AN ABSTRACT OF THE THESIS OF

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Patricia S. Muir

Fuel reduction treatments are being applied to public lands, affecting significant acreage at considerable expense. This study compares the short term effects on a chaparral plant community of two different fuel reduction methods, brush mastication and "hand piling and burning" (HPB). *Ceanothus cuneatus* dominated the southwestern Oregon study sites where permanent paired plots were established on either side of treatment-control boundaries. Two years of sampling included a census of all vascular plant species within each plot and an abundance measure for each species. Species composition and abundance were analyzed using multivariate statistical techniques. Differences in species composition were detected for plots grouped by presence-absence of small *Ceanothus,* as well as plots grouped by abundance of mature *Ceanothus*. There were more *Ceanothus* seedlings in treatments than in controls. Abundance of all stages of *Ceanothus* was more reduced by the mastication treatment than the HPB treatment. The plot characteristic that had the most influence on species composition was the presence of a tree canopy which was positively correlated with abundance of perennial species. Both *Ceanothus* and oak canopy provided areas with higher abundance of natives and perennials compared to open areas that were dominated by exotic annual grasses. The effects of treatment were surprisingly small. Time passed since treatment, 1 yr or 2 yr, had a stronger effect on species composition than did the method of treatment. Species abundance and richness were greatest in the first year after treatment compared to the second year or to controls. In the mastication treatment, species abundance and richness were lower than in their controls in the second year after treatment. These measures were reduced in the second year HPB treatment

plots compared to the first year, but were still higher than in controls. In general, fuel reduction treatments appeared to increase the abundance of annuals, forbs, exotics, introduced weeds, and special status plants (taxa monitored by the Bureau of Land Management) during the first two years after treatments. Special status plants did not appear to be negatively affected by treatment, but treatment areas excluded known sites of occurrence for these species so there was scant data. The HPB treatment had a greater effect on plant communities than the mastication treatment because of the inclusion of fire rings remaining after the burning of piles. In the second year after treatment, fire rings had a higher proportion of annuals, exotics, and introduced weeds than their surrounding HPB treatment plots. *Ceanothus* germination was stimulated in fire rings but also occurred in the majority of plots, whether treatment or control. Resprouting of cut *Ceanothus* stems was also common in both types of treated plots. Short term evidence suggests that the HPB treatment may lead to an increase in weedy and/or exotic species and the mastication treatment may reduce species diversity. The HPB treatment may increase species diversity by allowing fire-cued species to establish. When applied to limited areas, both treatments will increase the heterogeneity of the overall chaparral community in the absence of wildfire, which also increases heterogeneity.

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The Effects of Two Fuel Reduction Treatments on Chaparral Communities in Southwest Oregon

by
Kendra G. Sikes

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Kendra G. Sikes, Author

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The Effects of Two Fuel Reduction Treatments on Chaparral Communities in Southwest Oregon

INTRODUCTION

Fuel reduction treatments have become an increasingly important aspect of public lands management in the western United States. Decades of fire suppression have increased fuel loads in forests and shrublands, increasing the danger of wildfire. Recent large-scale wildfires and expanding construction of housing in proximity to wildlands have brought this danger to the attention of the public. The primary purpose of fuel reduction is to remove combustible plant material that can act as fuel for a fire, thus reducing the danger of wildfire. Other potential benefits include rejuvenation of senescent shrubs, increasing availability of forage for wildlife (Lillywhite 1977; Rogers et al. 2004), and improving conditions for forbs and grasses.

A recent inventory of central and southern California chaparral found that 16 to 27% of areas tracked burned between 1984 and 1994 (Fried et al. 2004). Using fire hazard classes estimated from the amount of dead material and shrub cover, chaparral older than ten years can generally be considered a fire hazard that would burn readily under the right conditions (Fried et al. 2004). Increasing amounts of property loss due to wildfire have been attributed to urban sprawl and population growth in southern California, where chaparral dominates, rather than fire suppression, since complete fire exclusion has not been achieved (Keeley 2002). The key factor in reducing property losses is reducing the danger of home ignition. Experimentation has demonstrated that a 40-m buffer of non-flammable areas around homes can be adequate protection (Cohen 2000). Since regeneration occurs quickly, there is some doubt about the long term effectiveness of brush clearing. *Ceanothus* species may revert to closed crowns within five to seven years (Green 1977).

The Medford District of the Bureau of Land Management (BLM) in southwestern Oregon manages a checkerboard of public lands intermixed with private and inhabited lands. Over the past five years the BLM has allotted significant funds to reducing fuels in both chaparral and oak woodland. Managers and scientists at the BLM are concerned about whether their management techniques will adversely affect these already degraded communities. Therefore, they have chosen to fund research into the effects of the two types of fuel treatments that they are utilizing.

The two treatments compared in this study are brush mastication and "hand piling and burning" (HPB). Brush mastication, also known as "slashbusting," uses heavy machinery on caterpillar treads that is equipped with a rotating brushcutting disk on a moveable arm. The machine can shred woody material up to a foot in diameter. The chips and shredded debris that result are left where they fall. In contrast, in the HPB treatment, fuel removal is accomplished by crews with chainsaws that pile the cut material for later burning. Burn piles are relatively small, approximately 3 m in diameter and up to 2 m high. A few sheets of black plastic are incorporated into the piles to keep the center of the pile somewhat dry during the rainy season to allow ignition.

In the areas I studied, *Ceanothus cuneatus* (buckbrush) is the dominant shrub and the main target for removal with fuel treatments. This species dominates the chaparral of central and northern California and Oregon's Rogue Valley, "the northernmost outpost of typical North American chaparral" (Detling 1961, p. 354). Chaparral is found in regions with the hot dry summers and cool wet winters of a Mediterranean climate. It is a community often dominated by a single species of shrub with small, evergreen leaves. According to Detling (1961), chaparral is the most xeric vegetation type in southwestern Oregon, with *C. cuneatus* being the most drought and heat tolerant of the area's woody dominants. In the Rogue Valley, *Ceanothus* chaparral is usually found adjacent to the more mesic habitat of *Quercus garryana* oak woodlands, while *Arctostaphylos viscida* (manzanita) is found overlapping the two adjacent zones (Detling 1961). Generally the chaparral is restricted to rocky soils that are insufficient to support trees (Callaway & Davis 1993; Odion et al. 2004). It also intergrades with grassy areas without any woody vegetation.

Chaparral is important habitat for both plants and animals. "Approximately 48 bird species regularly breed within chaparral habitats of southwestern Oregon" (Altman et al. 2001, p. 276). Fifteen species of birds that are regularly associated with westside chaparral habitat have significantly declining population trends, while two species associates have significantly increasing ones. One of those declining, the wrentit, is obligate or semi-obligate to chaparral shrub during breeding season.

Five other bird species are not declining but share the same obligate or semi-obligate status to chaparral as the wrentit. Three bird species and one mammal, the California kangaroo rat, can be considered endemic to the chaparral, as far as their Oregon distribution is concerned. Twenty-four species of herpetofauna occur there. Western fence lizard and southern alligator lizard are found at their highest southwest Oregon densities in *Ceanothus*-dominated chaparral (Altman et al. 2001). Shrubs provide important cover for lizards and small mammals (Lillywhite 1977).

