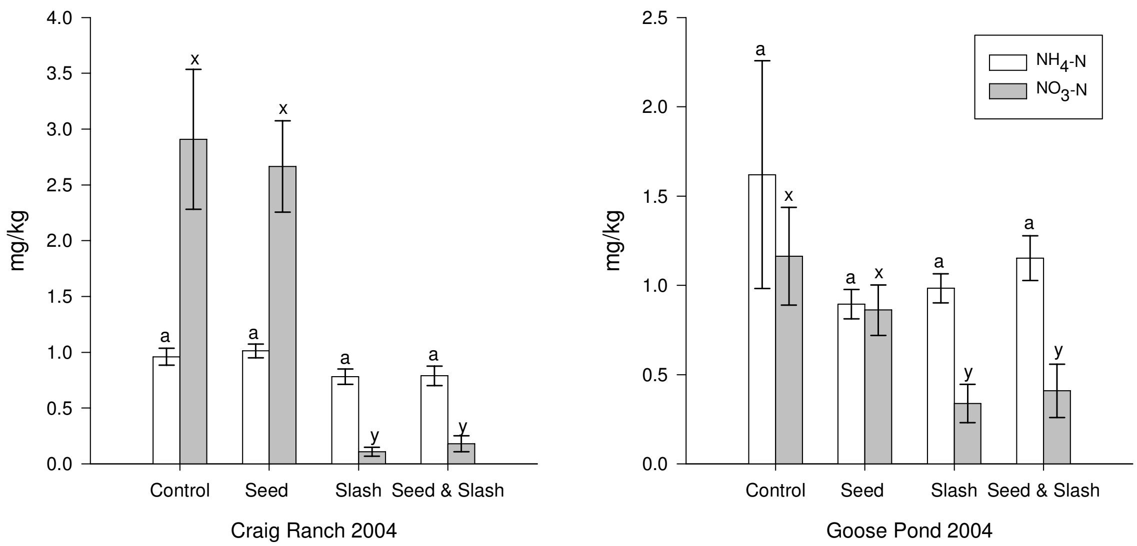

Figure 5. Average NH_4-N and NO_3-N mg/kg within treatments. Different letters indicate significant difference between treatment at $\alpha = 0.05$ per nutrient. Bars represent 1 standard error (n=15 per site)

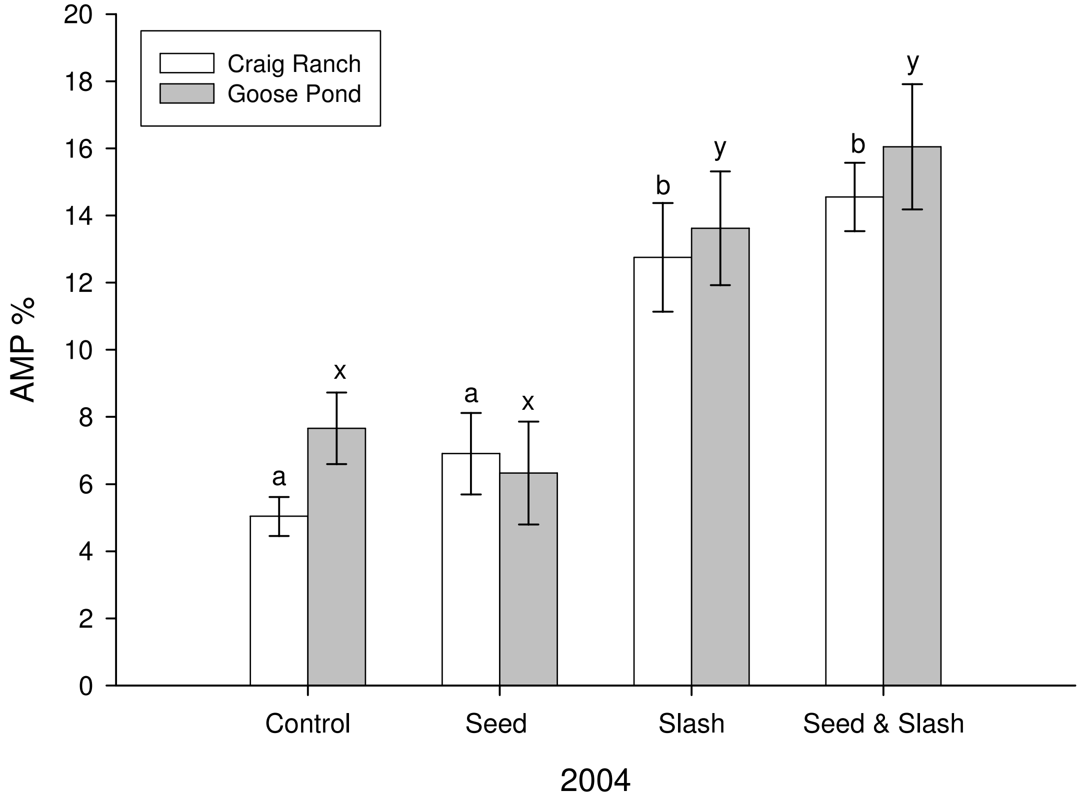

Figure 6. Relative root colonization with arbuscular mycorrhizal (AM) fungi within treatments. Different letters indicate significant difference between treatments at $\alpha = 0.05$. Bars represent 1 standard error (n=15 for each site).

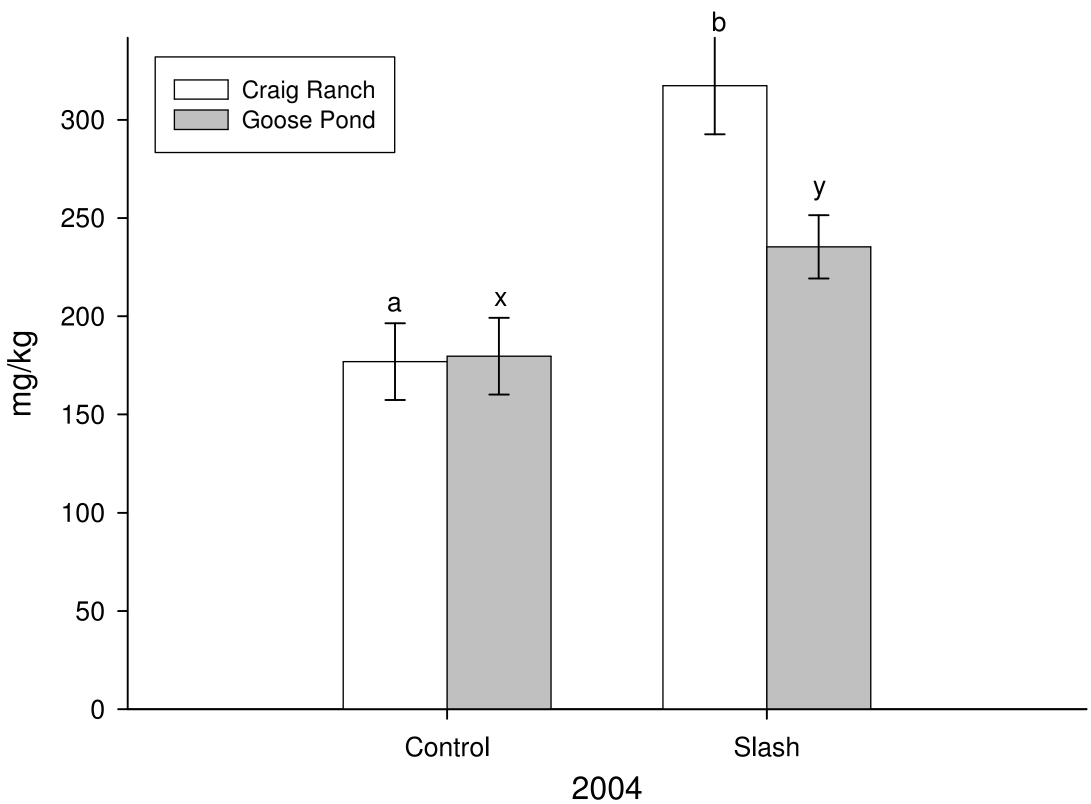

Figure 7. Average microbial carbon within treatments. Different letters indicate significant difference between treatments at $\alpha = 0.05$. Bars represent 1 standard error (n=15 for each site).

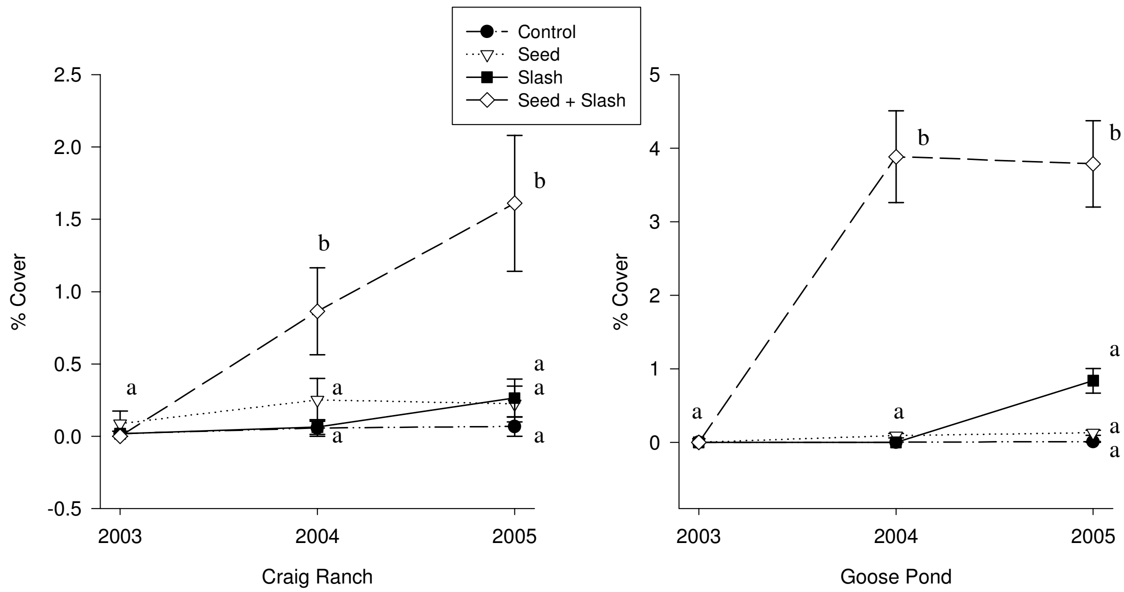

Figure 8. Average percent cover of seeded species within treatment. Different letters indicate significant difference between treatments at $\alpha = 0.05$. Bars represent 1 standard error (n=15 for each site).

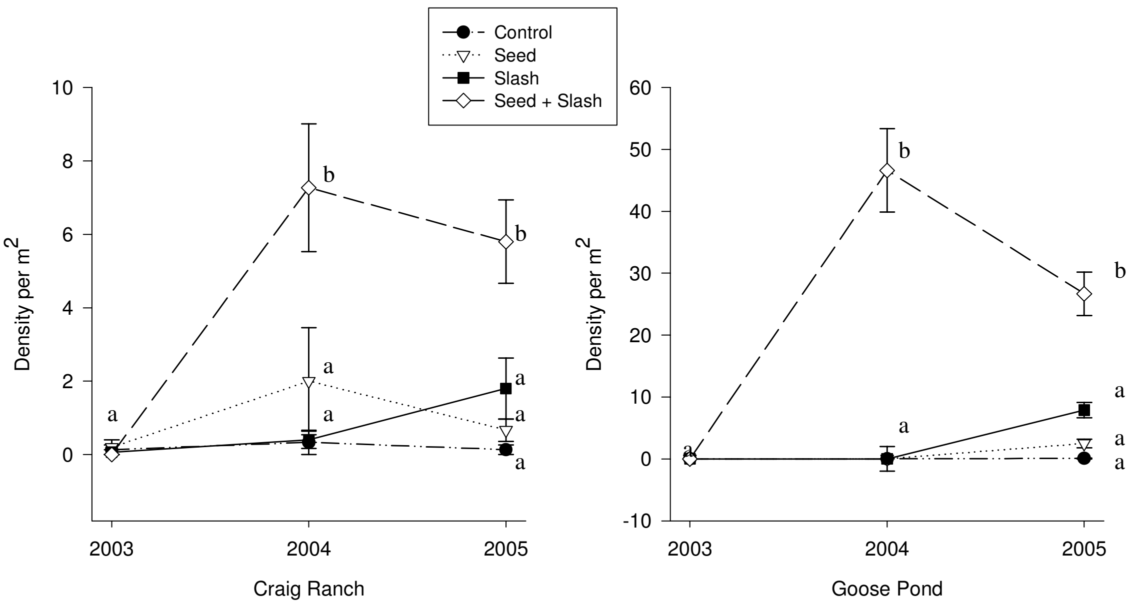

Figure 9. Average emergence per m^2 of seeded species within treatment. Different letters indicate significant difference between treatments at $\alpha = .05$. Error bars represent 1 standard error (n =15).

Appendix 1. List of species found within study plots. Nomenclature based on the USDA Plants Database (USDA, NRCS 2004)

Craig Ranch				Goose Ponds			
Functioinal groups		**Species**	**Common Name**	**Functioinal groups**		**Species**	**Com**
Annual		*Cordylanthus parviflorus*	purple bird's beak	Annual		*Chenopodium album*	lamb
		Polygonum douglasii	Douglas' knotweed			*Epilobium brachycarpum*	tall a willowh
		Portulaca oleracea	little hogweed			*Eriogonum davidsonii*	Davi buckwh
Exotic		*Lactuca serriola*	prickly lettuce			*Nicotiana attenuata*	coyo
Graminoid	C4	*Aristida purpurea*	purple threeawn			*Polygonum douglasii*	Doug knotwe
	C4	*Bouteloua curtipendula*	blue grama			*Portulaca oleracea*	little
	C4	*Bouteloua gracilis*	sideoats grama	Exotic	C3	*Bromus tectorum*	cheat
	C3	*Elymus elymoides*	western bottle-brush grass			*Lactuca serriola*	prick
	C3	*Hesperostipa comata*	needle & thread	Graminoid	C4	*Aristida purpurea*	purp
N fixer		*Psoralidium tenuiflorum*	slimflower scurfpea		C4	*Bouteloua gracilis*	blue
Perennial		*Agoseris glauca*	pale agoseris		C3	*Elymus elymoides*	weste brush g
		Arabis fendleri	Fendler's rockcress	N fixer		*Lotus uthahensis*	Utah trefoil
		Chaenactis douglasii	Douglas' dustymaiden			*Lupinus kingii*	King
		Chamaesyce albomarginata	whitemargin sandmat			*Phaseolus angustissiums*	sliml
		Eriogonum	crispleaf			*Psoralidium*	slimf

	corymbosum	buckwheat
	Hymenopappus filifolius	fineleaf hymenopappus
	Penstemon linarioides	toadflax penstemon
	Sphaeralcea parvifolia	smallflower globemallow
Shrub	*Purshia mexicana*	Mexican cliffrose

	tenuiflorum	scurfpe
Perennial	*Chaenactis douglasii*	Doug dustym
	Chamaesyce albomarginata	white sandma
	Eriogonum corymbosum	crisp buckwh
	Eriogonum umbellatum	sulph wildbuc
	Hymenopappus filifolius	finel hymeno
	Hymenoxys richardsonii	pingu hymeno
	Ipomopsis aggregata	skyro
	Machaeranthera canescens	hoary
	Packera multilobata	lobel
	Penstemon linarioides	toadf pensten
	Penstemon virgatus	uprig beardto
	Sphaeralcea parvifolia	smal globem
	Townsendia incana	hoary
Shrub	*Artemisia tridentata*	big s

Appendix 2. Average percent cover of functional group species within treatment for Craig Ranch Site. Standard error in parenthesis (n = 15 within each site). Percent of total understory species cover for each functional group.

Year	Treatment	Total	Annual	Exotic	Graminoid	N fixer	Perennial	Shrub
2003	Control	**2.48** (0.91)	**1.11** (0.48)	0	**0.02** (0.02)	**0.85** (0.66)	**0.01** (0.01)	**0.49** (0.18)
			44.8%		0.8%	34.3%	0.4%	19.8%
	Seed	**1.55** (1.67)	**0.73** (0.34)	0	**0.09** (0.09)	**0.50** (0.16)	**0.14** (0.11)	**0.09** (0.05)
			47.1%		5.8%	32.3%	9.0%	5.8%
	Slash	**2.40** (0.92)	**1.11** (0.72)	0	**0.02** (0.02)	**1.00** (0.54)	0	**0.28** (0.14)
			45.8%		0.8%	41.7%		11.7%
	Seed + Slash	**1.57** (0.36)	**0.67** (0.26)	0	0	**0.68** (0.34)	**0.01** (0.01)	**0.21** (0.10)
			42.7%			43.3%	0.6%	13.4%
2004	Control	**4.49** (0.94)	**2.55** (0.96)	0	**0.07** (0.06)	**0.29** (0.67)	**0.43** (0.25)	**1.15** (0.25)
			56.8%		1.6%	6.5%	9.6%	25.6%
	Seed	**3.50** (1.07)	**2.00** (0.94)	0	**0.29** (0.15)	**0.04** (0.02)	**0.52** (0.40)	**0.65** (0.30)
			57.1%		8.3%	1.1%	14.9%	18.6%
	Slash	**1.66** (0.57)	**0.36** (0.21)	0	**0.10** (0.06)	**0.72** (0.36)	**0.13** (0.13)	**0.35** (0.16)
			21.7%		6.0%	43.4%	7.8%	21.1%
	Seed + Slash	**2.10** (0.37)	**0.11** (0.04)	0	**0.87** (0.30)	**0.81** (0.28)	**0.19** (0.17)	**0.12** (0.07)
			5.2%		41.4%	38.6%	9.0%	5.7%
2005	Control	**16.8** (2.79)	**4.82** (1.14)	0	**0.10** (0.07)	**9.85** (2.31)	**0.41** (0.27)	**1.62** (0.49)
			28.7%		0.6%	58.6%	2.4%	9.6%
	Seed	**10.96** (2.95)	**3.13** (1.31)	0	**0.24** (0.12)	**6.27** (2.05)	**0.71** (0.54)	**0.61** (0.24)
			28.6%		2.2%	57.2%	6.5%	5.6%
	Slash	**11.64** (2.41)	**4.14** (1.13)	**0.01** (0.01)	**0.26** (0.13)	**6.32** (2.24)	**0.36** (0.25)	**0.55** (0.24)
			35.6%	0.1%	2.2%	54.3%	3.1%	4.7%
	Seed + Slash	**16.34** (2.15)	**6.24** (1.43)	0	**1.63** (0.48)	**8.00** (1.69)	**0.13** (0.09)	**0.34** (0.14)
			38.2%		10.0%	49.0%	0.8%	2.1%

Appendix 3. Average percent cover of functional group species within treatment for Craig Ranch Site. Standard error in parenthesis (n = 15 within each site). Percent of total understory species cover for each functional group.