Chaparral is a flammable and fire-adapted plant community. Leenhouts (1998) estimates a fire return interval of 20-40 years for chaparral in the pre-industrial United States. Many chaparral plant species have relatively short life spans, and are dependent on fire for reproduction or creation of necessary habitat. A range of regeneration strategies coexists in this diverse plant community, making chaparral resilient to fire and other disturbances (Lavorel 1999). *Ceanothus cuneatus* is one species cited in the literature as a fire-dependent obligate seeder, because fire promotes seed germination and adult plants generally do not survive fire (Keeley 1992a). In the absence of fire, long-term shrub cover may eventually crowd out the herbaceous and graminoid component of shrublands potentially facilitating their invasion by weeds following wildfire or management intervention.

Fire suppression is often assumed to be detrimental to the health of ecosystems in which fire is known to play a role. Recently burned chaparral supports higher and healthier deer populations due to the "increased availability and diversity of palatable, high-nutrient browse" (Lillywhite 1977, p. 369). Older *Ceanothus* tend to decline and have greater proportions of dead and flammable material within each shrub. Long unburned stands will lead to hotter fires when burns do occur, because of fuel buildup, and these temperatures may be outside the natural range of fire intensity (Menges & Hawkes 1998). Increased fire temperature may kill propagules (both seed and resprouting tissue) that would normally survive fires that occurred at more frequent intervals. Therefore structure and composition of post-fire communities may be affected by fire suppression and fire reintroduction (Menges & Hawkes 1998). Fire suppression may also lead to loss of local species through outlasting the life cycle of plants that require fire for new recruitment.

Fuel reduction treatments currently in use may ameliorate or magnify the community-altering effects of fire suppression. Brush mastication reduces fire danger by making fuels less likely to ignite and carry fire. It releases areas from the shade of a shrub canopy but adds debris to the soil surface. The HPB treatment also creates open areas, but with little woody debris remaining in comparison to mastication. Small patches of the HPB treatment area are subjected to high intensity fire from the brush piles while most areas receive no heating. Neither mechanical thinning method removes the litter layer or approximates the soil surface conditions found after fire (Kauffman 2004).

The objective of this study is to determine how both fuel reduction treatments affect plant community composition. Are weedy exotics promoted by either treatment? Are special status plants, those rare plants monitored by the BLM, affected? This study examines the overall variation in the plant community and compares the short term effects of the two treatments on species occurrence and abundance.

METHODS

Study Area

This research was conducted on public lands in the Butte Falls Resource Area (42° 32' N, 122° 37'W) of the Medford District BLM. Study areas were located within a 10 km radius in the foothills of the Cascade Range in the Rogue River watershed, northeast of Medford, Jackson County, in southwestern Oregon. Elevation at the sites ranges from 509 to 917 m (1670-3010 ft). Precipitation varies with elevation, ranging from normal annual precipitation in Medford at 395 m (1297 ft) of 46.66 cm (18.37"), to 91.49 cm (36.02") in Butte Falls at 762 m (2500 ft) (NOAA 2004). Only 20 percent of the annual precipitation falls between April and September (Johnson 1993). The normal average July temperature is 20.7°C (69.3°F) at 482 m (1580 ft; Lost Creek Dam), and increases to 22.6°C (72.7°F) at Medford. Normal average January temperature of 3.2°C (37.8°F) at 482 m differs little with elevation change in the study area (NOAA 2004).

The Rogue Valley is bounded by the Siskiyou Mountains to the west and south, and by the Cascade Mountains to the east. The valley floor, up to about 750 m in elevation, has been categorized as the Interior Valley Zone which includes oak woodlands, coniferous forests, grassland, and chaparral (Franklin & Dyrness 1973). At higher elevation, mixed-conifer begins to dominate (Franklin & Dyrness 1973). Chaparral occurs in the driest areas up to 1100 m (Detling 1961).

South-facing moderate slopes predominate at my study sites, ranging from 2-17° or 4-31 percent slope. Soils have been mapped as Carney clay, McMullin-Rock outcrop complex, and Medco-McMullin complex (Johnson 1993). Both the Carney clay and Medco soils have very slow permeability. The Carney soil has a high clay content throughout and therefore may produce areas of standing water. The clay subsoil of the Medco soil keeps the water table high in the winter months and limits the effective rooting depth. The McMullin soil is moderately permeable, but shallow, supporting mainly shrubs, grasses and forbs (Johnson 1993).

Ceanothus cuneatus (buckbrush) is the dominant woody plant at all of the study sites. *Quercus garryana* (Oregon white oak) is common, with some oak canopy occurring in about half of the plots sampled. Herbaceous vegetation is dominated by annuals, especially exotic annual grasses including *Aira caryophyllea* (silver hairgrass), *Bromus hordeaceus* (soft brome) and other bromes, and *Taeniatherum* *caput-medusae* (medusahead). All sites have a history of livestock grazing. No evidence of recent fire or clearing was discovered from a search of aerial photos on file at the BLM (taken approximately once each decade since 1966).

Treatment Prescription

In both HPB and mastication treatments in chaparral, approximately 75% of the shrub cover was removed. Prescriptions varied slightly by management unit, but in general, all shrubs under or within 3 m of tree crowns were removed. Shrubs in the open were thinned with the intent to space clumps with an average of 7.6 x 7.6 m between clumps. Shrubs were clumped to provide islands of habitat. An effort was made to locate special status plants and exclude them from treatment by buffering any sites found.

The chosen sites were dominated by *Ceanothus* and therefore other shrubs and trees were at low density and less likely to be thinned. Some removal of small trees did occur, as evidenced by oak stump resprouts. The prescription sought to leave the largest and healthiest trees, varying tree spacing accordingly. Conifers greater than 18 cm in diameter at breast height (DBH) were not removed. Hardwoods greater than 25 cm DBH were not removed. Trees that were smaller than those thresholds were retained with a 7.6 m spacing.

Management units consisted of contiguous areas with a single treatment prescription that was contracted out to a single entity to carry out treatment. Units in this study varied in size from 12,000 sq m to 0.75 sq km (3 to 184 acres). Treatments took place over periods varying from several days to several months, so that treatment conditions often varied from place to place within a unit.

All burning of hand piles for the HPB treatment was accomplished in the months of November or December. Cutting and piling, however, began as early as November of the previous year. Mastication treatments were carried out over many months as well. Those that were treated in 2003 were completed in November or December. Those that were treated in 2002 were completed in April, May or June.

Field Methods

Plot establishment

Permanent paired plots were established in the summer of 2003 near the boundaries of fuel reduction units slated to be treated the following fall or winter. Areas were sought where the plant community and environment appeared to be similar on both sides of the boundary. Plot pairs were randomly located on either side of such boundaries. Positioning was determined using a random number chart to dictate a distance along the boundary and a perpendicular distance into the management unit or control area. From the resulting point, a random angle was used to determine the direction of the 50 m tape that would form one edge of the plot. The same random angle was used to determine the direction of both plots in the pair, while different random distances were used to determine their starting points. Random locations were rejected and redetermined if they resulted in pairs that were not similar in community or slope and aspect. Each plot was at least 15 m from the marked treatment boundary. Each plot was a 50 m x 1 m belt transect. This plot configuration was chosen to be compatible with that already established for a related monitoring project in an adjacent BLM Resource Area.