Year	Treatment	Total	Annual	Exotic	Graminoid	N fixer	Perennial	Shrub
2003	Control	**1.28** (0.49)	0	0	0	0	**0.53** (0.46)	**0.75** (0.28)
							41.4%	58.6%
	Seed	**0.35** (.017)	0	0	0	**0.01** (0.01)	**0.05** (0.05)	**0.29** (0.17)
						2.9%	14.3%	82.9%
	Slash	**0.27** (.011)	0	0	0	**0.04** (0.03)	**0.11** (0.08)	**0.12** (0.05)
						14.8%	40.7%	44.4%
	Seed + Slash	**0.36** (.012)	0	0	**0.01** (0.01)	0	**0.17** (0.08)	**0.18** (0.06)
					2.8%		47.2%	50.0%
2004	Control	**3.51** (1.33)	**0.12** (0.08)	0	0	**0.68** (0.67)	**1.89** (1.03)	**0.82** (0.24)
			3.4%			19.4%	53.8%	23.4%
	Seed	**1.46** (.071)	**0.12** (0.07)	**0.15** (0.13)	**0.09** (0.03)	**0.37** (0.28)	**0.39** (0.25)	**0.34** (0.13)
			8.2%	10.3%	6.2%	25.3%	26.7%	23.3%
	Slash	**2.55** (1.25)	**0.01** (0.01)	0	0	**1.26** (1.20)	**0.98** (0.58)	**0.30** (0.14)
			0.4%			49.4%	38.4%	11.8%
	Seed + Slash	**6.84** (1.30)	**0.02** (0.02)	**0.13** (0.13)	**3.92** (0.61)	**1.12** (0.93)	**1.03** (0.52)	**0.62** (.27)
			0.3%	1.9%	57.3%	16.4%	15.1%	9.1%
2005	Control	**8.02** (3.10)	**0.89** (0.87)	0	**0.01** (0.01)	**1.48** (1.33)	**4.89** (2.09)	**0.75** (0.28)
			11.1%		0.1%	18.5%	61.0%	9.4%
	Seed	**3.59** (1.04)	**0.15** (0.10)	0	**0.13** (0.03)	**1.43** (0.75)	**1.34** (0.55)	**0.54** (0.25)
			4.2%		3.6%	39.8%	37.3%	15.0%
	Slash	**10.67** (3.02)	**0.41** (0.22)	**0.25** (0.23)	**0.84** (0.17)	**6.93** (2.64)	**1.98** (0.89)	**0.26** (0.11)
			3.8%	2.3%	7.9%	64.9%	18.6%	2.4%
	Seed + Slash	**12.34** (3.10)	**0.13** (0.06)	**0.04** (0.03)	**3.74** (0.60)	**5.43** (2.91)	**2.19** (0.82)	**0.81** (0.36)
			1.1%	0.3%	30.3%	44.0%	17.7%	6.6%

Appendix 4. Pictures of before treatment (August 2003) and after treatment (August 2004).

Appendix 5. Picture of before treatment (August 203) and post treatment (August 2005)

4. Cheatgrass encroachment on a ponderosa pine ecological restoration project in northern Arizona, USA

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Abstract

Land managers frequently thin small-diameter trees and apply prescribed fire to reduce fuel loads and restore ecosystem structure, function, and process in forested areas. There is increasing concern that disturbances associated with these management practices can facilitate nonnative plant invasions. *Bromus tectorum* is an annual grass from the Mediterranean region. It has invaded large areas of the Interior West and has become the dominant species in many of these areas. In 2003, a ponderosa pine ecological restoration site on Mt. Trumbull in the Uinkaret Mountains of northern Arizona experienced a large increase in *Bromus*. Thinning and burning projects had been conducted on this site since 1996. *Bromus* frequency increased on the thinned and burned plots by six-fold between 1996 and 2003. While *Bromus* also increased on thinned plots that were not burned and the untreated control plots, the frequency of *Bromus* was significantly lower than on the thinned and burned plots. There were two additional factors that may have influenced the *Bromus* invasion. In 2002, the region experienced the most extreme drought recorded in the past 100 years. Substantial rainfall returned to the area in September 2002, coincident with the timing of *Bromus* germination. Additionally, cattle were reintroduced to the study area in July 2002 after a five year hiatus in grazing. We present data that suggest the interaction of prescribed fire, small-diameter tree thinning, cattle grazing, and drought were the primary causes of the spread of *Bromus*.

Introduction

There is increasing concern among ecologists, land managers, and other stakeholders over the risk of aggressive, nonnative plant species spreading into forested areas treated for ecological restoration (Moore et al. 1999, Allen et al. 2002, Keeley et al. 2003). Ecological restoration applications such as thinning small diameter trees and prescribed fire are intended to reinvigorate all aspects of forest health, particularly increasing understory vegetation cover and reducing the risk of severe wildfires (Covington et al. 1997). However, disturbances generated by tree removal, prescribed fire, and associated human activities can also create opportunities for nonnative plant invasions (D'Antonio and Vitousek 1992, D'Antonio and Meyerson 2002, Korb et al. 2004). Increases in nonnative plant abundance have been documented in ponderosa pine forests that were thinned of small diameter trees and treated with prescribed fire (Griffis et al. 2001, Korb et al. 2004). In spite of the risk of invasion, there is evidence that forests treated with prescribed fire are less susceptible to nonnative plant encroachment than areas burned in wildfires (Crawford et al. 2001, Griffis et al. 2001). However, not all wildfires in the Southwest have facilitated nonnative plant invasions (Laughlin et al. 2004).

A nonnative plant species of particular concern in the American Interior West is cheatgrass (*Bromus tectorum* L., hereafter referred to as *Bromus*). *Bromus* was introduced to the United States in the late 1800's and has since spread throughout much of the Great Basin Desert and the surrounding mountains and grasslands (Mack 1981, Knapp 1996). In areas it has invaded, *Bromus* has reduced plant biodiversity (Mack 1981), altered soil characteristics (Evans et al. 2001, Norton et al. 2004), and substantially changed the local fire regime (Whisenant 1990). While disturbance due to grazing, development or other anthropogenic causes is usually credited with driving *Bromus* invasion (Mack 1981, Knapp 1996), relatively undisturbed sites have also been invaded (Belnap and Phillips 2001, Evans et al. 2001). Once it is established in an area, *Bromus* populations are often stable and persistent, even if there are no further disturbances (Knapp 1992, Brandt and Rickard 1994).

In 1995, the Bureau of Land Management (BLM), in conjunction with Northern Arizona University and the Arizona Game and Fish Department, initiated a large-scale ecological restoration project in the Uinkaret Mountains in northern Arizona (Moore et al. 2003). In 2003 we observed a striking shift in the herbaceous plant community within portions of the treated areas from a native perennial-dominated system to a *Bromus*-dominated system. There were two important events that preceded this invasion: 1) in 2002, this region received less than half of its average precipitation (Fig. 1), and 2) cattle were reintroduced to the restoration site after five years of exclusion from grazing. We propose that either or both of these factors, in combination with ongoing tree thinning and prescribed burning, were the catalysts for the *Bromus* invasion. In this article we will present data to document the severity of the *Bromus* invasion in the ponderosa pine forests of the Uinkaret Mountains and give insight into the causes of this invasion. This study was originally designed to monitor the long-term changes in the vegetative community, not the mechanisms behind single-species dynamics. For this reason we refrain from approaching these data in a traditional hypothesis-testing format.

Methods

Study Site

This study was conducted in the Uinkaret Mountains in northwestern Arizona, in a basin between the Mt. Trumbull and Mt. Logan Wilderness areas (hereafter Mt. Trumbull) located at latitude 36° 22' N and longitude 113° 8' W. The Uinkaret Mountains are sky islands of ponderosa pine forest and pinyon-juniper woodland. The mountains are surrounded by cool desert scrub vegetation (Welsh et al. 1993) to all directions except the south which is bounded by the Grand Canyon. Mt. Trumbull is part of the Grand Canyon/Parashant National Monument which is managed jointly by the BLM and the National Park Service. The study area is currently under BLM management. The elevation of the study site ranges from 2,000 m to 2,250 m (6560 ft. to 7380 ft.). The area averages approximately 35 cm (13.8 in.) of precipitation per year, but there is considerable annual variation (Fig. 1). Frontal storms generate snow and rain in the winter followed by a dry spring and early summer with monsoonal rains bringing ephemeral thunderstorms in the mid-late summer and early fall.

The study area is dominated by ponderosa pine (*Pinus ponderosa* P. & C. Lawson), though Gambel oak (*Quercus gambelii* Nutt.) and New Mexico Locust (*Robinia neomexicana* Gray) are also major components. Dominant shrubs include big sagebrush (*Artemisia tridentata* Nutt.), wax currant (*Ribes cereum* Dougl.), mountain snowberry (*Symphoricarpos oreophilus* Gray), and Utah serviceberry (*Amelanchier utahensis* Koehne). The principal perennial grasses are muttongrass (*Poa fendleriana* (Steud.) Vasey), squirreltail (*Elymus elymoides* (Raf.) Swezey), and western wheatgrass (*Pascopyrum smithii* (Rydb.) A. Löve). There is a highly diverse community of annual and perennial forbs. There are several nonnative species found on Mt. Trumbull (Table 1).

Plot Design

Beginning in 1995, 269 plots were established in a 1500 ha (3700 acre) area on Mt. Trumbull (Fig. 2). All plots were laid out in a systematic grid pattern at 300 m (984 ft.) intervals. Exceptions to the plot spacing occurred if the plot landed on a road or other anthropogenic structure or if there was less than 10% tree canopy cover on the plot. In these instances, the plots were offset by 50 m (164 ft.) to a more suitable location.

The plot design was modified from the National Park Service Fire Monitoring protocol (Reeberg 1995). Each plot was 20 x 50 m (0.1 ha/66 x 164 ft./0.25 acres) in size and was oriented with the long side running parallel with the slope of the terrain. The corners and centers of the plot were marked with rebar.

A point line-intercept method was used to collect cover data for understory plants. Measurements were taken every 30 cm (1 ft.) along two 50 m (164 ft.) line transects laid out on the long sides of the plot for a total of 166 points per line and 332 points per plot. If any part of a living plant intersected the point, the plant was identified and recorded as a hit. All plants were included except trees greater than breast height (137 cm/4.5 ft.).

Overstory canopy cover was determined using a verticle densitometer. Measurements were taken every 3 m (39 in.) along each 50 m line transect for a total of 16 points per transect or 32 points per plot. Canopy cover was recorded as either present or absent and a percentage was calculated per plot.

Restoration Treatments

Trees were thinned to restore pre-1870 stand density and structure (Covington et al. 1997, Moore et al. 1999, Waltz et al. 2003). All living trees that germinated prior to 1870 were retained. Additionally, 1.5-3 replacement trees were retained for every piece of remnant pre-1870 evidence (stumps, snags, etc.). Details of the criteria for replacing remnants are described by Waltz et al. (2003).

To protect old growth trees from heat-induced cambial girdling (Sackett et al. 1996), forest floor fuels were raked away from the boles to approximately 30 cm (1 ft.). Merchantable timber (>12.4 cm/5 in. dbh) was removed prior to burning. Slash and smaller logs were left on site. Treatment units were burned using drip torches for ignition. After burning, the treatment units were seeded with a mix of native plant seeds (Moore et al. 2003, Springer and Laughlin 2004)

Thinning and burning operations began on the study site in 1996 and continued into spring 2003 (Table 2). One treatment unit, High Meadow, had been thinned but not burned by 2003. A single plot from Cinder was also thinned but not burned. For analyses, that Cinder plot was included with the High Meadow unit. An additional 160 plots not included in this study are either scheduled for treatment at a future date or located in the Mt. Logan Wilderness. For logistical reasons we elected not to remeasure these plots in 2003. The remaining four plots were located on an exposed, uneroded basalt flow (Lava Flow and EB-1 Units; Fig. 2). Due to the anomalous nature of the parent material on these plots, they were excluded from the study.

Pretreatment measurements were taken primarily in the summer of 1996; although two plots were measured in October 1995 and 14 plots were measured for the first time in the summer of 1997 (hereafter all pretreatment measurements will be combined into the "1996" measurements). Since the plots were treated in different years, posttreatment measurements were taken at several times through the course of this study (Table 2). All treated and control plots were remeasured in the summer of 2003. To minimize seasonal differences in the vegetation, the 2003 measurements were timed to coincide with the original pretreatment measurements.

Plant Identification

Plant nomenclature and nativity followed the USDA Plants Database (USDA, NRC 2004). Where possible, all plants were identified to the specific level. When field identification was not reliable or hybridization was suspected, the plants were identified to the generic level. Notable examples of this are the broad-leaved *Chenopodium* spp., all *Lotus* spp., and all *Solidago* spp. In each of these instances, all taxa were only identified to the genus.

It is highly likely that the broad-leaved members of the *Chenopodium* genus include the nonnative species *Chenopodium album* L. var. *album*. There are, however, three other species of broad-leaved *Chenopodium* present in this region which are difficult-to-impossible to reliably distinguish in the field. These species are *C. fremontii* S. Wats., *C. berlandieri* Moq., and *C. incanum* (S. Wats.) Heller which are all native to the United States. We have chosen a conservative identification and consider all of the *Chenopodium* spp. as native. It should be recognized that *C. album* var. *album* may be present on our plots and therefore the numbers we report for nonnative species are potentially low.

Cattle Grazing

Mt. Trumbull has been grazed by domestic livestock at various intensities since Euro-American settlement in the late 1800's. Cattle grazing was excluded from the study area upon initiation of the restoration treatments. In July 2002, 64 head of cattle were reintroduced and grazed through October of the same year. The following year, 76 head of cattle were grazed during the same months as in 2002.

Data Analysis

Our study incorporates a Before-After/Control-Impact (BACI) design (Stewart-Oaten and Murdoch 1986, Green 1993, Underwood 1994). While this design is not consistent with true replication and randomization (Hurlbert 1984), it allowed us to examine ecosystem response to restoration treatments across a large landscape (Van Mantgem et al. 2001). The treatment area was organized into separate land units based on topography and management objectives (Fig. 2). Restoration treatments were implemented based on the geographic boundaries of these management units. For the purposes of this study, each study plot was analyzed as an independent sample point. We have not attempted to account for spatial autocorrelation in our analyses.

Percent frequency was calculated by dividing the number of plants recorded on both line transects by 332 (the number of possible hits per each plot). The data were not consistently normally distributed nor could they be made normal by transformation. To test for differences between treatments and years we used the nonparametric Kruskal-Wallis test. Tests between individual years within treatment units were conducted using the Wilcoxon Signed Ranks Test. We tested for correlations between changes in canopy cover and changes in understory cover using a simple linear regression. Significances were based on $\alpha=0.05$. All analyses were conducted using SPSS software (SPSS 2003).

Climate Data

Precipitation data are based on information recorded at the Nixon Flats Remote Automated Weather Station (RAWS) site at Mt. Trumbull, AZ (1980 m/6500 ft.). The data at this site cover 12 years (1992-2003). If data were not available from the Nixon Flats site, they were supplemented by data from the Mt. Logan RAWS site which is located near the top of Mt. Logan, AZ (2190 m/7185 ft.).

Results

Change in Understory Vegetation

By 2003, *Bromus* frequency had increased by over sixfold on the thinned and burned plots (Table 3). The frequency of *Bromus* in 2003 was greater on the treated plots than on the controls (Kruskal-Wallis test; $\chi^2=36.098$; df=1; $p<0.001$). Significant increases in *Bromus* were also seen in thin only and control plots, but the average frequency remained low. When we examined the different treatment units, we detected significant increases in *Bromus* in the Trick Tank, Rye Flat, and High Meadow units (Fig. 3). There was a significant year interaction in the Lava unit for *Bromus* (Kruskal-

Wallis test; χ^2=16.393; df=7; p<0.022) and native species (Kruskal-Wallis test; χ^2=18.307; df=7; p<0.011). We could not, however, detect a difference between individual years for either *Bromus* or natives (Fig. 3). This may be a function of the low sample size (n=3). In the treatment units that we sampled in more than one posttreatment year, the increase in *Bromus* was not detected until 2003 (Fig. 3). Changes in *Bromus* cover were positively correlated with the number of growing seasons between the year the plot was burned and 2003 (r = 0.42; p = 0.02). No correlation was detected between changes in *Bromus* cover and reduction in tree canopy cover (r=0.09; p=0.52).

Native understory vegetation increased on the thinned and burned plots and significantly decreased in the controls between 1996 and 2003 (Table 4). No change was detected on the thin only plots. By 2003, native species frequency was significantly greater than that of *Bromus* regardless of treatment (Table 4). On the thinned and burned plots, however, *Bromus* was by far the most prevalent single species. The most frequent native species were *Elymus elymoides*, *Lupinus argenteus* Pursh, and *Robinia neomexicana* with average frequencies per plot of 3.5%, 2.7%, and 2.7% respectively. In the thinned and burned units, *Bromus* occurred on 19.6% of all points per plot in 2003 (Table 4).

Changes in Precipitation

The year 2002 had the lowest precipitation recorded in the 12 years of data from the Nixon Flats RAWS site (Fig. 1). Longer term records in Arizona show 2002 to be the most severe drought in over a century (CLIMAS 2004). Additionally, five of the eight years since initiation of the ecological restoration project have had sub-average moisture (Fig. 1).

Discussion

Cheatgrass Response to Restoration

The *Bromus* invasion on the Mt. Trumbull restoration project was linked to the reintroduction of fire to this area. Plots that were thinned and burned had significantly higher frequency of *Bromus* than plots that were only thinned or the control plots (Table 4). While thinning alone generated a significant increase in *Bromus*, frequencies were still low in 2003 (3.0% = 10 hits in 332 points) and the change may just be an artifact of sampling. The lack of correlation between changes in canopy cover and changes in *Bromus* frequency, suggest that fire was important in establishing the proper ecological conditions for *Bromus* to invade. However, in the absence of data from plots that were burned but not thinned, we cannot dismiss thinning disturbance as a factor in facilitating the invasion by *Bromus*.

Although there are reported cases of *Bromus* invading burned ponderosa pine forests in the Southwest, the results are inconsistent. In a study of invasions of areas in northern Arizona burned in wildfires, Crawford et al. (2001) reported an increase in *Bromus* cover from <0.5% in unburned sites to 3% in moderate burns and 19% in high-severity burns. Another study examining the response of the understory vegetative community in isolated areas on the North Rim of Grand Canyon National Park, Arizona (hereafter North Rim) to a wildfire in 1999, *Bromus* was found to be less prevalent within the fire's perimeter than neighboring unburned areas (Laughlin et al. 2004). Crawford and Straka (2004) noted increases in *Bromus* distribution and cover over four years of post-fire study in burned areas of the Outlet Fire near Walhalla Plateau, North Rim.

While some studies have shown an increase in nonnative species in Southwest ponderosa pine forests treated with prescribed fire (Sackett et al. 1996, Abella and Covington 2004, Korb et al. 2004), *Bromus* is rarely cited as a major component of the nonnative vegetative community. In a study in northern Arizona, Griffis et al. (2001) did not detect a significant increase in exotic graminoids after fire, regardless of whether the fire was prescribed or wild or in conjunction with thinning projects. On the North Rim, *Bromus* was found on 40% of plots after a prescribed burn, but its average relative abundance was <1% of all plants recorded (K. Huisinga 2004, Northern Arizona University, Flagstaff, AZ *Personal Communication*).

Although fire disturbance created the appropriate conditions for *Bromus* to invade at Mt. Trumbull, we suggest that it was not the only mechanism that triggered the invasion. The *Bromus* invasion was not synchronous with the burning of the plots, but occurred in a single year between late 2002 and summer 2003 (Fig. 3; *Personal Observation*). We therefore suggest that the *Bromus* invasion was driven by the interaction of fire and thinning disturbances with other factors unique to this time frame. The reintroduction of cattle grazing to the study site and the severe drought of 2002 both coincided with the initiation of the *Bromus* invasion.

The Influence of Cattle Grazing on *Bromus*

Cattle grazing has been associated with the spread of *Bromus* in the western United States (Mack 1981, Sparks et al. 1990, Knapp 1996). Few studies, however, have directly addressed this in a controlled fashion and the results from these studies have been mixed and inconclusive. In northern Wyoming, *Bromus* was less prevalent in plots excluded from grazing than in grazed areas (Stohlgren et al. 1999). Conversely, Pierson and Mack (1990) reported lower *Bromus* recruitment when the plot was protected from grazing. Several studies examining the influence of cattle grazing on nonnative grasses have reported that grazing alone is not sufficient to explain the invasion (Anable et al. 1992, Stohlgren et al. 1999, Harrison et al. 2003).

Our study did not control for cattle grazing, so any conclusions we draw are speculative. As with the previously mentioned studies, it is unlikely that cattle grazing alone can explain the level of invasion on Mt. Trumbull by *Bromus*. Cattle grazed the entire study area in late summer 2002. Sites that had relatively low *Bromus* frequency in 2003 were subjected to the same grazing regime as sites with higher levels of invasion. This is particularly evident when comparing the High Meadow and Cinder units to the other treated sites (Fig. 3). In 2003, High Meadow had the lowest low level of *Bromus* of any treated unit, but was still grazed at the same rates as the neighboring Trick Tank unit, which had the highest level of *Bromus*.

We are not suggesting that cattle grazing played no role in the spread of *Bromus*. By grazing on the drought-stressed native vegetation, cattle may have further reduced native grass vigor and growth. This may have generated open resource niches which *Bromus* seedlings could utilize (Tilman 1997). Additionally, cattle may have been a vector for transport of *Bromus* seed from the lower-elevation pastures where they graze in the early summer.

The Influence of Drought on *Bromus*

The drought of 2002 was not only unique in its severity; the timing of the sparse precipitation was particularly detrimental to the native perennial vegetation and facultative to the success of *Bromus*. From August 2001 through August 2002, Mt. Trumbull received only 29% of the average precipitation for the area (Fig 4). In September 2002 and spring 2003, the area received above-average precipitation, which would be coincident with the timing of *Bromus* germination and growth. *Bromus* germinates in late summer, overwinters as a seedling, and then grows rapidly after snowmelt in early spring (Upadhyaya et al. 1986). It is very likely that the timing of the drought suppressed the native perennial growth in 2002, freeing up resources that *Bromus* was able to capture before the native perennials could recover from the effects of the drought.

A similar weather pattern facilitated a *Bromus* invasion in Canyonlands National Park, Utah (Belnap and Phillips 2001, Evans et al. 2001). Prior to fall 1994, *Bromus* was a minor component of the vegetative community. After a mild, wet winter, *Bromus* became the dominant vegetation in several areas of the park, including areas that were subject to minimal disturbances.

Management Implications and Conclusions

The recent expansion of *Bromus* in higher elevations of northern Arizona was not isolated to the Mt. Trumbull region. In a grazing study on Anderson Mesa, near Flagstaff, Arizona, *Bromus* cover increased nearly 100-fold between 2002 and 2003 (M. Loeser 2004, Northern Arizona University, Flagstaff, AZ, *Personal Communication*). Additionally, increases in *Bromus* have been reported at the North Rim (Crawford and Straka 2004) and Kaibab National Forest (*Personal Observation*).

Many researchers suggest that prevention and early intervention are the best mechanisms for preventing nonnative plant invasions (Hobbs and Humphries 1995, D'Antonio and Meyerson 2002). The project at Mt. Trumbull did include some measures for promoting the establishment of native plant species. The application of native seeds to the treated areas was the primary method of manipulating the understory vegetation. While posttreatment seeding has been shown to be effective in mitigating nonnative invasions (Bakker et al. 2004), hindsight demonstrates that this was insufficient to prevent *Bromus* from invading Mt. Trumbull. Unfortunately, no proactive measures were taken to eradicate or contain nonnatives prior to or immediately following treatment. The presence of nonnatives prior to the generation of restoration-associated disturbances suggested that these areas were at risk for invasion. If the prevention of invasion by nonnative plant species is a priority for practitioners, then they will need to contain and eliminate these species prior to imposing any new or elevated disturbance regimes on the landscape. This is complicated by the need for urgency in treating forested areas for fuel reduction and ecological restoration to prevent severe wildfires and loss of biodiversity (Covington 2000).

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Table 1. List of all nonnative species detected on the Mt. Trumbull study plots. Nativity and nomenclature based on the USDA Plants Database (USDA, NRCS 2004).

Latin Name	**Common Name**	**Growth Form**
Agropyron desertorum	Desert Wheatgrass	Perennial Grass
Bromus inermis	Smooth Brome	Perennial Grass
Bromus tectorum	Cheatgrass	Annual Grass
Chorispora tenella	Crossflower	Annual Forb
Circium vulgare	Bull Thistle	Biennial Forb
Convolvulus arvensis	Bindweed	Perennial Forb
Erodium cicutarium	Redstem Stork's Bill	Annual Forb
Lactuca serriola	Prickly Lettuce	Biennial Forb
Leonurus cardiaca	Common Motherwort	Perennial Forb
Malva neglecta	Cheeseweed	Perennial Forb
Marrubium vulgare	Horehound	Perennial Forb
Medicago sativa	Alfalfa	Annual Forb
Onopodrum acanthium	Scotch Thistle	Biennial Forb
Polygonum convolvulus	Black Bindweed	Annual Forb
Salsola tragus	Russian Thistle	Annual Forb
Sisymbium altissimum	Tumble Mustard	Annual Forb
Taraxacum officianale	Dandelion	Perennial Forb
Thinopyrum intermedium	Intermediate Wheatgrass	Perennial Grass
Tragapogon dubius	Yellow Salsify	Biennial Forb
Verbascum thapsus	Common Mullein	Biennial Forb

Table 2. Treatment schedule for ecological restoration project on Mt. Trumbull.
N/A = Not Applicable.

Treatment Unit	Area (Ha)	# Plots	Year Thinned	Year Burned	Years Measured (posttreatment)
Lava	18	3	1996	1996	1997-2003
Trick Tank	64	8	1998	1998	1999, 2001, 2003
EB 2 & 3	39	4	1999	2000	2003
Rye Flat	66	8	1999	2001	2001, 2003
Cinder	92	7	2000	2002	2003
High Meadow	84	10	2000/2003	N/A	2003
Control	538	65	N/A	N/A	2003

Table 3. Average percent frequency of *Bromus* versus all other nonnatives combined on treated and control plots on Mt. Trumbull. Different letters within rows indicate significant differences at α=0.05. All significances determined using Wilcoxon Signed Ranks test. Standard Mean Error in parenthesis. Control n=65; Thinned and burned n=30; Thin only n=10.

Treatment	1996		2003	
	Bromus	Other Nonnatives	*Bromus*	Other Nonnatives
Control	2.4 (0.9)[a]	0.1 (0.1)[b]	4.6 (1.6)[c]	0.5 (0.3)[b]
Thinned and burned	3.0 (1.1)[a]	0.2 (0.1)[b]	19.6 (3.5)[c]	1.9 (0.6)[d]
Thin only	0.01 (0.01)[a]	0.03 (0.02)[b]	3.0 (1.7)[c]	2.2 (2.0)[b]

Table 4. Average percent frequency of *Bromus* versus native understory species on treated and control plots on Mt. Trumbull. Different letters within rows indicate significant differences at α=0.05. All significances determined using Wilcoxon Signed Ranks test. Standard Mean Error in parenthesis. Control n=65; Thinned and burned n=30; Thin only n=10.

Treatment	1996		2003	
	Native Sp.	*Bromus*	Native Sp.	*Bromus*
Control	13.7 (2.1)[a]	2.4 (0.9)[b]	9.1 (1.5)[c]	4.6 (1.6)[d]
Thinned and Burned	12.6 (2.6)[a]	3.0 (1.1)[b]	21.8 (2.4)[c]	19.6 (3.5)[d]
Thin only	23.0 (5.1)[a]	0.01 (0.01)[b]	18.1 (3.4)[a]	3.0 (1.7)[c]

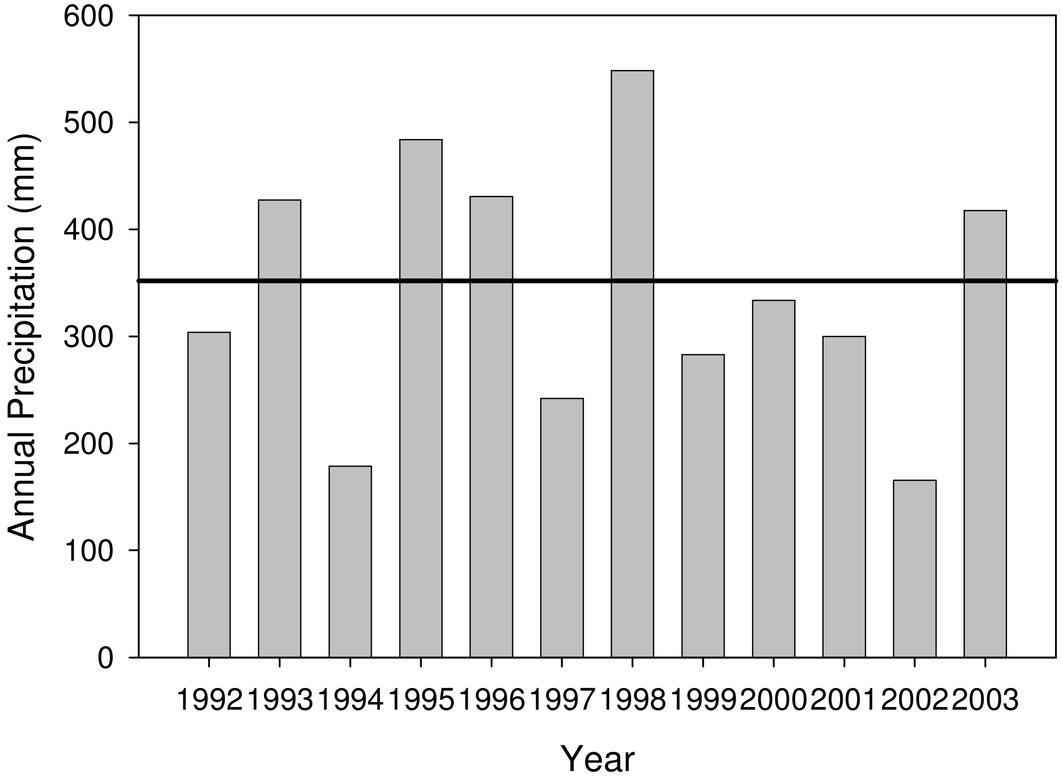

Figure 1. Total and average annual precipitation on Mt. Trumbull. Bars show total precipitation. Solid line shows average annual precipitation from 1992-2003. All data are from the Nixon Flats RAWS site.

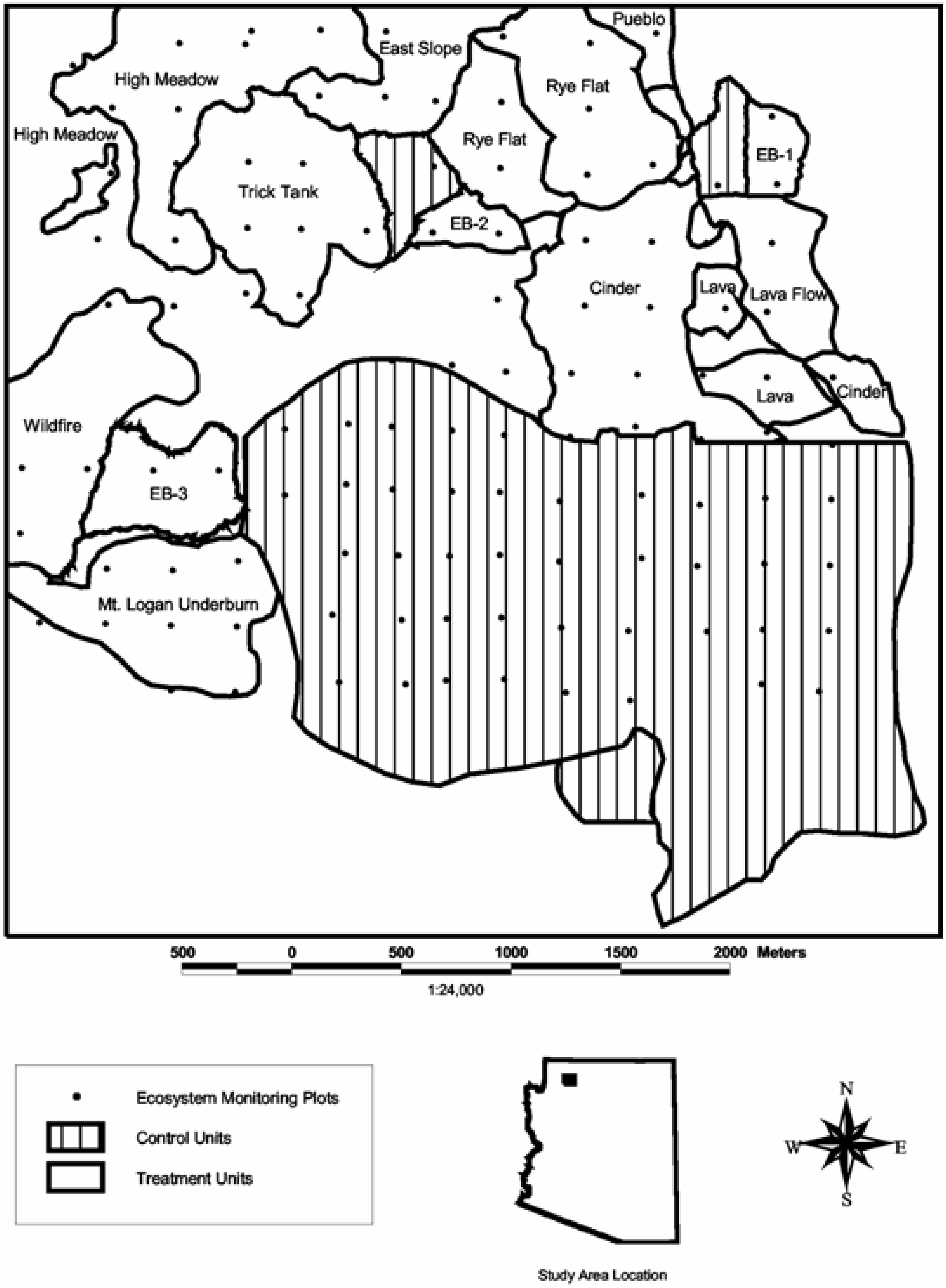

Figure 2. Map of Mt. Trumbull Research Site showing location of study plots and treatment units.