In 2003, 26 plots were established, including six pairs of treatment and control plots for each treatment type (mastication and HPB; Table 1). The other two plots were a matched pair of pre-treatment plots, on either side of a boundary between a mastication unit and an HPB unit, without a control. Sampling took place between June 26 and August 15, 2003 and between May 5 and July 28, 2004. Only 18 of these plots were sampled in both 2003 and 2004, because units that went untreated were not resampled (Table 1).

Additional plots were established in 2004 near the boundaries of fuel reduction units that had already been treated (Table 1). Some were located in areas that had been treated in 2003 at similar times as the pre-treatment plots set up the previous year. These nine retrospective plots were added to compensate for pairs established in 2003 whose scheduled HPB treatments had not occurred. They resulted in five new pairs since one of the treatment plots was matched with a control established in 2003. Other plots established in 2004 were located in areas that received treatment in 2002. For retrospective pairs, judging whether treatment and control were well matched required more conjecture than with prospective pairs. I reduced the required distance between plot and boundary to 7.5 m, since I no longer needed to compensate for differences between the boundary flagged for treatment and the actual treatment boundary. Otherwise the same procedure for locating plots was used as in 2003. Twenty-five plots were established for 2002 treatments. As in 2003, there were six pairs installed for each treatment type (Table 1). The extra plot was a treatment plot that I decided was too different from its matched control only after its sampling procedure was completed. I replaced it with another randomly located treatment plot to match its paired control.

Table 1. Summary of plots established for this study. Abbreviations include: **Mastic.** = mastication; **trt** = treatment; **Diff t-c** = the difference between treatment and control plots sampled in 2004, which is one of the two 2004 datasets.

Number of plots	**Year estab-lished**	**Years sampled**	**Year treated**	**Type of treatment**	**Special exceptions**	**Included in datasets**
6 pairs	2003	both pre- & post-trt	2003	Mastic.	(1 trt plot deleted from *Ceanothus* stage data)	All
1 pair	2003	both	2003	HPB	-	All
3 pairs	2003	pre-trt	-	HPB	Scheduled trt not performed	Pre-treatment
1 pair	2003	pre-trt	-	HPB	Statistical outlier	-
1 pair	2003	both pre- & post-trt	2003	HPB	Piled but not burned	All but HPB trt
1 pair	2003	both pre- & post-trt	2003	both	Two trts, no control	All but MRBP & Diff t-c
2003 TOTAL: 26 plots						
4 pairs	2004	post-trt	2003	HPB	-	both 2004
1 plot	2004	post-trt	2003	HPB	Trt plot matched with a 2003 control	2004 but removed from MRBP
6 pairs	2004	post-trt	2002	Mastic.	-	both 2004
6 pairs	2004	post-trt	2002	HPB	-	both 2004
1 plot	2004	post-trt	2002	HPB	Unmatched trt plot	2004 but removed from MRBP & Diff t-c
2004 TOTAL: 34 plots						

In total, 60 permanent plots were established, located over 21 different management units. Some management units were adjacent to each other, so that some pairs in different units were less distant from each other than pairs in the same unit. The distances between pairs ranged from approximately 100 m to 20 km. The plots were distributed over 13 survey sections, each equal to 2.59 sq km (1 sq mi) in area.

Sampling methods

One 50 m side of each plot was permanently marked using two 1.5 m long posts of 1.27 cm (½ inch) diameter rebar sunk vertically approximately 0.5 m into the ground. A GPS was used to record and map the location of each post. To permit pinpointing of the position in the event of a stolen post, four nails were pounded into the ground demarcating an X with the post in its center. Posts were marked with flagging or covered with a length of PVC pipe to make them more visible.

Each plot was bounded by a straight 50-m tape on one side, strung between the two posts. Meter sticks abutting and perpendicular to the tape were used to delimit the 1 m width of the plot for sampling, moving them along the tape as needed. Generally, the plot lay either to the south or west side of the tape. The plot's position relative to the tape was recorded for each plot.

Photos were taken at all plots at the time of sampling. Two photos were taken from the top of each post. One photo centered on a 22 x 28 cm sign placed 5 m away along the tape and a second photo centered on the top of the opposite post. Therefore each plot had two photos facing one direction down the tape and towards the opposite end, and two more from the opposite post facing the other direction. Plot photos are provided on the data CD included with this thesis (see Appendix 4).

I estimated tree canopy coverage for each plot using a line intercept method. Each canopy occurrence that overlapped the tape was recorded (in cm) by determining where a perpendicular extension from the canopy would intersect the tape. The extent of fire rings formed by piles of the HPB treatment was also measured by recording the distance of overlap along the tape.

Physical data recorded for each plot included elevation according to the GPS and aspect estimated using a compass with a declination of 20°. A clinometer was used to estimate a minimum value for the slope of the plot, recording the angle between the top of the posts. Cover estimates were made for bare soil (including gravel-sized rock), litter, woody debris, standing dead (*Ceanothus cuneatus* only, rooted in plot and greater than a foot in height), and rocks, using the following cover classes: 0 = none; 1 = <10% cover; 2 = 10-25% cover; 3 = 25-50% cover; 4 = >50% cover. Maximum litter depth was estimated by poking a ruler into the litter in several places where it appeared thickest within the plot. Diameter at breast height (DBH) of any trees rooted within the plot was also recorded.

In 2003, presence of any *Ceanothus* smaller than 0.3 m (1 ft) tall was recorded, as an indicator of *Ceanothus* regeneration. In 2004, more detailed information on *Ceanothus* life stages was recorded. The number of *Ceanothus* in each stage was recorded separately, so that their abundance could be combined or not as needed. The four stages of *Ceanothus* were: seedlings (plants that germinated that year with little or no woody tissue), smalls (plants with some woodiness up to 0.3 m tall), resprouts (plants that had been cut to the base during treatment but had put up new growth), and uncut (mature plants greater than 0.3 m tall).

All vascular plant species rooted in each plot were recorded using broad abundance classes. Species abundance was indicated using the following abundance ratings, taken from the USDA/Forest Service Health Monitoring protocol for lichen monitoring (USDA/FS 2002).

0	Absent, not present in plot
1	Rare, 1-3 individuals in the plot
2	Uncommon, 4-10 individuals in the plot
3	Common, more than 10 individuals but less than 50% cover
4	Abundant, greater than 50% cover

Species were recorded starting at one end of the plot and using the meter sticks to outline each section censused, moving a pipe to delimit the next new section to census, until the entire length of the plot was searched. Length of time to complete census was recorded, varying from 1½ to 3½ hr. Samples of unknown or uncertain species were taken for later identification or verification. Some of these samples, as well as other collections from the areas around the plots, were later accessioned by the Oregon State University herbarium (see Appendix 3).

Because the species composition of fire rings resulting from burned piles appeared to be quite different from the rest of an HPB treated plot, and because the proportion of the plot covered by piles was small, a separate censusing procedure was undertaken to describe their species composition. Three fire rings were selected for each HPB treatment plot. The fire rings that were closest to the plot, or most overlapped the plot, were selected. If two fire rings were equidistant from the plot, the one that best contributed to covering the entire span of the plot (in combination with the others selected) was chosen.

A circular plot with an area of ¼ m^2 (28.2 cm radius) was placed at the visually estimated center of the fire ring. Species abundance within the circular plot was recorded using the same abundance codes as above. Percent cover of rock, litter, and woody debris was also estimated using substrate cover codes described above. Each fire ring plot center was marked using a wire flag, so that the same plot may be recensused in future years, and the location of each sample fire ring was recorded in relation to the 50 m tape and a post (see Appendix 4).

Data Analysis

Combining data collected in 2003 and in 2004, the initial dataset consisted of 233 taxa in 78 plots. This dataset, along with other data, is provided in a data CD (Appendix 4). Eighteen plots were read in both 2003 and 2004, which accounts for 36 of the 78 readings. Of the 233 taxa, most were identified and recorded as species, however some species that were not reliably distinguished in the field were combined with other species in their genus (see Appendix 1). Multivariate analyses were performed using PC-ORD for Windows, Version 4.25 (McCune & Mefford 1999). Sørenson distance was the default distance measure of choice for community data. This city-block type of distance measure is best suited for species composition data because of the way they are distributed (McCune & Grace 2002). Other statistical tests were performed using S-PLUS 6.2 for Windows.

Pre-treatment data, 2003

All plots sampled in 2003 were as yet untreated. Data from these plots were analyzed to determine if there were pre-treatment differences in species composition between groups of plots. Of the original 166 taxa in the raw dataset, 28 species that occurred in only one plot out of 26 were deleted. Deleting these rare species reduced noise in the data and strengthened overall species composition patterns. Outlier analysis using the average Sørenson distance between each plot or species and others, showed that one plot varied over 2.5 standard deviations from the average distance across plots. The plot was downslope from an irrigation ditch, which may have accounted for its large complement of introduced weeds that were not found in any other plots. Because of its unusual character this plot and its paired match were removed from the analysis. Additionally, these two plots were not resampled in 2004 because the scheduled treatment had not been performed. One additional species that occurred only in these two plots was excluded by their removal, leaving 139 taxa in 24 plots.

Multi-response permutation procedure (MRPP) was used to test differences between groups. MRPP is a nonparametric method that works by reassignment to groups using an average of distance between each pair of group members (Zimmerman et al. 1985; McCune & Grace 2002). The procedure provides A-values that represent chance-corrected within-group agreement. A = 1 indicates complete agreement where all items are identical within groups, and A = 0 when heterogeneity within groups is the same as expected by chance (McCune & Mefford 1999). I tested the two groups formed by treatment assignment (mastication and their controls versus HPB and controls) and management units for difference in species composition using Sørenson distance.

In addition, a blocked MRPP (MRBP) was used to test for differences between future treatment plots versus control plots by pair. Blocking focused the analysis on differences within the pair, accounting for variation among pairs. Euclidean distance was used since Sørenson distance is not an option for MRBP in PC-ORD. Because MRBP required a balanced design, two plots were removed from the data for this test. These were a pair that consisted of an HPB treatment opposite a mastication treatment without any control plot.

Ceanothus *regeneration, 2003*

In 2003, almost all plots contained mature *Ceanothus* shrubs, while less than half of the plots had small plants. The data on *Ceanothus* included species abundance codes and an indicator for whether small *Ceanothus* (those under 0.3 m tall) were present or absent. MRPP (with Sørenson distance) was used to test for differences in species composition between plots with small *Ceanothus* present versus absent. Abundance of *Ceanothus* was removed from the species composition matrix for this test, since the presence of small *Ceanothus* was more likely with higher abundance of the shrub. Indicator species analysis (ISA; Dufrêne & Legendre 1997) was used to further characterize the two groups defined by current regeneration of *Ceanothus*. ISA produces indicator values that combine relative abundance and relative frequency of each species by group. A Monte Carlo test of 1000 randomizations was used to test the significance of species indicator values in all uses of ISA.

Changes between 2003 and 2004

Eighteen plots were sampled in both 2003 and 2004, before treatment and after treatment had occurred, for a total of 36 samples. With these data, pre-treatment species composition was compared to post-treatment and controls were compared between years. I started with 177 taxa and deleted the rare species that occurred in only 1 or 2 plots, resulting in 136 remaining taxa. In addition I removed woody species from the dataset, since woody plants were the target for removal with fuel treatment, leaving 131 taxa. MRPP was used on this dataset to test for differences in species composition between 2003 versus 2004, and between groups defined by treatment status (post-treatment plots versus pre-treatment and controls).

To better assess the changes between 2003 and 2004 in controls and in plots after treatment, the differences in species composition between the two years were calculated. Returning to the dataset with 177 taxa in 36 samples, 2003 abundance codes were subtracted from 2004 values, resulting in a difference matrix of 159 taxa in 18 plots. Eighteen taxa dropped out because they had the same values in both years. They were equally distributed between treatment and control plots. No rare species were deleted from the difference matrix because it had a fairly normal distribution of data. Woody species were again deleted, leaving 155 species.

This matrix of changes in composition contained negative numbers, requiring use of a distance measure other than Sørenson. MRPP on Relative Euclidean distance was used to test for differences among management units in the plots' year-to-year changes. After deleting the two-treatment pair to achieve a balanced design, MRBP on Euclidean distance was used to assess differences between the changes in treatment plots and those of control plots, blocked by pair.

Traits of life form (forb, graminoid, shrub, tree, annual, or perennial), as well as native or exotic, were assigned to each species where possible. Additionally, weedy plants (characteristic of disturbed habitats) were identified as such and categorized as either native weeds or introduced weeds. Traits were determined from two sources, primarily the Jepson manual (Hickman 1993), with the Plants database as a backup source of information (USDA NRCS 2004). Special status plants were also indicated, according to the Medford District BLM's current list. Species trait assignments are provided in Appendix 1. Multiplying a species trait matrix by the change in species composition matrix created a change in species trait by sample unit matrix. Shrubs and trees were included in the species trait by sample unit matrix, although they were removed from the species composition data in statistical analyses. In addition, change in species abundance by plot was converted to an indicator for increase (1) or decrease (-1) to calculate a change of species trait-by-sample unit matrix with the number of species as the unit of measure.

Changes in abundances of trait groups were examined for general trends and compared between treatments and controls with two-sample t-tests. T-tests were chosen because the variation approximated a normal distribution. For these tests, and those used in further analyses, values for significance should be treated with caution because samples were not independent; plots were spatially correlated. Because multiple comparisons were made, probabilities of significance (p-values) should be assessed conservatively.

An ordination of the change in species composition between the two years was accomplished using non-metric multidimensional scaling (NMS). NMS produces best-fit ordinations by trial and error using multiple iterations (Kruskal 1964; McCune & Grace 2002). Relative Euclidean distance, random starting configuration, and 40 runs were used on the matrix of change in 155 species abundances in 18 plots. The three-dimensional solution recommended by NMS autopilot was accepted. The final solution used 182 iterations, for a final stress of 12.9 (final instability = 0.00001). One hundred randomized data runs gave a minimum stress of 14.0, demonstrating that the solution was stronger than expected by chance ($p = 0.010$).