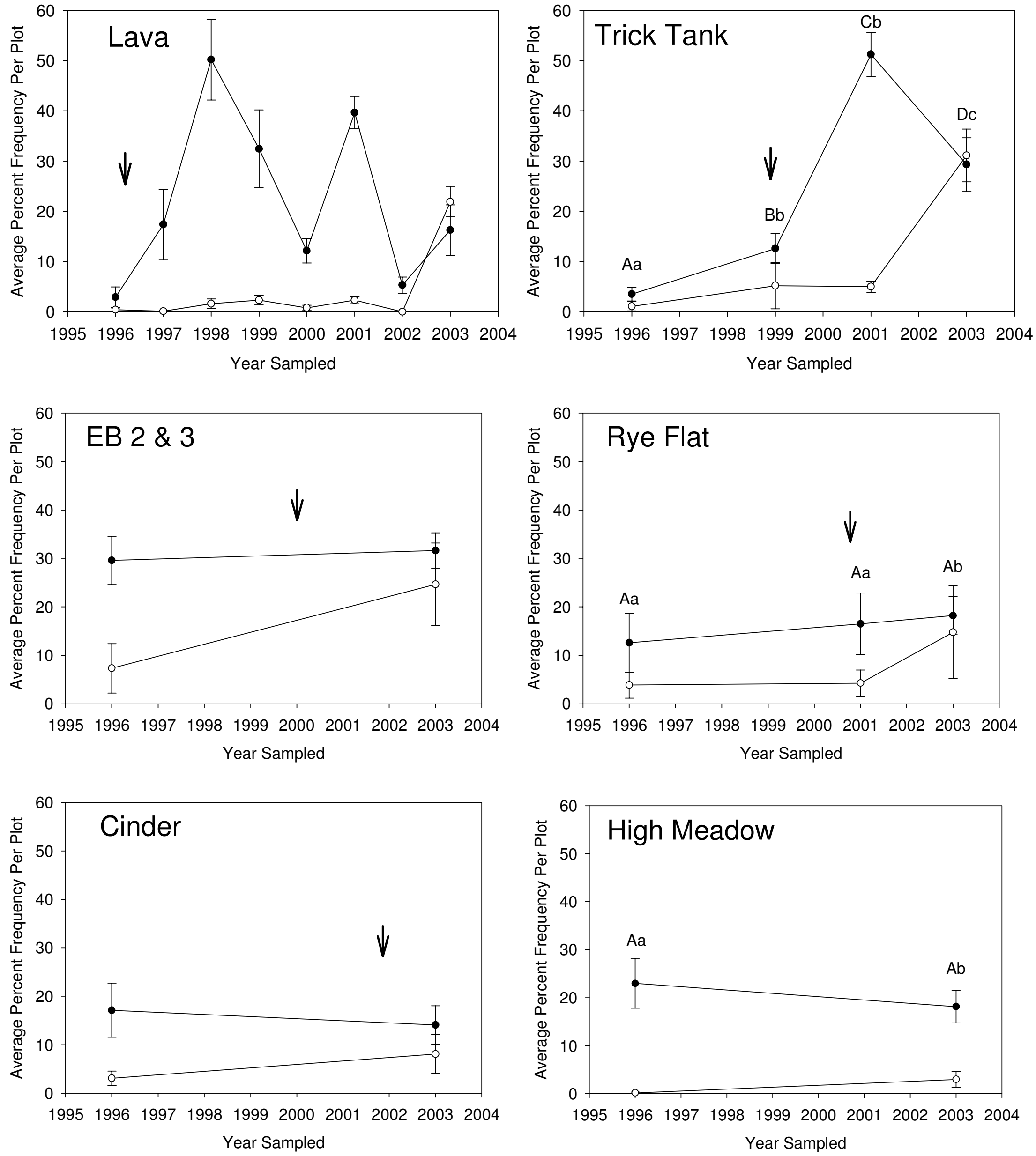

Figure 3. Average Percent Frequency of native species (black circles) and *Bromus* (open circles) on the different treatment units at Mt. Trumbull. Arrow indicates year of burn. Different letters indicate significant differences α=0.05. Capital letters used for native species, small case letters used for *Bromus*. If no letters present, then no differences were detected. All significances determined using Wilcoxon Signed Ranks test. Error bars = Standard Mean Error.

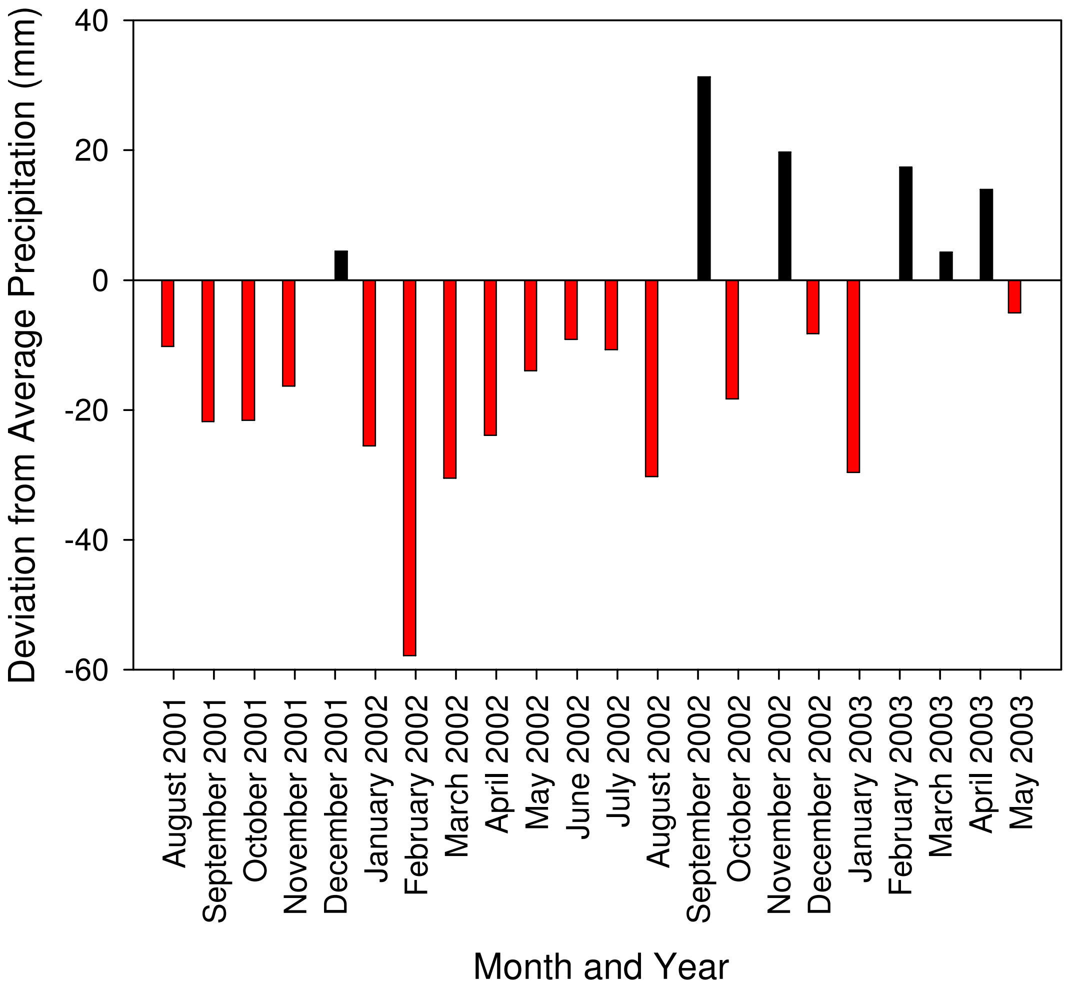

Figure 4. Deviation in monthly precipitation for August 2001-May 2003 as compared to the 12-year average on Mt. Trumbull as recorded at the Nixon Flats RAWS site from 1992-2003. Black bars show above-average precipitation; red bars indicate below-average precipitation.

5. Ecological factors influencing the persistence of remnant native grass patches in a recent cheatgrass invasion

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Abstract

Past land management practices have led to an increase in fire intensity and severity in Southwestern ponderosa pine forests. This increase has elevated the level of fire-generated disturbances in forests burned by wildfires. To restore the fire regime to a frequency and severity more consistent with historical levels, land managers often thin small diameter trees then burn the area. While an important goal of these management practices is to reinvigorate the understory vegetation, thinning and burning can also make the ecosystem more vulnerable to invasion by aggressive, nonnative plant species. In 2003, cheatgrass invaded thinned and burned areas of an ecological restoration project at Mt. Trumbull in northern Arizona. Where it has invaded, cheatgrass has become the dominant species in the herbaceous community. Within this matrix of cheatgrass, however, patches of remnant native vegetation have resisted invasion. In this study we will examine the ecological factors that influence the maintenance of these remnant patches. We hypothesize that the heterogeneous distribution of cheatgrass and the native remnant patches is determined by one or more of the following mechanisms: 1) interspecific plant competition, 2) propagule availability, and/or 3) small-scale variability in soil nutrient content. To test these hypotheses, we have established a research project at Mt. Trumbull that will experimentally manipulate plant competition and propagule pressure in both native- and cheatgrass-dominated areas. Additionally, we will describe the soil nutrient characteristics within the native and nonnative community types. The study was initiated in summer 2004 and will continue till summer 2007.

Keywords: Cheatgrass, Ecological Restoration, Plant Competition, Propagule pressure,

Introduction

The role of fire in Southwestern ponderosa pine ecosystems has changed dramatically in recent times. The natural fire regime has shifted from frequent, low-intensity ground fires to infrequent, high-intensity crown fires (Swetnam and Baisan 2003). This shift has elevated the level of fire-generated disturbance by completely consuming understory vegetation and litter layers, exposing bare mineral soil across large areas, and reducing crown cover (Zimmerman 2003). Consequently, land managers have implemented ecological restoration programs consisting of thinning trees to attain stand density more similar to pre-settlement conditions and prescribed burning that more closely emulate the lower-intensity fire regime under which ponderosa pine systems evolved (Moore et al. 1999).

While ecological restoration treatments are intended to reinvigorate all aspects of forest health, including the native understory vegetative community, there is a growing concern among land managers, scientists, and stakeholders about the risk of encroachment by aggressive, nonnative plant species (hereafter nonnatives) (Allen et al. 2002). There is evidence that areas burned in wildfires can be more susceptible to nonnative invasion than those areas burned by prescribed fire (Griffis et al. 2001). Conversely, in some instances prescribed burns have promoted nonnatives (Korb et al. 2004) and forests burned by wildfires are still dominated by native vegetation (Laughlin et al. 2004). Land managers will sometimes attempt to make a burned area more resistant to nonnative encroachment by reseeding the area with native seeds and/or amending the top soil to encourage native plant germination and success (Moore et al. 1999). These practices can be effective in facilitating the recovery of native understory vegetation in ponderosa pine forest restoration projects (Korb et al. 2004), as well as in combating invasion by cheatgrass (*Bromus tectorum* L., hereafter referred to as cheatgrass) (Belnap et al. 2003). Conversely, pinyon-juniper systems in northern Arizona observed greater levels of cheatgrass invasion on plots that were seeded and harrowed after burning than plots that were only seeded or received no post-fire remediation (Scoles et al. 2003).

The Bureau of Land Management (BLM) Arizona Strip Field Office is conducting ponderosa pine forest restoration projects in the Mount Trumbull Resource Conservation Area (Mt. Trumbull) in the Uinkaret Mountains of northern Arizona. An unintended response to these restoration treatments has been a rapid expansion of nonnative cheatgrass into the treated areas (Fig. 1). This invasion occurred between the summers of 2002 and 2003 (Fig. 2). There are two additional factors that might be relevant to the invasion of cheatgrass at Mt. Trumbull. First, the summer of 2002 was a severe drought year (Fig. 3). Second, cattle were reintroduced onto Mt. Trumbull that same summer for the first time since the initiation of the restoration treatments.

Throughout much of the treated landscape, cheatgrass has become the dominant understory species. Cheatgrass infestations of this magnitude and extent have not been documented in any other Southwestern ponderosa pine site, even following severe wildfires (Sieg et al. 2003). However, within the matrix of cheatgrass domination, there remain patches of native-dominated vegetation (Fig. 4). Data suggest that a narrow suite of native species are dominating these remnant patchs (Table 1).

Cheatgrass is a native of the Mediterranean regions of Europe and Africa. It is an annual grass that germinates in late summer and overwinters as a seedling. Flowering occurs in mid-summer (late June to early July). It was first introduced into western North America in the late 1800s probably as a contaminant of wheat seed (Mack 1981). Cheatgrass is now ubiquitous in the Great Basin Desert and in many regions has become the dominant species (Knapp, 1996). Where it is abundant, cheatgrass has been shown to reduce biodiversity at several trophic levels and accelerate the local fire cycle (Whisenant, 1990). Changes in soil nutrient content, particularly in available nitrogen, have also been cited as an important ramification of cheatgrass invasion. Results on this topic have been contradictory, with some studies showing decreases in available nitrogen (Evans et al. 2001) while others demonstrate an increase (Norton et al. 2004). Although cheatgrass is considered a serious problem throughout much of the Intermountain West of North America, Colorado is currently the only state that lists it as a Noxious Weed (USDA, NRCS 2004).

Plant invasions are often driven by disturbance (Hobbs and Huenneke, 1992). While these disturbances are usually associated with development (roads, urban areas, etc.) (Anable 1990, Gelbard and Belnap 2003), wildfire (Crawford et al. 2001), timber harvest (Upadhyaya et al. 1986), and food production (grazing, farming, etc.) (Mack 1981, Pierson and Mack 1990, Swope 2003), ecological restoration treatments can also promote nonnative plant invasion (Griffis et al. 2001, Korb et al. 2004). Prescribed burns, thinning of overgrown forests, and even eradicating non-native weeds from an area can create opportunities for invasive species to enter the ecosystem (D'Antonio and Meyerson 2002).

This study will examine the ecological factors influencing the preservation of the native-dominated patches within a recently-invaded, cheatgrass-dominated landscape. We hypothesize that these patches are preserved through one or more of the following mechanisms: 1) soil characteristics, 2) plant competition, and/or 3) propagule availability.

Soil properties have been associated with cheatgrass invasions in other ecosystems (Stohlgren et al. 1999, Belnap and Phillips 2001, Norton et al. 2004). Nutrient availability, particle size, and pH have been determined to be important in regulating cheatgrass distribution in some settings. A tight correlation between soil properties and the presence or absence of cheatgrass would suggest that either edaphic characteristics are regulating, at least in part, the distribution of native and nonnative plant species on Mt. Trumbull or vice versa.

A second proposed mechanism for determining whether or not cheatgrass will invade an area is competition from native plant species. It has been proposed in other habitats that well-established plant communities will be more resistant to nonnative invasion (Tilman 1997). Alternatively, research has suggested that specific native species are capable of competing with cheatgrass (Booth et al. 2003). Competition can occur at either the seedling stage mature stage or both.

Finally, the areas not invaded by cheatgrass might have been lacking in cheatgrass seeds. The proximity of cheatgrass plants to these native-dominated areas suggests that ample seed should have been available. However, in the absence of animals (particularly mammals) to facilitate dispersal, cheatgrass is limited to short-distance, wind-driven dispersal (Upadhyaya et al. 1986). This suggests the possibility that the boundaries of the cheatgrass distribution are synchronous with the boundaries of the seed dispersal at the time of the 2002 drought.

Methods

Study Site

This research is being conducted at Mt. Trumbull located in the Uinkaret Mountains in the northwestern corner of Arizona. The elevation of the site ranges from 2,000 to 2,250 m (6560 to 7380 ft.). Soils are derived predominantly from basaltic parent materials with a few areas dominated by volcanic cinders. The area receives an average of 35 cm (13.8 in.) precipitation per year (BLM Nixon Flats RAWS Site). This site is managed by the BLM and is part of the Grand Canyon-Parashant National Monument.

The overstory vegetation in the area is dominated by ponderosa pine (*Pinus ponderosa* Laws.) and Gambel oak (*Quercus gambelii* Nutt.). Additional tree species found in the area include New Mexico locust (*Robinia neomexicana* Gray), pinyon pine (*Pinus edulis* Engelm.), Utah juniper (*Juniperus osteospermus* (Torr.) Little), and quaking aspen (*Populus tremuloides* Michx.). The understory vegetation is similar to other ponderosa pine forests in northern Arizona and southern Utah, coupled with a strong influence from the surrounding Great Basin flora.

Experimental Design

We are using a randomized complete block design with a 2 x 2 x 2 factorial incorporating three treatments: native- vs. cheatgrass-dominated patches, clipped vs. not clipped, seeded vs. not seeded. Details of the clipping and seeding treatments are described later in this section. All possible combinations of these treatments are used in the study. In summer 2004, we established 10 blocks each containing one replicate of each combination of treatments. The native-dominated, not-clipped, not-seeded and cheatgrass-dominated, not-clipped, not-seeded plots will serve as controls for the study.

Plot Design and Selection

The study plots are 2 x 2 m (6.6 x 6.6 ft.). Within each study plot, 10 20 x 50 cm (8 x 20 in.) subplots will be sampled for a total sampled area of 1 m^2 (10.8 ft^2). To ensure an even distribution of the subplots across the study plot, five of the subplots were randomly selected within one 1 x 2 m (3.3 x 6.6 ft.) half of the plot. The placement of the remaining five was determined by the mirror image of the first five (Fig 5). The plots are surrounded by a 4 x 4 m (13 x 13 ft.) cattle exclosure made of 3-strand barbed wire.