Overlays included date of sampling (in days since May 1), estimated elevation (taken from a 6-m-interval contour map), slope (estimated using the contour map and measurements taken on site), heat load index calculated from aspect (Neitlich &

McCune 1997), maximum litter depth, canopy cover, total basal area of trees within the plots, and cover class estimates of bare soil, litter, woody debris, standing dead, and rocks. Some variables were static such as elevation and slope, while others changed from 2003 to 2004. Values for 2003 were subtracted from 2004 values to calculate change in canopy cover; day of reading; maximum litter depth; cover of bare soil, litter, woody debris, standing dead, and rocks; and abundance of small *Ceanothus*. I also included 2004 values of the same variables to assess whether they were related to the change in species composition.

Environmental variables were tested for differences among management units using one-way Analysis of Variance (ANOVA) and between treatment and control plots using two-sample t-tests.

Post-treatment data, 2004

Data from 2004 provided the largest, most comprehensive set for analysis, including four categories of post-treatment plots, varied by treatment type and time since treatment, along with paired controls. Two hundred twenty-eight taxa were seen in 2004 in 52 plots. Number of species present in each plot was included as a plot attribute. The average species richness per plot (alpha diversity) was 71.7, and beta diversity (the total number of species in all plots / *alpha*) was 3.18. After recording species richness, the 59 species that occurred in fewer than four plots were deleted, leaving 169 species. All woody plants were deleted from the dataset resulting in 161 taxa. Outlier analysis showed that one plot varied almost 3 standard deviations from the average Sørenson distance among plots. However, deleting the plot caused other outliers to increase above 3 standard deviations. Analysis was tried with and without deleting one or more of these plots, and because the results were similar, all plots were retained.

MRPP was used to test whether 2004 species composition differed between plots established in 2003 versus those established in 2004. MRPP was also used to test for compositional differences between plots grouped by management units and by township-range-section. Sørenson distance was used for all analyses with this dataset, except for the following MRBPs. MRBP was used to compare species composition between treatment and control groups, blocking by pair to account for variation among pairs. Because MRBP requires a balanced design, four plots were removed for this test. These included a pair that consisted of an HPB treatment opposite a mastication treatment, a plot that was paired with an already existing control of another pair, and a plot that had been judged unrepresentative in the field and had been replaced in its pair by the addition of a new plot. MRBP was used to test for differences between treatment and controls in the following groups using Euclidean distance: all matched pairs, HPB treatments, mastication treatments, plots treated within the past year, plots treated 2 years ago, and the four subgroups made up of each treatment type by each treatment year.

Ordination of the sample units in species space was accomplished using NMS. PC-ORD's autopilot mode was used on 161 species in 52 plots with Sørenson distance, random starting configuration, and 40 runs on the data (McCune & Grace 2002). The three-dimensional solution recommended was accepted. The final solution used 400 iterations, for a final stress of 12.1 (final instability = 0.00036). A randomization test of 50 runs yielded a minimum stress of 23.4, which demonstrated the strength of the solution ($p = 0.020$). Overlays of environmental variables and species traits were used to assess whether each attribute varied with major gradients in species composition. Abundances of shrubs and trees by plot were included as overlays even though their species abundances were not used in producing the ordination. In addition, the ratio of the abundance of natives to that of exotics was included as a plot attribute. The soil type for each plot was determined using soil survey maps (NRCS 2005). All of my plots occurred in four different mapped units, which were included as an environmental variable. Matched treatment and control plots always occurred in the same mapped soil unit.

Differences between treatments and controls, 2004

To help clarify any differences between treatment and control plots in 2004, I returned to the full 228 taxa dataset to create a new matrix. For each matched pair of treatment and control, the species abundance values for the control were subtracted from those for the treatment. The resulting difference matrix represented the treatment effect on each pair. Seven species dropped out because they were in unmatched plots (four species) or there was no difference between any treatment and control pairs (three species, evenly distributed by type and age of treatment).

A change in species trait by pair matrix was created by multiplying the difference matrix by a transposed trait matrix. In addition, change in species abundance by pair was converted to 1 or -1 (increase or decrease) for each species to calculate a change of species trait by sample unit matrix with the number of species changed as the unit of measure. To provide a basis of comparison for how many species of each trait had changed, the full 228 taxa dataset was used to make a presence-absence matrix of control plots and calculate the number of species per trait in the control plots. Percent of species changed by trait was calculated using the average number of species different between treatment and control for a given trait divided by the average number of species with that trait in the control plots.

Differences in species traits by pair were tested individually for differences from zero using one-sample t-tests. Two-sample t-tests tested whether changes in traits differed between the types of treatment and between time-since-treatment groups. Changes in species richness and average canopy cover were also tested for differences between these groups.

Returning to the matrix of treatment - control differences, I deleted all woody taxa, resulting in 208 taxa for 25 pairs of plots. Evenly distributed data in this matrix made the deletion of rare species, to reduce skewness or the coefficient of variation, unnecessary. The matrix contained negative numbers, so Relative Euclidean distances were used in the multivariate analyses. MRPP tested for differences in the treatment effect on species composition between the following groups: plots established in 2003 versus 2004, time since treatment (1 yr or 2), and mastication versus Hand Pile & Burn. In addition, I used MRPP to test whether geographic sections differed in species composition changes, after the five sections with only one pair were deleted.

An NMS ordination of species composition differences between paired treatments and controls used settings similar to the autopilot mode. Relative Euclidean distance, random starting configuration, and 40 runs were used on the difference in 208 species in 25 pairs. The three-dimensional solution recommended was accepted. The final solution used 186 iterations, for a final stress of 19.8 (final instability = 0.00001). One hundred runs on randomized data gave a minimum stress of 20.1, demonstrating the strength of the solution (p = 0.010). Associated environmental variables were derived by combining the information for the matched treatment and control plots. The average value for the two plots in each pair was used for date of reading, elevation, slope, and heat load. The difference between treatment and control was calculated for species richness, non-plant cover, and abundance of mature and seedling *Ceanothus*. Both the average value and difference were used for canopy cover and basal area. The environmental variables together with species trait differences comprised an overlay matrix.

Overstory relationships to community composition

To describe visible differences in community composition between areas with tree canopy present and those without, I divided plots into two groups distinguished by a canopy cover of more or less than 10%, defined as 500 cm of canopy overlapping the plot edge. Using the 2004 dataset, minus rare species, I deleted species that were or could be present in the tree canopy, retaining shrubs. One hundred sixty-five species remained. MRPP tested for differences between the two groups. ISA further characterized any differences.

Intending to assess whether previously detected differences in species composition between and among groups of plots could be attributed to canopy influence, I tested these groups for differences in the amount of canopy cover. Since amount of canopy varied widely from 0 to 2790 cm, I converted these values to cover classes with the same coding that was used for abiotic ground cover (0 = none; 1 = <10% cover; 2 = 10-25% cover; 3 = 25-50% cover; 4 = >50% cover), so that groups would share similar variances for data analysis. Using two-sample t-tests, I tested for differences in canopy cover between plots established in 2003 versus 2004, and plots associated with management units that were treated 1 yr ago versus 2 (including both treatment and control plots). I also tested for differences among plots grouped by management unit and by geographic section using one-way ANOVA.

Interested in how the abundance of large *Ceanothus* plants affected the surrounding vegetation, I focused on the control plots in the 2004 dataset. With shrub cover recently removed, current abundance of *Ceanothus* in treatment plots would not have accurately reflected long-term influences of shrub shading and competition on other species. Therefore I removed all the treatment plots from the 2004 dataset of 52 plots (228 species). In the remaining 24 plots, I removed rare species that occurred in only 1 plot. I also deleted *Ceanothus,* leaving 161 species. MRPP was used to test for differences in species composition between control plots with more than ten mature *Ceanothus* plants (> 0.3 m tall) versus those with ten or fewer (11 plots). ISA was used to further characterize the two groups defined by abundance of mature *Ceanothus*. Using the same control plots, I tested the relationship between canopy cover and the abundance of mature *Ceanothus* or by calculating the correlation between the two variables.

Effect of treatment on Ceanothus

In 2004, I took detailed data on size classes of *Ceanothus*. I distinguished first year seedlings from small but woody plants ranging from the size of the seedlings to 0.3 m tall. I also recorded stumps that showed resprout growth separately from mature shrubs. All four stages were also combined to provide overall abundance of the species. Of the 52 plots used in the 2004 analysis, the single plot without any *Ceanothus* was removed from the *Ceanothus* age class dataset because it was a strong outlier (4.9 standard deviations away from the average of plots using Sørenson distance). MRBP blocked by pair was used to test for differences in *Ceanothus* stage abundance between treatments and controls, after deleting 5 additional unmatched plots. The 27 treatment plots were tested with MRPP for differences in stage abundances between the two types of treatments and between years since treatment. I tested for differences in abundance of each individual stage between treatment and control using a two-sample t-test.

I also calculated within-pair differences in abundance of each *Ceanothus* stage by subtracting values for controls from treatments for 25 pairs. Differences in stage abundances were compared between years since treatment and between types of treatment using two-sample t-tests.

Fire rings of HPB treatment

I compared the species composition of fire rings in the first year and the second year after burning. In addition, the proportions of species by trait found in the fire rings were compared to those of their associated treatment plots. Forty-two fire rings were sampled in June or July 2004, 3 for each of 14 HPB treatment plots. All had been burned in the months of November or December, half of them in 2003, the other half in 2002. Eighty-nine species were recorded in fire rings, and were compared to the 184 species found in the associated treatment plots from the 2004 dataset. All species were retained to better represent species richness. Species lists were compared between fire rings, HPB treatments, and HPB controls (2004), to determine whether any species were associated with burning.

MRPP on Sørenson distance was used to compare species composition between fire rings grouped by age. An ISA was done to help differentiate the two ages of fire ring. In addition, I modified the species abundance matrix to indicate only presence or absence of each species. I used presence-absence because I wanted to compare results with the much larger HPB treatment plots, in which the same abundance codes would represent different frequencies. This presence-absence matrix was multiplied by a species trait matrix, which resulted in a count of species present by trait for each fire ring. I tested for differences in numbers of species by trait between years using two-sample t-tests. I also calculated the mean number of species per trait per fire ring plot. The same analyses were performed on data from the associated HPB treatment plots.

RESULTS

Pre-treatment data, 2003

Species composition did not differ between pre-treatment and control plots when blocked by pair (MRBP: A = 0.007, p = 0.175). There was a small but statistically significant difference in species composition between the pairs assigned to mastication treatments versus HPB treatments (MRPP: A = 0.042, p = 0.005), and a stronger difference among management units (MRPP: A = 0.273, p < 0.001).

Ceanothus *regeneration, 2003*

Species composition differed between plots that had small *Ceanothus* present in 2003 and plots that did not (MRPP: A = 0.023, p = 0.043). Eight species were significant indicators of plots with or without *Ceanothus* regeneration (Table 2). All species indicating absence of *Ceanothus* regeneration are considered weedy, whereas only one of the presence indicators, *Pectocarya pusilla* (little combseed), is characteristic of disturbed habitats.

Table 2. Indicator values for species (percentage of perfect indication) for the two groups distinguished by *Ceanothus* regeneration in 2003. Only statistically significant indicator species are shown (p < 0.05). Boldface indicates non-native species.

Small *Ceanothus* present	IV	Small *Ceanothus* absent	IV
Daucus pusillus	55.7	***Cynosurus echinatus***	53.9
Eriophyllum lanatum	60.0	***Poa bulbosa***	60.3
Pectocarya pusilla	59.6	*Trifolium willdenowii*	51.9
Potentilla glandulosa	61.1		
Prunus subcordata	45.5		

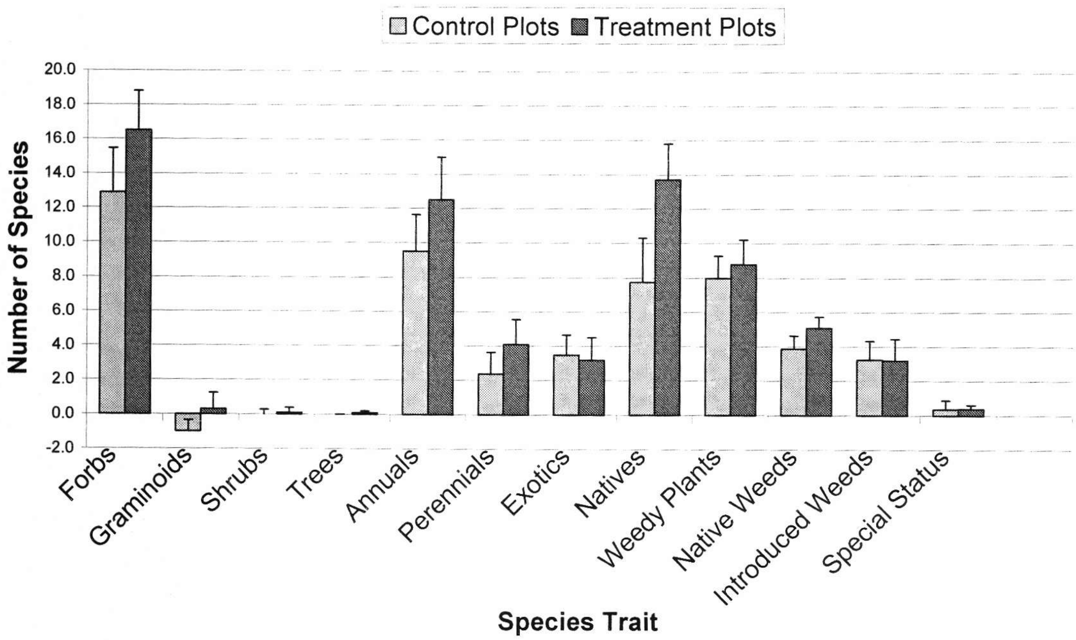
Changes between 2003 and 2004

Species composition differed significantly between 2003 and 2004 for plots that were sampled in both years (MRPP: A = 0.020, p = 0.024). Differences in species composition among management units were pronounced (MRPP: A = 0.215, p < 0.001), as they were in pre-treatment results. A suggestion of difference was indicated between post-treatment plots and the combined untreated plots (pre-treatment and controls; MRPP: A = 0.013, p = 0.078).