The blocks were established across the entire landscape that had been invaded by cheatgrass. Within each block, plots were established based on the remnant native-dominated patches. For a patch to be accepted as a native-dominated plot in this study, it had to be large enough to contain the 4 x 4 m exclosure, be within a cheatgrass-dominated area, contain a substantial native perennial grass component (>35% of the vegetative cover), and have little or no cheatgrass within the plot area (no more than 1% of the vegetative cover). Since *Lupinus argenteus* is potentially a strong competitor with cheatgrass (Table 1), we also gave preference to areas where this species was present. This, however, was not always possible. Each native-dominated plot was paired with a cheatgrass-dominated plot. The cheatgrass-dominated plots were placed 20 m (66 ft.) from the center of the native-dominated plots in a randomly chosen direction. If the plot did not meet the minimum criteria for inclusion as a cheatgrass-dominated plot, it was rejected and another direction was randomly selected. This process was repeated until an acceptable plot was located. The criteria for acceptance as a cheatgrass-dominated plot were: cheatgrass must be the dominant species on the plot (based on visual estimation of cover); there must be cheatgrass present across the entire area of the plot; and the slope, aspect, and general soil type must be comparable to the paired native-dominated plot. Additionally, since the prescribed fire did not burn evenly across the landscape, each plot had to show evidence of having been burned in the treatment (charred wood, missing duff layer, char on trees, etc.) Each native plot was randomly assigned a treatment. The paired cheatgrass plot received the same treatment assignment.

Experimental Treatments

To experimentally manipulate plant competition, we will remove all aboveground vegetation from the clipped plots at two times. This first clipping occurred in August 2004 after the second set of vegetation measurements for that year. The second clipping will take place in mid-May of 2005, prior to the first vegetation measurements of that year. These clippings are timed to coincide with the peak growth of native perennials (late summer 2004) and cheatgrass (May 2005). In the second round of clippings, no native seedlings will be clipped from the seeded cheatgrass plots and no cheatgrass seedlings will be clipped from the seeded native plots. All vegetation will be clipped at ground level. Clippings will be oven-dried and weighed to document the amount of biomass removed from each plot.

Seeding will also occur at two times, fall of 2004 and spring of 2005. Cheatgrass plots will be seeded with the three dominant native species found on the study plots: *Lupinus argenteus, Elymus elymoides* (Raf.) Swezey, and *Pascopyrum smithii* (Rydb.) A. Löve. A mixture of seeds from all species will be broadcast sown by hand at a rate of 13kg/ha. This rate is consistent with past BLM seeding rates on Mt. Trumbull (Moore et al. 2003). All native seeds will be purchased from Granite Seeds in Lehi, UT, a common supplier of seeds for the BLM on other research projects on Mt. Trumbull. Cheatgrass seeds will be collected at Mt. Trumbull in the summer of 2004. These seeds will be sown in the seeded native-dominated plots in the same manor and quantity as the native seeds.

Vegetation Measurements

We will visually estimate aerial cover of all herbaceous and shrubby plants within the 20 x 50 cm subplots. All measurements will be taken as percentage of a square meter and pooled among the 10 subplots. The smallest measurement increment will be 0.1%. The annual species in this area are often small with very little aerial cover. Therefore, to obtain better resolution to variation in the annual populations, we will also tally the number of annual plants in each subplot. All plant species will be identified to the species-level unless reliable field identification is not possible. In such cases, the taxa will be identified to the generic level. Nomenclature will be consistent with the USDA Plants Database (USDA, NRCS 2004).

We will measure the vegetation twice a year; first in early summer when cheatgrass is at its peak growth and second in late summer when the native perennial plant community has responded to the monsoonal precipitation. Pretreatment measurements were taken in 2004. Posttreatment measurements will be taken in the summers of 2005, 2006, and 2007.

To obtain a general description of the forest stand structure, we measured basal area and overstory canopy cover on all plots. Basal area measurements were taken on a variable radius plot centered on the center point of the study plot using a basal area factor 10 wedge prism. Overstory canopy cover was determined using a vertical densitometer. Measurements were taken at the four corners of the 2 x 2 m study plot. All measurements were taken in August, 2004.

Soil Sampling

Soils were collected in late August 2004 coincident with maximum growth of the native perennials and the initiation of cheatgrass germination. There were two soil collections made at this time. One sample was tested at Mt. Trumbull for pH; the second was returned to Flagstaff for all other analyses. The rationale for sampling pH at Mt. Trumbull is purely convenience, since another study was conducting soil pH analyses at the same time. For each sampling event, soils will be collected at four points within the 1 m (39 in.) buffer zone between the study plots and the exclosure fence to a depth of 10 cm (4 in.) using a 4 cm (1.6 in.) soil corer (Fig. 5). The four core samples were composited for analysis, sieved through a 2 mm (0.08 in.) sieve, and all coarse organic material were removed. From each sample of the second soil collection, 10 g (0.4 oz.) were placed in 100 ml (3.4 oz.) of KCl solution and stored on ice for the available nitrogen analyses. All samples will be analyzed at the Analytical Laboratory at NAU, except for LOI and textures which will be analyzed at the Ecological Restoration Institute's Plant Ecology Lab. All analyses will be performed following Klute (1986).

Work Accomplished

In summer 2004, we established the 80 plots to be used in this study. All plots were permanently marked with rebar and we built a cattle exclosure around each plot to control for grazing. We measured the vegetation two times, once in June and again in August. Forest stand measurements were taken at the same time as the August vegetation measurements. We collected soil samples in August and initiated the clipping and seeding manipulations. The soil samples were prepared for analysis and the nutrient analyses are currently being conducted at the Analytical Laboratory at Northern Arizona University. At this time we do not have any results to report from the data we've collected.

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Table 1. From nearly 100 native perennial forbs and grasses documented to be present in this region, only 3 understory species formed patches not invaded by cheatgrass: *Lupinus argenteus*, a legume; *Elymus elymoides* and *Pascopyrum smithii*, both C3 grasses. Shown are the number of patches of these species that formed the terminus of patches of cheatgrass on thinned and burned areas of the Mt. Trumbull RCA. Of 110 point-line intercept transects, 65 contained patches of cheatgrass. (McGlone, 2004 unpublished data).

Species	Patches
Pascopyrum smithii	5
Elymus elymoides	7
Lupinus argenteus	9
Pascopyrum smithii *Elymus elymoides*	1
Elymus elymoides *Lupinus argenteus*	6
Elymus elymoides *Pascopyrum smithii* *Lupinus argenteus*	1

Figure 1. Reference photos of ecological monitoring plot 1133 at Mt. Trumbull at three time intervals. Topleft photo was taken in 1996, prior to treatment. The upper right and lower photos were taken in 1999 (one year posttreatment) and 2003 (five years posttreatment), respectively. The brown grass in the lower photo is cheatgrass.

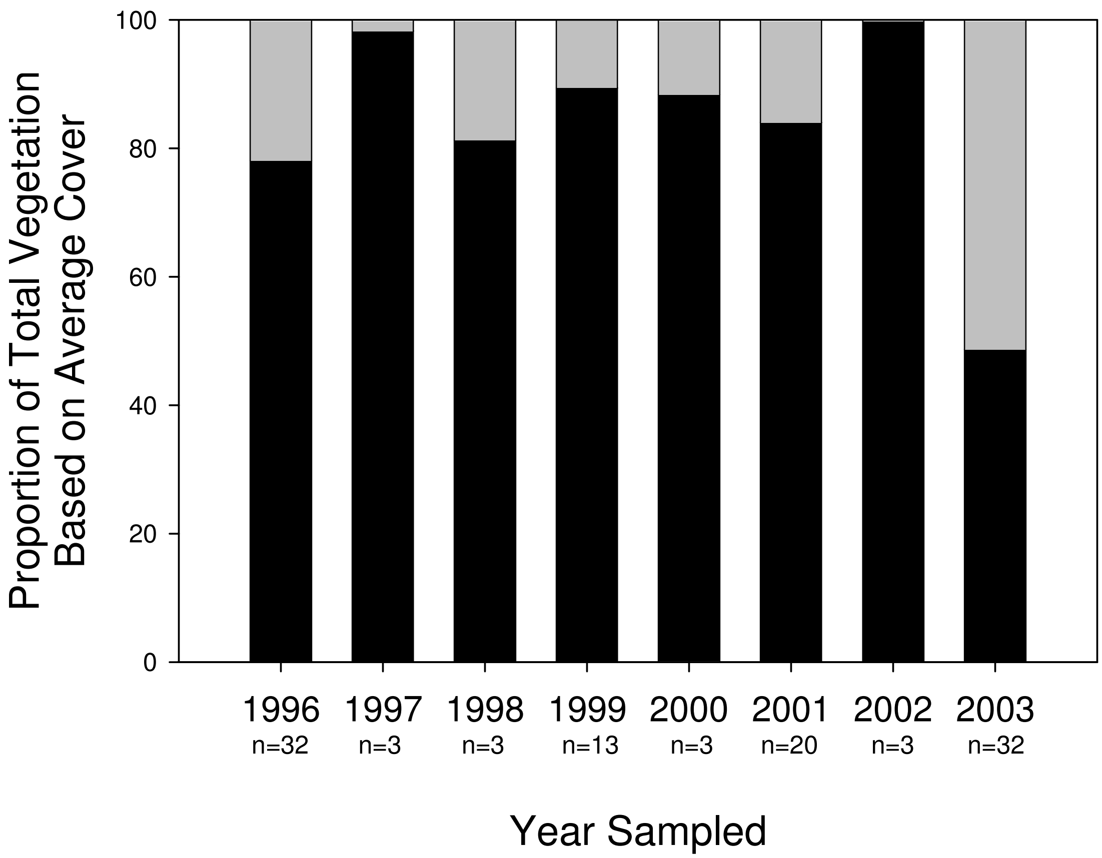

Figure 2. Proportion of total vegetation for native (black bars) and nonnative (gray bars) plant species based on average vegetative cover per plot. All 1996 data are pretreatment. Thereafter, all plots measured were posttreatment. Number of plots sampled is listed below the year sampled.

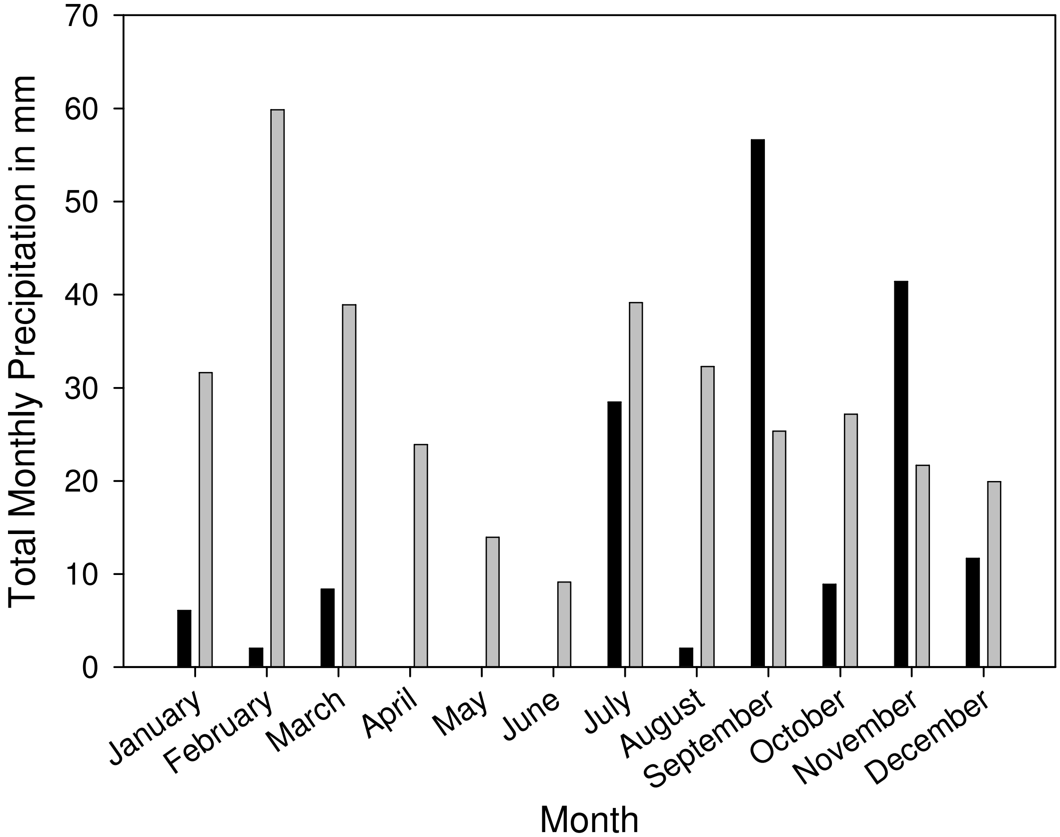

Figure 3. Monthly Precipitation on Mt. Trumbull for the year 2002 (black bars) versus the average from 1992-2003 (gray bars). The data were compiled from the Nixon Flats RAWS site. Note: 40mm of the September 2002 precipitation occurred in two days (September 6 & 7).

Figure 4. An example of a remnant patch of native vegetation in a cheatgrass-dominated matrix in the Mt. Trumbull region of the Grand Canyon/Parashant National Monument, AZ. The brown grass seen in this photo is cheatgrass. The dominant native plant species in the remnant patch are *Elymus elymoides, Lupinus argenteus,* and *Artemisia tridentata.*

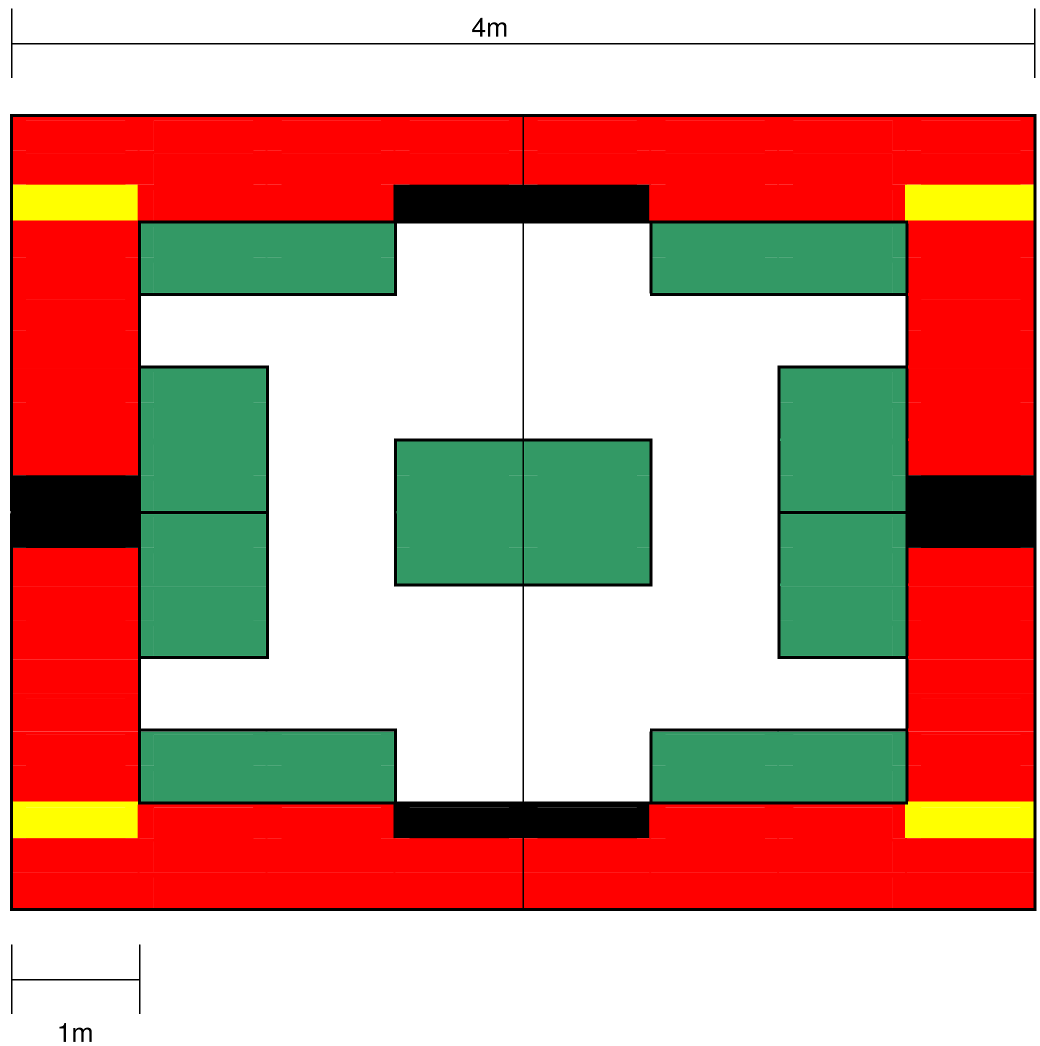

Figure 5. Stylized schematic of plot design. Red = buffer zone; Green = 20 x 50cm subplots; Yellow = soil sampling points for pH; Black = soil sampling points for all other analyses. The five subplots located on the right half of the study plot were selected at random. The five subplots on the left half of the plot were selected by taking the mirror-image of the layout of the right half of the plot.

6: Post-treatment mortality affects forest structure after ecological restoration treatments, Mt. Trumbull, Arizona, USA

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Abstract

Forest restoration is considered a promising approach for reducing severe fire hazard and emulating natural ecosystem structure and function in southwestern ponderosa pine-Gambel oak forests, but longer-term assessments are crucial for making sound and efficient decisions about treatment approaches. We remeasured a replicated experimental restoration site in northwestern Arizona five years after tree thinning and prescribed fire treatments. Basal area, tree density, and canopy cover declined in the treated units during the five-year post-treatment at a faster rate than in controls. Delayed mortality, not evident in the first year after treatment, was relatively modest in terms of density (13% in treated units, 10% in controls) but disproportionately affected large trees ("large" defined as ponderosa with diameter at breast height [dbh] ≥ 14.8 inches, other species dbh ≥ 7.9 inches). In absolute measurements, mortality was approximately twice as high in treated units: 4.4 trees ac^{-1} died in the treated units, of which 4.0 trees ac^{-1} were ponderosa pines, and 2.5 trees ac^{-1} died in control units, of which 1.9 trees ac^{-1} were ponderosa pines. But the relative decline was greater, averaging 34% of large ponderosa trees in treated units compared with 7% in controls, due to the lower post-treatment density of large ponderosa trees in the treated units. Compared with reference conditions of forest structure reconstructed in 1870, before fire exclusion and other perturbations to the ecosystem, the experimental blocks remain above reference levels in pine density and, in 3 of the 4 blocks, in basal area. Surviving pine trees grew significantly faster in treated units than in controls, indicating that the reference level of basal area could be regained in 6 years in the most-affected block. The cause of delayed mortality is unclear, as surviving and dying trees did not differ in measured heat effects (crown scorch, bole char). Severe drought prevailed through most of the period. Post-treatment mortality in the surrounding Mt. Trumbull site and on nearby study sites was lower than on the experimental blocks. Management recommendations include following procedures for minimizing fire effects on large residual trees, primarily through slash treatment, and careful monitoring of tree growth, mortality, and regeneration. Although mortality of large trees is a concern, the treated units

have characteristics of fast growth and low crownfire hazard, indicating that the treated forests will be relatively resistant to future drought and fire events.