The differences between 2004 and 2003 readings, for plots sampled in both years, should reflect treatment effects as well as background changes from year to year. Overall there was greater species abundance in 2004 compared to 2003.

Species abundance increased in all trait groups except for shrubs and trees in control plots, which did not change, and graminoids in control plots, which decreased between 2003 and 2004. Number of species per trait that showed a change in abundance followed the same trends, illustrated in Figure 1. None of the trait group differences between treatment and control were significant (all $p > 0.05$, two-sample t-test).

Figure 1. Average number of species that differ between years in abundance per plot by trait (2004 minus 2003), with standard error.

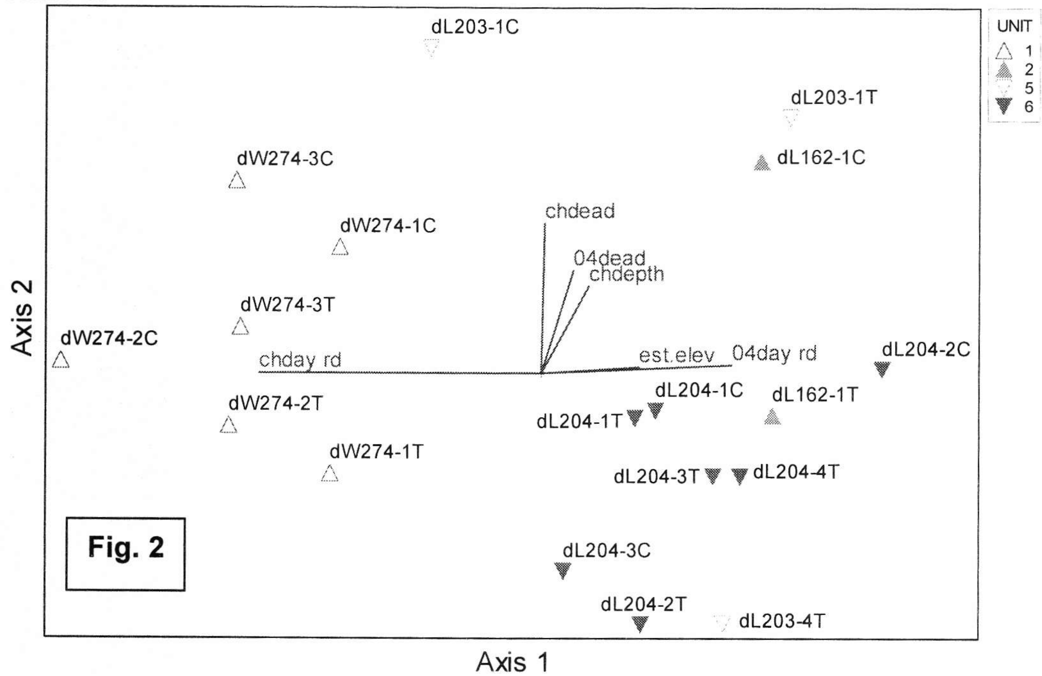


Year-to-year species composition changes in treatment plots did not differ from those in control plots (MRBP: A = -0.00005, p = 0.500). As usual, there was a significant difference in community composition among management units (MRPP: A = 0.058, p < 0.001).

The NMS ordination of the species composition changes between 2003 and 2004 (Figures 2-4) was rotated 40° on axes 1 and 3, to align Axis 1 with decreasing difference in date of sampling, maximizing variance along the strongest environmental gradient. The first axis represented 51.7% of the variance, the second axis 8.9%, the third axis 13.0%, with a cumulative variance explained of 73.6%.

Simple linear correlations between various attributes and the ordination scores are provided in Table 3.

Figures 2 & 3. NMS ordination of change in species composition for plots sampled in both 2003 and 2004. Plot symbols are coded by management unit. Labels ending with "C" represent control plots, while those ending with "T" represent treatment plots. Joint plots, rotated to maximize relationship with date of sampling, show significantly correlated environmental variables ($r^2 > 0.20$). Acronyms for variables are given in Table 3.



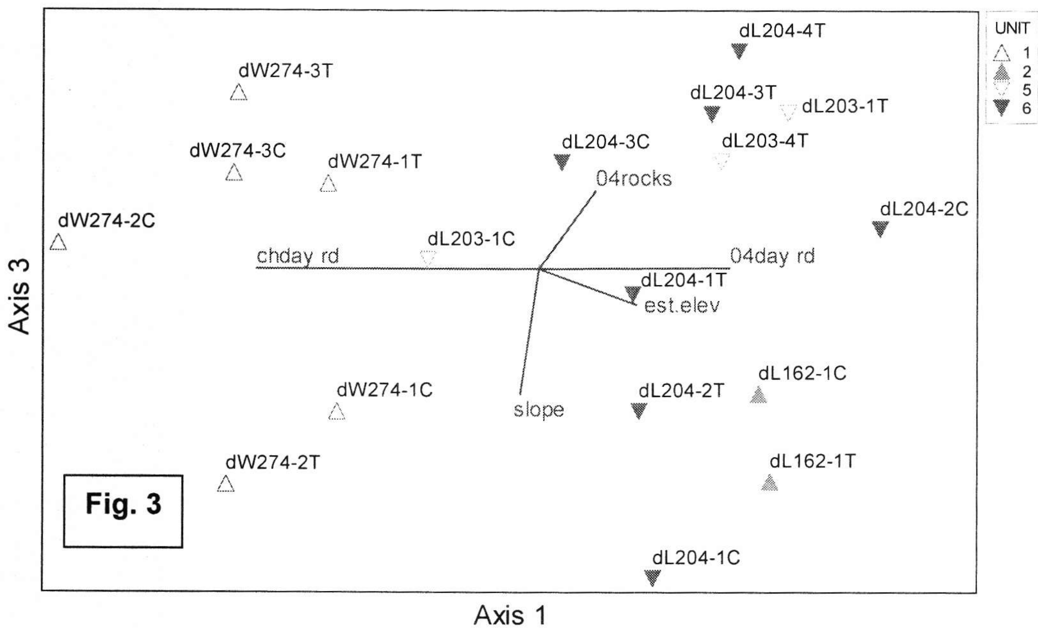


Figure 4. NMS ordination of change in species composition for plots sampled in both 2003 and 2004. Plot symbols are coded by management unit. Labels ending with "C" represent control plots, while those ending with "T" represent treatment plots. Joint plots, rotated to maximize relationship with date of sampling, show significantly correlated environmental variables ($r^2 > 0.20$). Acronyms for variables are given in Table 3.