Introduction

Severe wildfires pose a major challenge to forest sustainability and to vulnerable communities in western forests adapted to frequent surface fire regimes. Despite substantial investment at the national scale, relatively little is known about the effectiveness of forest treatments (GAO 2005), largely because new research initiatives such as the Fire/Fire Surrogates program have only recently begun to provide initial measurements of treatment effects (e.g., Stephens and Moghaddas 2005). Yet longer-term assessments are crucial for making sound and efficient decisions about treatment approaches, especially because changing climate is expected to impose novel environmental conditions (McKenzie et al. 2004), under which post-treatment forests may respond in unforeseen ways.

In the Southwest, ecologically based treatments have been tested for simultaneously reducing fuel hazards and restoring natural patterns of structure and function in ponderosa pine (*Pinus ponderosa*) forests (Covington et al. 1997, Moore et al. 1999, Allen et al. 2002, Romme et al. 2003). Methods have varied, but the general idea has been to conserve rare and important native ecosystem elements such as old trees, thin dense stands of young trees that established since circa 1880-1900, when region-wide fire exclusion, livestock grazing, and logging began (Swetnam and Baisan 1996), treat accumulated surface fuels, and re-introduce surface fire through prescribed burning or wildland fire use (Fulé et al. in press). Early findings have generally met expectations in terms of plant community recovery (Moore et al. in press), fire hazard reduction (Fulé et al. 2001), physiological response of old trees (Wallin et al. 2004), and ecosystem function (Kaye et al. 2005). However, the only studies with longer than immediate post-treatment data are from the 7.4-ac Gus Pearson Natural Area (Wallin et al. 2004) and an unreplicated research site near Grand Canyon (Fulé et al. 2002a, 2005).

Evidence that changes were occurring several years after treatment appeared in one of the first and largest well-monitored restoration studies in the Southwest, at Mt. Trumbull in the Grand Canyon-Parashant National Monument, northwestern Arizona. The research design included a replicated, controlled test on four experimental blocks, within a larger landscape-scale (~ 7,400 ac) pair of treated and control areas. Associated studies included plant community and wildlife research. The Mt. Trumbull restoration project has been a useful example for ecologists, managers, and policymakers. Tree thinning, forest floor removal around the base of old trees, and prescribed burning were completed in the experimental blocks by 2000. Forest density was significantly reduced and diameter distributions were shifted more closely toward pre-fire-exclusion patterns, while retaining structural diversity from block to block (Waltz et al. 2003). Crownfire fuel hazards declined dramatically (Fulé et al. 2001). However, severe drought prevailing in the Southwest since 1996 continued in 2000 and 2002, fostering region-wide tree mortality from lack of moisture and bark beetle attack (Breshears et al. 2005). In addition, we found patches of post-treatment mortality in the Mt. Trumbull landscape associated with heat damage from prescribed burning of broadcast slash (Jerman et al. 2004). It was clear that a simple pre- and post-treatment comparison was inadequate to assess the longer-term treatment effects.

We remeasured the Mt. Trumbull experimental blocks in 2005 to assess post-treatment forest dynamics in the presence of severe drought, focused on the following questions: (1) how has stand structure changed in the post-treatment period? (2) what are the survival and mortality patterns of large and old trees? (3) how might tree growth and new regeneration affect future forest development?

Methods

Study area

The study area is a ponderosa pine (*Pinus ponderosa*) and Gambel oak (*Quercus gambellii*) forest in the Uinkaret Mountains, northwestern Arizona (36° 22’ N, 113° 7’ W). Elevation ranges from 6,560 to 7,380 ft. Two remote automated weather stations are in the area: precipitation averaged 16.9 inches annually between 1992 and 2005 (Nixon Flats RAWS) and 12.8 inches annually between 1985 and 2005 (Mt. Logan RAWS). Drought affected the study area throughout most of the experimental period. Total precipitation at Nixon Flats and Mt. Logan was below average in six out of nine years between 1997 and 2005, including a severe drought in 2002, when precipitation levels were the third lowest in 100 years in Arizona (NOAA 2006a). The Palmer Drought Severity Index (PDSI) also indicated the occurrence of extended drought between 1994 and 2004 in northwestern Arizona (NOAA 2006b). Soils are derived from basaltic parent material. The two main soil types found in the study area are the Wutoma-Lozinta complex which consists of ashy-skeletal over fragmental or cindery, mixed, mesic Vitrandic Haplustepts, and Sponiker soils, classified as fine, smectitic, mesic Pachic Argiustolls (Natural Resources Conservation Service 2004).

Field

Four experimental blocks were established, representing the heterogeneity of the landscape in terms of vegetation, topography, and past management history (Figure 1). Each block was divided into two units, randomly assigned to control or restoration treatment. The blocks were described by Waltz et al. (2003). Briefly, block 1 (EB1) was dominated by ponderosa pine on shallow lava soils. Block 2 included abundant Gambel oak on basalt-derived clay soils. Block 3 was pine-oak forest at the highest elevation on cinder soils. Block 4 consisted of dense, young ponderosa trees and some pinyon (*Pinus edulis*) and juniper (*Juniperus osteosperma*), on basalt-derived clay soils. New Mexican locust (*Robinia neomexicana*) was scattered across most of the units.

Twenty permanent monitoring plots were established in each control and treatment area, for a total of 40 plots per experimental block, on a 196.8 ft grid representing a measured area of 17.8 ac in each experimental unit. Block 1 was established in 1997 and EB2-4 in 1998. Overstory trees taller than breast height (4.5 ft) were tagged and measured on a 4,306 ft^2 circular fixed-area plot. Measurements included species, diameter at breast height (dbh), and condition class (Thomas et al. 1979). Trees below breast height and shrubs were tallied by condition and height classes on a nested 1,076 ft^2 subplot. Dead woody biomass and forest floor depth were measured on a 49.9 ft planar transect in a random direction from each plot center, following the method of Brown (1974).

The forest restoration prescription was designed to rapidly emulate the historic forest structure, pre-dating European settlement and fire exclusion that began in 1870 (Moore et al. 1999), followed by re-introduction of surface fire. Living trees predating 1870 were retained. Wherever evidence of pre-1870 remnant conifer material was encountered (i.e., snags, logs, stumps, stump holes), three younger trees of the same species within a 30-60 ft radius were selected as replacements (Covington et al. 1997, Waltz et al. 2003). Fire-susceptible deciduous species, oak and locust, were not thinned. Thinning was carried out by commercial contractors and BLM crews in 1999. Block 4 was so dense that not all the trees marked for retention could be safely thinned and burned. Details on block-to-block differences were described by Waltz et al. (2003). Slash was lopped and scattered. Deep duff layers were raked 1-3 ft away from boles of all pre-1870 trees and snags. Treatment units were burned in the winter of 1999-2000. Six plots in the control area of EB1 (plots #14-19) and one in the control of EB2 (plot #21) were inadvertently burned during the prescribed fire; they were excluded from analysis of surface fuel loading by Waltz et al. (2003) and in the present study. In April 2000, a severe wildfire burned the control unit of EB3. The intense wildfire eliminated the unit as a valid untreated control for the block. In July, 2000, we established a new paired control. No pre-treatment data (1997-98) were available for the control 2000, but tree structure was similar to the original block 3 control (basal area 118.5 and 132.8 ft^2 ac^{-1}, respectively, and density 299 and 385 trees ac^{-1}).

All plots were remeasured in May-July, 2000, and five years later, May-June, 2005. Measurements of tree height and crown base height were added, and crown scorch and bole char were measured in 2000. Through repeated measurements of the same plots, we occasionally found and corrected errors, most notably when trees which had been listed as dead were found to be alive.

Analysis

The study was designed as a replicated experiment (n = 4). We used an alpha level of 0.10 to denote significant differences because of the small sample size; the same alpha level was used by Waltz et al. (2003) in the first report on this study. The response variables canopy cover, basal area, and tree density were compared with blocked repeated-measures ANOVA. The blocking factor was consistently significant. Variables were transformed (square-root or natural logarithm) where necessary to meet ANOVA assumptions of normality and homoskedasticity. Following a significant multivariate test statistic (Wilk's lambda) for change over time, univariate ANOVA comparisons within years were examined. We also made pairwise comparisons (t-test) between the three time periods of within-subjects factors, with Bonferroni-corrected P values. Tree growth data were compared with paired t-tests. Unless otherwise specified, P values in the results refer to univariate ANOVA test statistics. Even though the statistical testing approach served to summarize the four replicated blocks, we present the results by each individual block to illustrate similarities and differences.

Fates of individual large trees (ponderosa pine ≥ 14.8 inches dbh, other species ≥ 7.9 inches dbh) were followed to track survival. Because the original EB3 control unit was relocated, only trees from the other 3 control units and 4 treatment units were tracked. We tested differences in age, percent canopy scorch, and bole char of surviving vs. dying trees in the treated units with Kruskal-Wallis one-way analysis of variance.

Results

Changes in stand structure

Basal area, tree density, and canopy cover in the treated units declined not only in the immediate treatment period but also in the subsequent five years. Combining all four EB units, total basal area (Table 1, Figure 2) was indistinguishable in 1997/98 (*P* = .98) but was significantly reduced due to treatment, dropping 47% from an average of 154.2 ft^2 ac^{-1} before treatment to 81.9 ft^2 ac^{-1} afterward, while controls declined only 3% (153.7 to 149.4 ft^2 ac^{-1}) (*P* = .01). Control basal area declined an additional 1% from 2000 to 2005, but basal area in treatment units went down by an additional 22% to 64.0 ft^2 ac^{-1}, though this change over time was not statistically significant (paired t-test, *P* = .58). Total tree density did not differ between treatments in 1997/98 (*P* = .95), averaging 555 trees ac^{-1} (controls) and 556 trees ac^{-1} (treatments). Density declined by 7% in 2000 in controls, to 514 trees ac^{-1}, but 75% in treatments, to 140 trees ac^{-1} (*P* = .01). Further declines by 2005 of 10% in controls and 13% in treatments were not statistically significant (paired t-test, *P* = .13). Density in 2005 in control units averaged 461 trees ac^{-1} (83% of the pre-treatment level) and treatment units averaged 122 trees ac^{-1} (22% of the pre-treatment level). Canopy cover (Figure 3) did not differ between treatments in 1997/98 (*P* = .41) but cover was significantly reduced by treatment by 2000 (*P* < .001), declining an average of 35% while cover increased 5% in controls. All units declined between 2000 and 2005, dropping 13% in controls and 22% in treatments, but the post-treatment differences were not significant (paired t-test, *P* = .23).

Average size (quadratic mean diameter) of ponderosa pine trees increased substantially in the treatment, reflecting the cutting of small trees. Before treatment, pines averaged 9.1 inches in control units and 9.8 inches in treatment units, a slight (< .78 inch) but statistically significant difference (*P* = .07). Tree sizes in the control units changed little (8.7 inches in 2000, 9.01 inches in 2005) but increased significantly in treatment units (15.98 inches in 2000, a rise of 62%, and 15.47 inches in 2005) (*P* = .03). Oak sizes before treatment averaged 3.5 inches in control units and 4.48 inches in treatment units, changing to 4.2 inches and 5.07 inches, respectively, by 2005. The changes for oaks were not statistically significant.

Survival and mortality of large trees

Tree mortality in the treatment phase (between 1997/98 and 2000) averaged 416 trees ac^{-1}, including ingrowth of 1.2 trees ac^{-1}. Approximately 55%, mostly ponderosa pines, were identified as being cut. Another 13%, mostly small-diameter trees, were not encountered alive but the basal part of the trees could not be relocated so the cause of death is uncertain. Finally, 31% of the dead trees, mostly oak and locust, were not cut but died by 2000. Thus the non-cutting mortality in the treatment units may have ranged from ~ 134 to 186 trees ac^{-1}. For comparison, mortality in the control units averaged 33 trees ac^{-1} in the same time period and ingrowth averaged 3 trees ac^{-1} (ingrowth estimates exclude the EB3 control, because it was relocated). Post-treatment mortality between 2000 and 2005 averaged 40 trees ac^{-1} in the treated units, offset by ingrowth in the same period of 22 trees ac^{-1} (93% of which were oak and locust). Post-treatment mortality in the control units averaged 57 trees ac^{-1}, vs. ingrowth of 5 trees ac^{-1}.

A pattern of delayed mortality of old and/or large ponderosa pine trees appeared in the post-treatment period, in both treated and control units. During the treatment phase (1997/98-2000), an average of 3 large trees ac^{-1} died ("large": ponderosa pine ≥ 14.8 inches dbh, other species ≥ 7.9 inches dbh). Oaks comprised most of the dead trees, with ponderosa pines representing fewer than 0.4 tree ac^{-1}. Fewer than 2.5 tree ac^{-1} of any species died in the control units. In the post-treatment period 2000-2005, an additional 4.4 trees ac^{-1} died in the treated units, of which 4 trees ac^{-1} were ponderosa pines. Large tree mortality ranged from <1.6 trees ac^{-1} (EB4) to 8 trees ac^{-1} (EB3). The largest decline, in EB3, was primarily due to the death of large ponderosa pines (7.7 trees ac^{-1}); these trees represented 35.3 ft^2 ac^{-1} of basal area, making up 97% of the 2000-2005 decline in basal area in this unit. During the post-treatment period in the control, an additional 2.5 trees ac^{-1} died in the treated units, of which 1.9 trees ac^{-1} were ponderosa.

Large ponderosa pines were also removed by cutting, averaging 9.2 trees ac^{-1}. Based on increment cores collected from all ponderosa ≥ 14.8 inches dbh in pre-treatment sampling, 3% of the cut pines (0.8 trees) had center dates prior to 1870, though the trees were relatively young (1851 and 1860), and 2% could not be dated but were small in diameter (< 15.7 in). Therefore, cutting was 96-98% accurate in terms of the age-based thinning prescription. Retained large pines that could be dated were 45% older than 1870 (range 1464 to 1865) and 55% younger (1874 to 1932). Trees that died in the post-treatment period in the control units were significantly older (average center date, 1738; n = 8) than survivors (average center date, 1883; n = 108) (Mann-Whitney U test, $P < .001$). Dying trees in the treated units also tended to be older but the age difference was much smaller and not significant (survivors average center date, 1857 [n = 58]; died average center date, 1849 [n = 21]) (Mann-Whitney U test, $P = .42$). Mean crown scorch measured in 2000 was identical (30%) for large pines that survived and those that died by 2005 (Mann-Whitney U test, $P = .93$) and maximum bole char height was higher for surviving trees (6.9 ft) than dead trees (4.9 ft), though the difference was not significant (Mann-Whitney U test, $P = .15$). The combined effect of mortality and cutting was a reduction in the density of large ponderosa pines from an average of 23.1 trees ac^{-1} in 1997/98 to 9.5 trees ac^{-1} in 2005. Large pine densities in the controls declined from 25.8 trees ac^{-1} in 1997/98 to 23.95 trees ac^{-1} in 2005.

Tree growth and regeneration

Average annual diameter growth and basal area increment of ponderosa pine trees (Table 2) were significantly higher in treated units (average 0.066 inch yr^{-1} and 1.33 $inch^2\ yr^{-1}$) than control units (0.039 inch yr^{-1} and 0.50 $inch^2\ yr^{-1}$) (paired t-test, P = .03 and P = .02, respectively). Gambel oak diameter and basal area growth did not differ by treatment (paired t-test, P = .43 and P = .57, respectively).

Total tree regeneration was similar in control vs. treatment units before treatment (1,134 and 1,147 trees ac^{-1}, respectively). Regeneration declined in controls by 15% in 2000 and a further 27% in 2005, compared with increases of 2% and 68% in treatments (Table 3), resulting by 2005 in regeneration density approximately 228% higher in treatments (1,450 trees ac^{-1}) than controls (636.9 trees ac^{-1}). However, the differences were not statistically significant (Wilk's lambda, P = .18). The sprouting species Gambel oak and New Mexican locust made up the majority of regeneration in all periods.

Ponderosa pine seedlings declined with treatment, in contrast to the pattern of prolific oak and locust sprouting. In 1997/98, ponderosa seedlings had an identical average of 33.4 trees ac^{-1} in control vs. treatment units. By 2000, pine seedlings declined by 42% in controls (19.2 trees ac^{-1}) but by 89% in treatments (3.6 trees ac^{-1}). These patterns remained relatively consistent in 2005, with an average of 20.8 trees ac^{-1} in controls vs. 2.0 trees ac^{-1} in treatments. The changes were not statistically significant (Wilk's lambda, P = .21) despite the striking decline in pine seedling density in the treatments.

Duff depth (Table 4) did not differ between control and treatment units in 1997/98 (respectively, 1.5 and 1.2 inches, P = .32) but was significantly reduced in treated units by 2000 (1.3 vs. 0.6 inches, P = .02). Duff depth in treated units decreased 46% by 2000 and an additional 30% by 2005, in comparison to declines of 15% and 9% in control units. Litter and woody debris were highly variable and changes were not statistically significant (Wilk's lambda, P = .57 for litter depth and P = .13 for total woody debris).

Discussion

Five years after completion of the initial phase of forest restoration treatments at Mt. Trumbull, forest structure was substantially altered by tree mortality. Many of the changes were not statistically significant, however. Declining tree density *per se* was not inconsistent with the treatment objectives because a surplus of ponderosa pine trees was retained. Compared to pre-fire-exclusion reference conditions (1870), basal areas in the treatment units were still >100% or more higher in 2005 (Figure 2), indicating that the thinning prescription left an adequate margin of replacement trees. The exception was EB3 where the death of large trees caused basal area in 2005 to drop slightly below the 1870 level (Figure 2).

Although tree mortality in general does not necessarily constitute a problem, mortality of larger and older ponderosa pine trees is of concern, because these trees form important habitat elements, conserve genetic diversity over centuries, contribute to aesthetic qualities of the forest, and simply take a long time to replace. Assuming that the control units represent the "background" rate of mortality during the drought, large pines were approximately twice as likely to die in the treated units, ~ 2 vs. 4.8 trees ac^{-1}. Relative to the large pine density in 2000, post-treatment mortality corresponded to only 7% in the control but 34% in the treated units. These values were much higher than large-tree mortality in an unreplicated restoration study near Grand Canyon during a nearly overlapping post-treatment period (1999-2004), where pine mortality was 3% in the control unit and 9% in the treatment (Fulé et al. 2005). Unlike patterns observed elsewhere (McHugh and Kolb 2003, Fulé et al. 2005), surviving and dying trees on the experimental blocks at Mt. Trumbull did not differ in age, canopy scorch, or bole char. There is evidence that the thin lava soils at EB1 contributed to high fire-caused mortality (Fulé et al. 2002b), perhaps through heating of roots close to the surface (Swezy and Agee 1991).

Future development of forest structure will depend on the balance between continued mortality, growth of surviving trees, and new regeneration (Figure 4). Lingering effects of treatment-related damage may cause additional mortality, but most studies of fire-caused mortality indicate that the great majority of deaths are evident within 3 years after fire (McHugh and Kolb 2003, Fowler and Sieg 2004), so we expect that future mortality in the treated units would not differ from that in the controls. In fact, mortality may be reduced because evidence from physiological studies in the Pearson Natural Area restoration site indicates that for 7 years after treatment, old trees had higher predawn water potential, stomatal conductance, leaf nitrogen concentration, resin flow and tougher foliage than paired trees in the control area (Feeney et al. 1998, Stone et al. 1999, Wallin et al. 2004). If the surviving old trees at Mt. Trumbull respond similarly, they may be more likely to persist than those in the control units.

Ponderosa pine diameter growth was 70% higher and basal area growth was 165% higher in treated units than in the control. If the surviving trees in EB3, 27.9 pines ac^{-1} and 7.6 oaks ac^{-1} continued to grow at the average rates, the unit would regain the reference basal area within 6 years. In the other units, enhanced growth will maintain uncharacteristically high tree basal area, at the expense of understory vegetation.

Regeneration was highly skewed toward sprouting species, which commonly respond vigorously to disturbance and may come to dominate the forest (Gottfried 1980), although they are susceptible to fire and can be controlled with repeated burning (Harrington 1985). However, ponderosa pine regeneration densities in 2005 were the lowest reported in northern Arizona restoration treatments, averaging 2 trees ac^{-1}, compared with 2.8-16.5 trees ac^{-1} reported by Bailey and Covington (2002) at several sites in the Flagstaff area, approximately 124.2 mi southeast of Mt. Trumbull, and 4 trees ac^{-1} at Grandview, approximately 62 mi southeast (Fulé et al. 2005). Mast et al. (1999) calculated that the historically sparse forest structure of long-lived pine trees near Flagstaff could be maintained at a rate of only 1.5 seedlings ac^{-1} $decade^{-1}$ surviving to become established trees. Historical regeneration was also characterized by pulses of annual to multi-decadal width, rather than constant inputs (Savage et al. 1996, Mast et al. 1999). Since ponderosa seed trees remain well distributed across the Mt. Trumbull experimental sites and duff depth was significantly reduced by treatment, facilitating pine seedling establishment, it is likely that ponderosa regeneration will be successful in the coming years. Regeneration should be closely monitored, however.

The fact that the Mt. Trumbull experimental blocks were nested within a larger monitored treated-control landscape makes it possible to compare findings across multiple scales. In a landscape–scale study assessing restoration treatment effectiveness at Mt. Trumbull, Roccaforte (2005) found much lower mortality of old trees. Only 13% of the pre-1870 pines in the treated area died, compared with 9% in the control, indicating that pre-1870 pines in untreated areas were nearly equally vulnerable as those receiving restoration treatments. Treatments were completed in the landscape-scale study between 1996 and 2002; data were collected in 2003. The landscape-scale study included 117 permanent plots covering approximately 2,970 ac, split between treated and control areas. Leaving aside the possibility that additional mortality of large pines occurred between 2003 and 2005, the lower mortality of large trees across the larger landscape suggests that the experimental blocks were not representative of the broader area, at least in terms of tree survival. One of the four blocks, EB1, was located on shallow lava soils where we found that low-intensity burning even in the absence of any activity fuels led to high mortality (Fulé et al. 2002b). This soil type represented only a small portion of the landscape measured by Roccaforte (2005).

Management implications

The findings of post-treatment mortality and limited pine regeneration may be applied to suggest changes in treatment methods. For valued and vulnerable ecosystem components such as old trees, it would be useful to adopt any practices that might reduce the stress of treatment activities. Even though this study found no correlation between delayed mortality and heat effects (canopy scorch, bole char), prescribed burning of broadcast slash has been experimentally linked with tree death in other Mt. Trumbull study sites (Jerman et al. 2004). This source of damage can be controlled by slash compression before burning (Jerman et al. 2004) or by removing or piling slash.

A possible approach to the death of big trees would be to retain additional large trees during thinning, either through raising the ratio of replacement trees or through a diameter cap on cutting. However, there are tradeoffs to consider, because the spatial pattern of restoration thinning was designed to emulate the clumpy pattern of pre-fire-exclusion forests (White 1985). Moore et al. (1999) presented a stem-mapped example from the Mt. Trumbull area to illustrate how the selection of young replacement trees within a specified radius of remnant pre-1870 material resulted in the retention of trees close to where tree patches had been before as well as the re-opening of historical forest gaps. Understory plant productivity in such gaps can reach orders of magnitude higher than productivity under trees (Moore et al. in press). Since the biggest trees within the search radius were already the ones selected for retention, a rule specifying retention of more large trees would inevitably result in crowding or eliminating many gaps. The thinning guidelines already successfully discriminated between pre-1870 and younger trees, conserving the former category, and retained trees displayed high growth rates. The average size of pine trees in the treated units was already "large", approximately 15.7 in.

The only planned future treatment for the experimental units is repeated prescribed burning. Concern about the possibility of additional damage to old trees must be weighed against the role of surface fire in thinning the fire-susceptible oak and locust sprouts. Although in an idealized situation it might be attractive to view fire as the natural agent for maintaining open forest structure with minimal human guidance, the Mt. Trumbull forest is still quite removed from reference conditions and prescribed burns differ in many ways from the free-burning wildfires of the past. Therefore it may make sense for managers to use fire gently by minimizing it near vulnerable old trees, perhaps focusing on thinning sprouting trees with spot or ring ignition patterns.

Continued monitoring will be essential for evaluating the progress of the restoration activity. In most respects, the results have been positive in terms of forest structure, except for the undesired mortality of a relatively high proportion of large pine trees and the low level of pine regeneration.

Both factors should be closely tracked. The severe drought period from 1996 to 2002 may be an anomaly or a precursor of climate change (McKenzie et al. 2004, Breshears et al. 2005). Even under these difficult circumstances, however, overall forest structural impacts were relatively controlled, suggesting that restoration treatments may help pine-oak forests persist through climatic shifts.

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Table 1. Basal Area (ft^2/ac) at four experimental blocks at Mt. Trumbull, Arizona, in 1997-98 (pre-treatment), 2000 (post-treatment), and 2005. Values shown are **mean** and (standard error). 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 gambelii*, RONE = *Robinia Neomexicana*).

Block	Date	Total	JUOS	PIED	PIPO	QUGA	RONE
1C	1997-98	**117.6** (10.0)		**0.009** (0.009)	**116.3**(10.5)	**0.9** (0.9)	**0.9** (0.9)
	2000	**117.6** (10.0)			**116.7** (10.5)	**0.9** (0.9)	**0.04** (0.04)
	2005	**107.1** (13.9)			**106.3** (13.9)	**0.9** (0.9)	**0.04** (0.04)
1T	1997-98	**103.2** (7.4)			**101.0** (7.4)	**2.2** (1.7)	
	2000	**68.8** (7.4)			**67.1** (7.4)	**1.7** (1.3)	
	2005	**55.7** (7.0)			**54.0** (7.0)	**1.7** (1.3)	
2C	1997-98	**157.2** (12.2)	**0.4** (0.3)	**0.002** (0.002)	**137.2** (11.8)	**19.2** (3.9)	**0.2** (0.1)
	2000	**155.1** (11.8)	**0.4** (0.3)	**0.002** (0.002)	**135.9** (11.3)	**18.7** (3.9)	**0.2** (0.1)
	2005	**154.6** (12.2)	**0.4** (0.3)	**0.002** (0.002)	**139.4** (11.8)	**14.4** (3.5)	**0.2** (0.1)
2T	1997-98	**122.4** (16.6)	**0.02** (0.02)	**0.1** (0.09)	**75.3** (13.5)	**44.9** (11.8)	**2.2** (0.9)
	2000	**66.6** (10.9)	**0.009** (0.009)	**0.04** (0.04)	**7.7** (2.1)	**32.7** (8.7)	**0.3** (0.17)
	2005	**58.4** (8.7)	**0.02** (0.02)	**0.01** (0.01)	**25.3** (6.5)	**33.1** (8.7)	**0.2** (0.09)
3C[1]	1997-98	**132.8** (16.6)	**0.4** (0.4)	**0.9** (0.4)	**128.5** (17.0)	**2.6** (1.3)	**0.2** (0.09)
	2000	**118.0** (13.5)	**0.9** (0.9)	**0.2** (0.1)	**110.2** (13.5)	**6.5** (3.0)	**0.9** (0.4)
	2005	**122.4** (13.5)	**0.3** (0.2)	**0.2** (0.2)	**114.5** (13.9)	**5.7** (0.3)	**1.3** (0.9)
3T	1997-98	**172.0** (18.7)	**0.04** (0.04)	**0.2** (0.2)	**165.5** (19.6)	**4.35** (1.7)	**1.7** (1.7)
	2000	**74.5** (15.7)			**71.0** (16.1)	**1.7** (0.9)	**1.7** (1.7)
	2005	**38.3** (7.4)			**34.8** (7.8)	**2.2** (0.9)	**1.3** (1.3)
4C	1997-98	**207.3** (11.8)	**29.6** (9.1)	**5.2** (4.4)	**167.2** (16.1)	**5.2** (1.7)	**0.03** (0.02)
	2000	**206.0** (11.3)	**28.7** (9.1)	**5.2** (4.4)	**167.2** (16.1)	**4.8** (5.2)	**0.04** (0.03)
	2005	**204.7** (13.5)	**30.9** (10.9)	**5.7** (4.4)	**165.1** (17.0)	**3.5** (1.3)	**0.02** (0.009)
4T	1997-98	**219.1** (9.1)	**13.9** (3.9)	**1.7** (1.3)	**197.7** (10.5)	**5.66** (1.3)	**0.013** (0.01)
	2000	**118.0** (13.9)	**3.9** (2.2)	**0.4** (0.2)	**112.8** (10.0)	**0.87** (0.4)	
	2005	**103.2** (12.2)	**3.5** (1.7)		**99.3** (11.8)	**0.4** (2.6)	**0.004** (0.004)

1. The original control area for EB 3 was burned in a severe wildfire and replaced with a nearby site in 2000.

Table 2. Average annual diameter growth and basal area increment per tree from pre-treatment measurement (1997 for block 1, 1998 for others) to 2005 (five years after treatment) on experimental blocks at Mt. Trumbull, Arizona. SEM = standard error of the mean. Missing values mean that there were fewer than 10 trees in the category.

Block	*Average All Species*	*JUOS*	*PIED*	*PIPO*	*QUGA*	*RONE*
				Diameter growth, annual per tree (in)		
1C Mean	**0.047**			**0.047**	**0.017**	
(SEM) *N*	(0.002) 491			(0.002) 445	(0.003) 42	
1T Mean	**0.059**			**0.066**	**0.047**	
(SEM) *N*	(0.01) 84			(0.01) 58	(0.03) 26	
2C Mean	**0.029**	**0.07**		**0.032**	**0.023**	**0.026**
(SEM) *N*	(0.002) 887	(0.03) 11		(0.002) 493	(0.0015) 354	(0.0039) 28
2T Mean	**0.037**			**0.078**	**0.037**	**0.015**
(SEM) *N*	(0.002) 307			(0.01) 29	(0.002) 223	(0.004) 53
3C[1] Mean	**0.055**			**0.05**	**0.059**	**0.06**
(SEM) *N*	(0.003) 557			(0.0027) 476	(0.057) 30	(0.023) 46
3T Mean	**0.066**			**0.07**	**0.05**	
(SEM) *N*	(0.007) 76			(0.008) 54	(0.005) 14	
4C Mean	**0.023**	**0.017**	**0.020**	**0.024**	**0.022**	**0.0028**
(SEM) *N*	(0.002) 1646	(0.005) 190	(0.005) 96	(0.002) 1107	(0.003) 241	(0.01) 12
4T Mean	**0.039**	**0.0059**		**0.04**	**0.018**	
(SEM) *N*	(0.004) 315	(0.0125) 14		(0.004) 289	(0.007) 12	
				Basal area increment, annual per tree (in^2)		
1C Mean	**0.52**			**0.57**	**0.06**	
(SEM) *N*	(0.04) 491			(0.04) 445	(0.02) 42	
1T Mean	**1.28**			**1.74**	**0.24**	
(SEM) *N*	(0.17) 84			(0.22) 58	(0.04) 26	
2C Mean	**0.31**	**0.41**		**0.50**	**0.065**	**0.033**
(SEM) *N*	(0.02) 887	(0.2) 11		(0.04) 493	(0.01) 354	(0.01) 28
2T Mean	**0.42**			**1.46**	**0.38**	**0.026**
(SEM) *N*	(0.04) 307			(0.33) 29	(0.03) 223	(0.009) 53
3C[1] Mean	**0.62**			**0.61**	**0.81**	**0.62**
(SEM) *N*	(0.05) 557			(0.05) 476	(0.4) 30	(0.3) 46
3T Mean	**1.14**			**1.40**	**0.56**	
(SEM) *N*	(0.14) 76			(0.18) 54	(0.05) 14	
4C Mean	**0.24**	**0.093**	**0.13**	**0.32**	**0.040**	**-0.017**

(SEM) *N*	(0.02) 1646	(0.1) 190	(0.04) 96	(0.02) 1107	(0.01) 241	(0.03) 12
4T Mean	**0.65**	**0.025**		**0.71**	**0.096**	
(SEM) *N*	(0.06) 315	(0.2) 14		(0.06) 289	(0.05) 12	

1. The control unit of experimental block 3 was relocated in 2000, so tree measurements are available only for the period 2000 to 2005.

Table 3. Mean tree regeneration (trees below 4.5 ft in height) in no. of trees ac^{-1} at four experimental blocks at Mt. Trumbull, Arizona, in 1997-98 (pre-treatment), 2000 (post-treatment), and 2005. Regeneration is listed by species and height class (cm). C = control, T = treated.

		Pre-treatment (1997 – 1998)				Post-treatment (2000)				5 Years Post (2005)			
Block	**Species**	**Total**	0-16 inch	16-31 inch	31-54 inch	**Total**	0-16 inch	16-31 inch	31-54 inch	**Total**	0-16 inch	16-31 inch	31-54 inch
1C	PIPO	**30**	18	4	8	**18**	14	2	2	**16**	8	6	2
	QUGA	**10**	6	2	2	**38**	30	6	2	**24**	16	4	4
	RONE	**10**	2	6	2	**14**	4	8	2	**8**	0	8	0
	PIED									**2**	0	2	0
1T	PIPO	**69**	4	28	36	**8**	4	4	0	**4**	0	2	2
	QUGA	**87**	61	26	0	**95**	93	2	0	**172**	121	45	6
	PIED	**2**	2	0	0	**0**	0	0	0	**0**	0	0	0
2C	PIPO	**22**	4	14	4	**16**	6	6	4	**10**	2	8	0
	QUGA	**1717**	1016	510	190	**1312**	921	285	105	**994**	656	249	89
	RONE	**45**	24	18	2	**150**	87	45	18	**79**	28	38	12
	JUOS	**4**	0	4	0	**6**	2	0	4	**4**	0	4	0
	PIED	**8**	4	2	2	**10**	6	2	2	**6**	0	6	0
2T	PIPO	**34**	12	14	8	**6**	0	2	4	**2**	0	0	2
	QUGA	**1265**	832	352	81	**1409**	1379	28	2	**1308**	771	447	89
	RONE	**437**	123	198	115	**575**	411	103	61	**870**	180	379	312
	PIED	**2**	2	0	0	**0**	0	0	0	**0**	0	0	0
3C[1]	PIPO	**10**	4	2	4	**38**	14	18	6	**55**	8	38	8
	QUGA	**6**	6	0	0	**287**	279	8	0	**34**	34	0	0
	RONE	**976**	302	441	233	**91**	30	34	26	**101**	36	32	32
	PIED	**2**	0	2	0	**4**	2	2	0	**6**	2	4	0
3T	PIPO	**12**	8	2	2	**0**	0	0	0	**2**	0	2	0
	QUGA	**40**	40	0	0	**45**	43	2	0	**779**	488	283	8
	RONE	**277**	103	119	55	**526**	411	107	8	**464**	99	190	174
	PIED	**2**	2	0	0	**2**	2	0	0	**2**	0	2	0
4C	PIPO	**71**	69	0	2	**4**	2	2	0	**2**	0	0	2
	QUGA	**1553**	1144	291	117	**1200**	838	285	77	**1101**	755	253	93
	RONE	**28**	20	4	4	**22**	6	10	6	**24**	10	6	8
	JUOS	**40**	30	6	4	**16**	12	0	4	**38**	18	10	10
	PIED	**14**	10	4	0	**28**	16	12	0	**16**	6	4	6
4T	PIPO	**18**	2	12	4	**0**	0	0	0	**0**	0	0	0
	QUGA	**2298**	1389	713	196	**1271**	1257	14	0	**2164**	848	787	528
	RONE	**6**	0	2	4	**2**	0	2	0	**30**	6	8	16
	JUOS	**26**	12	14	0	**0**	0	0	0	**2**	2	0	0
	PIED	**10**	10	0	0	**0**	0	0	0	**0**	0	0	0

1. The original control area for EB 3 was burned in a severe wildfire and replaced with a nearby site in 2000.

Table 4. Forest floor depth and woody debris biomass at at four experimental blocks at Mt. Trumbull, Arizona, in 1997-98 (pre-treatment), 2000 (post-treatment), and 2005. Values presented are mean and (standard error). Woody fuels are classified by moisture timelag class (1H = 1 hour timelag, etc.).

Block	Year	Litter (inches)	Duff (inches)	1H (Ton ac^{-1})	10H (Ton ac^{-1})	100H (Ton ac^{-1})	1000H sound (Ton ac^{-1})	1000H rotten (Ton ac^{-1})
1C	1997-98	**0.29** (0.03)	**1.73** (0.26)	**0.20** (0.05)	**0.84** (0.17)	**1.47** (0.47)	**8.07** (4.82)	**1.81** (1.47)
	2000	**0.48** (.07)	**1.19** (0.19)	**0.05** (0.02)	**0.45** (0.12)	**1.28** (0.55)	**2.06** (2.00)	**0.18** (0.12)
	2005	**0.50** (0.09)	**1.32** (0.24)	**0.09** (0.03)	**0.45** (0.10)	**1.28** (0.71)	**4.64** (2.81)	**0.14** (0.10)
1T	1997-98	**0.19** (0.02)	**0.67** (0.09)	**0.10** (0.04)	**0.40** (0.06)	**0.70** (0.37)	**21.8** (12.5)	**3.89** (2.19)
	2000	**0.13** (0.04)	**0.66** (0.07)	**0.10** (0.03)	**0.71** (0.09)	**2.89** (0.58)	**5.09** (1.82)	**5.49** (2.66)
	2005	**0.36** (0.06)	**0.48** (0.07)	**0.06** (0.03)	**0.67** (0.17)	**2.24** (0.57)	**15.6** (6.02)	**0.06** (0.06)
2C	1997-98	**0.29** (0.02)	**1.03** (0.10)	**0.10** (0.03)	**0.81** (0.27)	**1.28** (0.33)	**6.25** (3.80)	**6.07** (4.43)
	2000	**0.88** (0.09)	**0.91** (0.13)	**0.10** (0.04)	**0.47** (0.08)	**1.01** (0.25)	**5.53** (2.62)	**2.94** (1.40)
	2005	**0.53** (0.09)	**0.85** (0.15)	**0.11** (0.03)	**0.56** (0.08)	**1.55** (0.44)	**3.18** (1.35)	**3.57** (1.69)
2T	1997-98	**0.46** (0.06)	**1.03** (0..19)	**0.10** (0.03)	**0.63** (0.11)	**1.08** (0.29)	**10.1** (8.97)	**0.72** (0.50)
	2000	**0.35** (0.07)	**0.31** (0.05)	**0.09** (0.03)	**0.47** (0.12)	**1.98** (0.41)	**7.85** (4.11)	**0.10** (0.07)
	2005	**0.32** (0.06)	**0.29** (0.04)	**0.08** (0.03)	**0.45** (0.09)	**2.24** (0.85)	**10.2** (5.67)	**0**
3C[1]	1997-98	**0.33** (0.04)	**1.64** (0.15)	**0.13** (0.03)	**1.04** (0.17)	**0.96** (0.38)	**3.12** (1.41)	**2.44** (1.78)
	2000	**0.76** (0.11)	**1.52** (0.22)	**0.12** (0.03)	**0.39** (0.08)	**0.70** (0.22)	**1.23** (0.25)	**4.51** (2.72)
	2005	**0.41** (0.06)	**1.15** (0.20)	**0.09** (0.02)	**0.41** (0.08)	**0.45** (0.21)	**2.41** (1.45)	**3.95** (2.19)
3T	1997-98	**0.35** (0.05)	**1.60** (0.20)	**0.06** (0.02)	**0.66** (0.19)	**0.70** (0.22)	**1.80** (0.72)	**4.13** (2.51)
	2000	**0.16** (0.02)	**0.41** (0.06)	**0.05** (0.01)	**0.56** (0.13)	**1.73** (0.39)	**6.69** (1.66)	**0**
	2005	**0.22** (0.05)	**0.21** (0.05)	**0.07** (0.02)	**0.53** (0.12)	**2.12** (0.92)	**10.3** (4.23)	**0**
4C	1997-98	**0.32** (0.04)	**1.47** (0.15)	**0.20** (0.06)	**1.12** (0.27)	**1.66** (0.72)	**4.59** (2.09)	**0.65** (0.43)
	2000	**0.87** (0.08)	**1.33** (0.13)	**0.20** (0.08)	**0.53** (0.14)	**1.66** (0.71)	**1.79** (0.91)	**4.43** (2.67)
	2005	**0.62** (0.08)	**1.11** (0.14)	**0.13** (0.04)	**0.59** (0.10)	**2.04** (0.69)	**4.95** (4.16)	**1.36** (0.85)
4T	1997-98	**0.45** (0.04)	**1.35** (0.10)	**0.18** (0.05)	**1.07** (0.25)	**1.60** (0.52)	**9.41** (5.17)	**0.27** (0.27)
	2000	**0.57** (0.12)	**0.83** (0.09)	**0.14** (0.04)	**0.74** (0.21)	**4.15** (0.76)	**14.5** (2.98)	**0.08** (0.05)
	2005	**0.29** (0.04)	**0.50** (0.06)	**0.11** (0.03)	**0.79** (0.21)	**4.15** (0.70)	**24.9** (5.89)	**0.09** (0.09)

1. The original control area for EB 3 was burned in a severe wildfire and replaced with a nearby site in 2000.

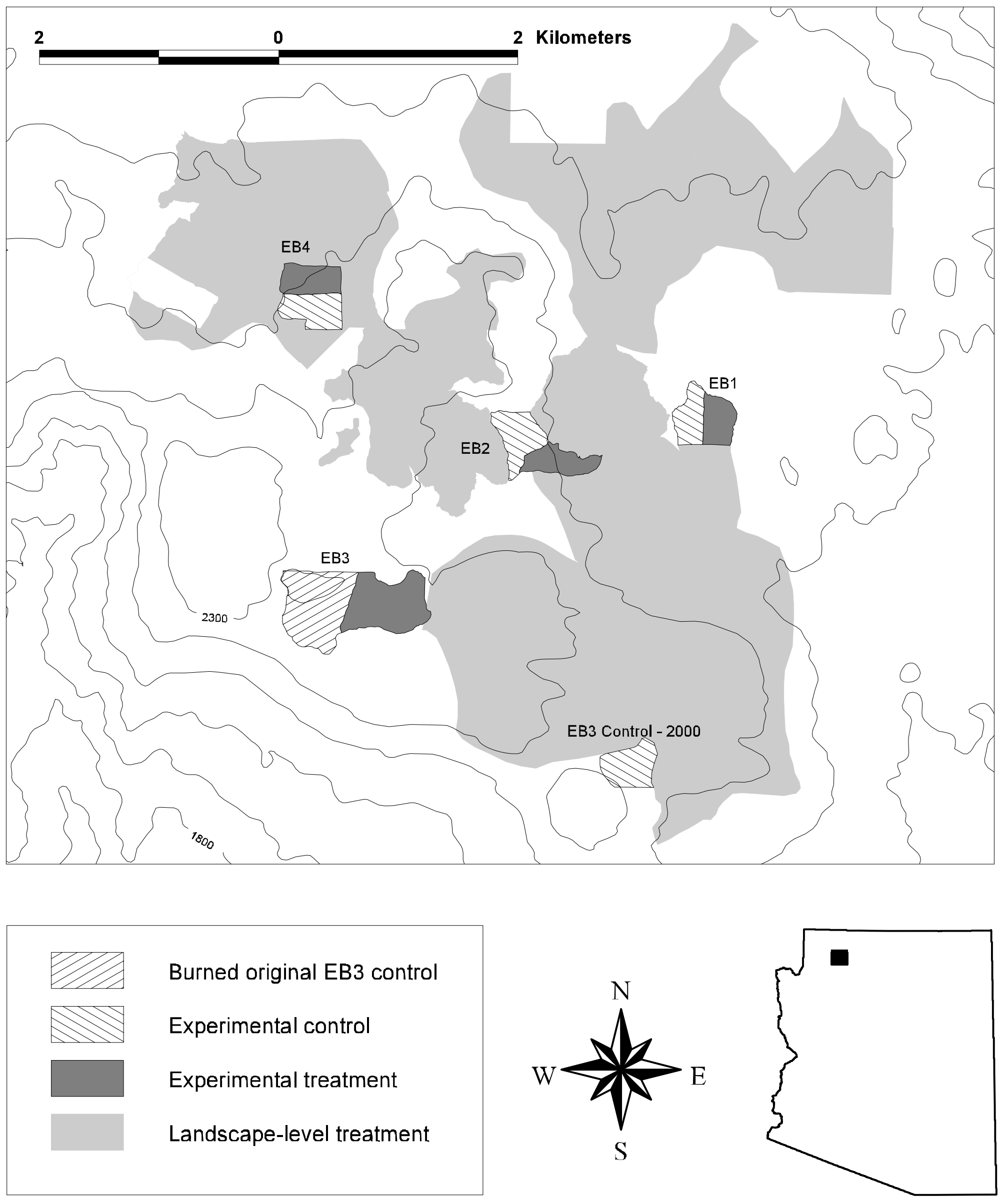

Figure 1. Study site in northwestern Arizona. The original control unit for experimental block 3 was burned in a severe fire in April, 2000. A new control was established to the southeast.

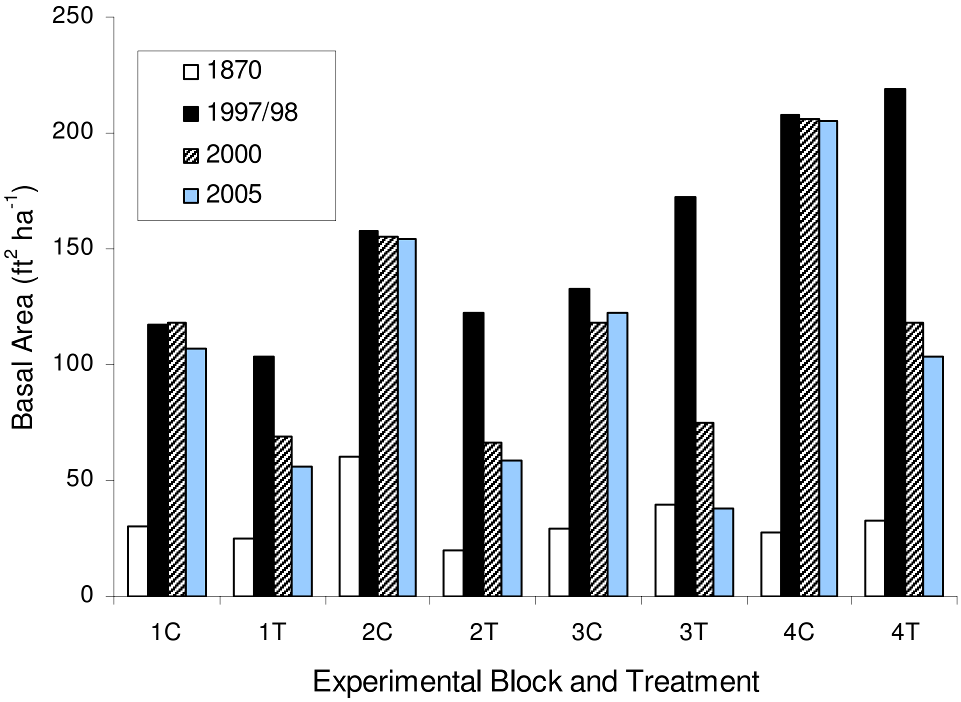

Figure 2. Basal area changes on four experimental blocks at Mt. Trumbull, Arizona, in 1997-98 (pre-treatment), 2000 (post-treatment), and 2005. Basal area in 1870 was reconstructed by Waltz et al. (2003).

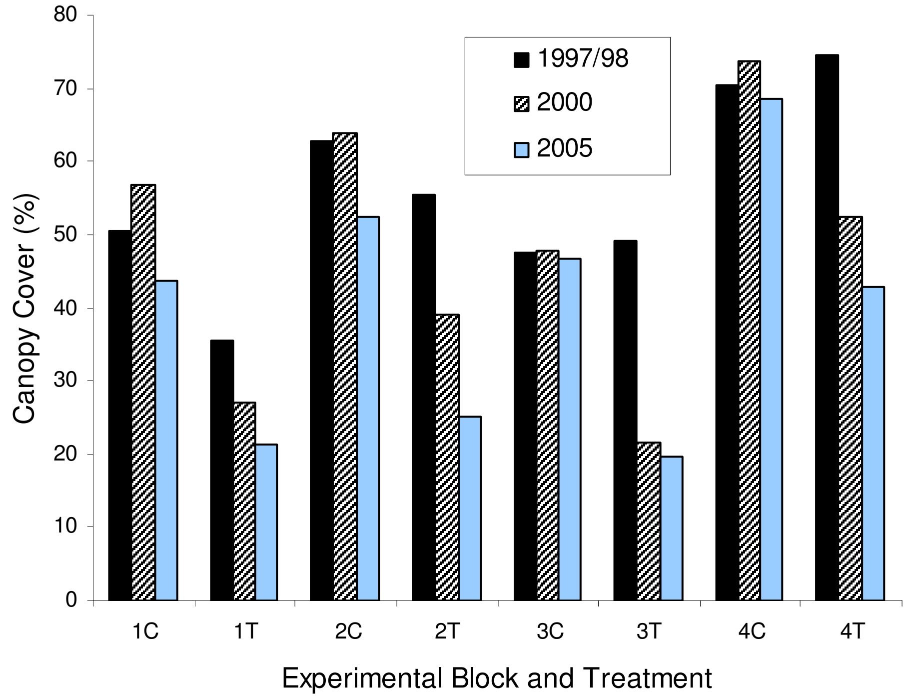

Figure 3. Canopy cover changes on four experimental blocks at Mt. Trumbull, Arizona, in 1997-98 (pre-treatment), 2000 (post-treatment), and 2005.

Figure 4. Changes on a single experimental plot from 1998 (pre-treatment, top), through 2001 (first year after thinning and burning), to 2005 (five years after treatment. In 2005, residual trees formed a patchy, savanna-like forest structure. Grasses in the foreground include both exotic and native species.

Appendix: Photos from 2005 Remeasurement

Figure A1. Little pine mortality, substantial cheatgrass (*Bromus tectorum*) in understory.

Figure A2. Mortality of large pines in background, Gambel oak clump in center.

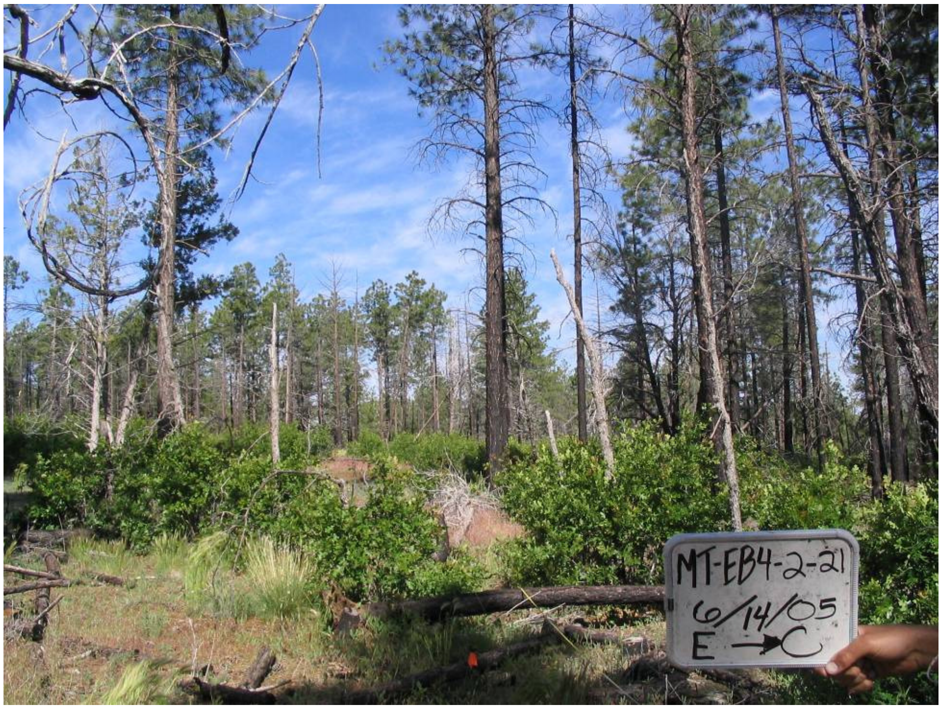

Figure A3. Dense oak regeneration.

Figure A4. Treated stand with high tree survival.