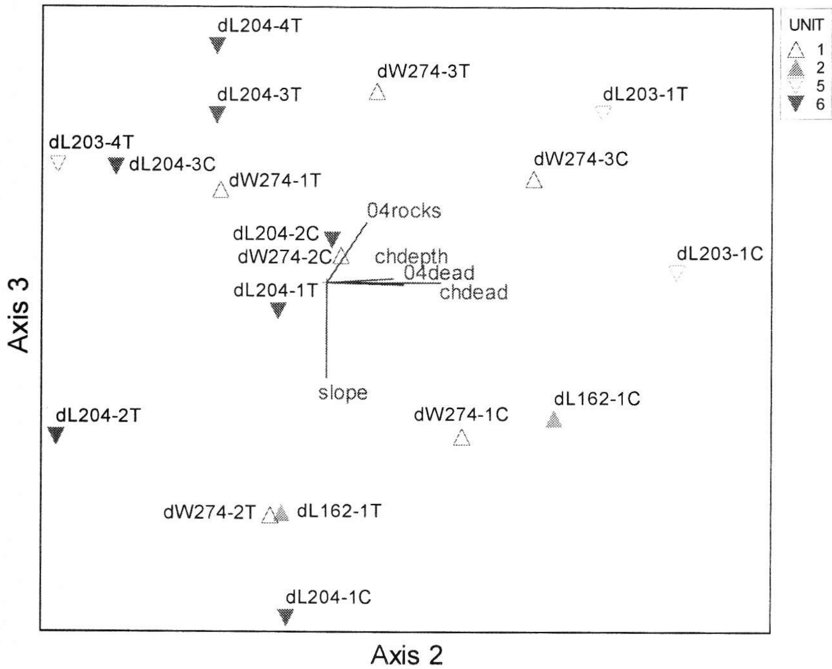


Table 3. Correlations of environmental variables with axes for ordination of changes between 2003 and 2004 in plots with pre- and post-treatment data, rotated to align with strongest environmental gradient. Acronyms listed are labels on joint plot (Figures 2-4).

		Correlation (r)		
Acronym	**Attribute**	**Axis 1**	**Axis 2**	**Axis 3**
est.elev	Estimated elevation	0.489	0.119	-0.297
slope	Slope	-0.212	-0.013	-0.554
chdepth	Change in maximum litter depth	0.339	0.461	0.116
chday rd	Change in date sampled	-0.831	-0.003	0.028
04day rd	2004 date sampled	0.682	0.139	0.049
chdead	Change in standing dead *Ceanothus*	0.080	0.605	0.018
04dead	2004 standing dead *Ceanothus*	0.281	0.498	-0.074
04rocks	2004 rock cover	0.373	0.359	0.439

There was a wide range of values for change in date of sampling. I sampled the same plots between 52 and 83 days earlier in 2004 than in 2003. Plots were sampled in the same order in both years, but the speed of sampling increased dramatically since the plots were already in place, and some untreated units were skipped in 2004. I finished the same plots 31 days more quickly than I had the year before. The fact that elevation is positively correlated with Axis 1 while change in date sampled is strongly negatively correlated on the same axis, suggests that much of the difference in species composition between the two years is due to phenology. Management units tend to separate along Axis 1 as well (Figures 2 & 3), in part due to differences in date of sampling. Change in sample date differed significantly by unit ($p = 0.004$, one-way ANOVA), as did rock cover recorded in 2004 ($p = 0.010$, one-way ANOVA) and slope ($p = 0.023$, one-way ANOVA). Axis 3 is most strongly related to steepness of slope, which is negatively correlated with rock cover (Figures 3 & 4).

Axis 2 correlates with the change in standing dead *Ceanothus* abundance between 2003 and 2004, which should change only with treatment or with recent mortality. Indeed, the change in standing dead differs significantly between treatment and control plots ($p = 0.009$, two-sample t-test). Change in standing dead is positively correlated with cover of 2004 standing dead, which also differs between treatment and control ($p = 0.035$, two-sample t-test), and change in maximum litter depth, which does not. In all cases but one, control plots appear on the ordination at a higher position along Axis 2 than their associated treatment plots (Figures 2 & 4).

Post-treatment data, 2004

The dataset that I used to most directly address the effect of treatment is that of species abundance from 2004. Using a single year's data largely eliminated the phenological effects described above. In 2004 I read plots that had been treated within the past year and their paired controls, and plots that had been treated in 2002 plus their controls. Table 4 summarizes the comparisons of community composition for treatment versus control plots, blocked by pair, within various treatment groups. Comparing all treatments to all controls (MRBP on 48 plots), there was a significant difference ($p < 0.05$) in species composition (Table 4). The species composition of the HPB-treated plots differed from their controls, whereas the mastication treatment plots and controls did not differ (Table 4). This indicates that the HPB treatment, in the short term, had a greater effect on species composition than mastication did.

When the treatment and controls were grouped by time since treatment, the 1-yr-since-treatment plots differed from their controls in species composition, while the 2-yr-since-treatment plots did not (Table 4). These results suggest that the treatment effect on community species composition was greatest immediately after treatment. When each treatment by time-since-treatment subgroup was tested individually, they conformed to their time-since-treatment outcomes rather than their treatment type results. That is, both treatment types differed between treatment and controls in their first year-since-treatment, while both did not differ in their second year-since-treatment (Table 4).

Table 4. Differences in community composition between treatment and control, 2004 plots, MRBP blocked by pair.

Group	Number of plots	Within-group agreement (A)	Probability (p)
All matched pairs	48	0.007	0.024
HPB treatment	24	0.012	0.034
Mastication treatment	24	0.004	0.234
1-yr-since-treatment	24	0.023	0.002
2-yr-since-treatment	24	0.005	0.172
1 yr since HPB treatment	12	0.030	0.035
2 yr since HPB treatment	12	0.009	0.254
1 yr since mastication	12	0.029	0.038
2 yr since mastication	12	0.002	0.428

Without regard to treatment or control, species composition in 2004 differed significantly between plots that were established in 2003 and those established in 2004 (MRPP: $A = 0.069$, $p < 0.001$). Groupings by both management units and geographic sections also differed in species composition (MRPP on unit: $A = 0.314$, $p < 0.001$; MRPP on section: $A = 0.265$, $p < 0.001$).

The NMS ordination for the 2004 species composition data (Figures 5-7) was rotated in three dimensions (50° on axes 1 and 2, and 25° on axes 1 and 3) to align Axis 1 with increasing abundance of perennials, maximizing variance along the strongest attribute gradient. After rotation the first axis represented 58.9% of the variance, the second axis 19.9%, the third axis 10.4% with a cumulative variance explained of 89.1%. Simple linear correlations between various attributes and the ordination scores are provided in Table 5.

Most of the interesting aspects of the ordination are visible in Figure 5. Axis 1 is most strongly associated with the abundance of perennials but it is also positively correlated with increasing tree canopy, depth and cover of litter, and species richness, as well as abundance of shrubs, trees, graminoids, and natives. The

Figure 5. NMS ordination of all plots sampled in 2004 in species space. Solid plots were established in 2003, and open plots were established in 2004. Joint plots, rotated to maximize relationship with abundance of perennials, show significantly correlated species traits and environmental variables ($r^2 > 0.20$). Acronyms for variables are given in Table 5. See also Figures 6 & 7 for alternative coding of plots on this ordination.

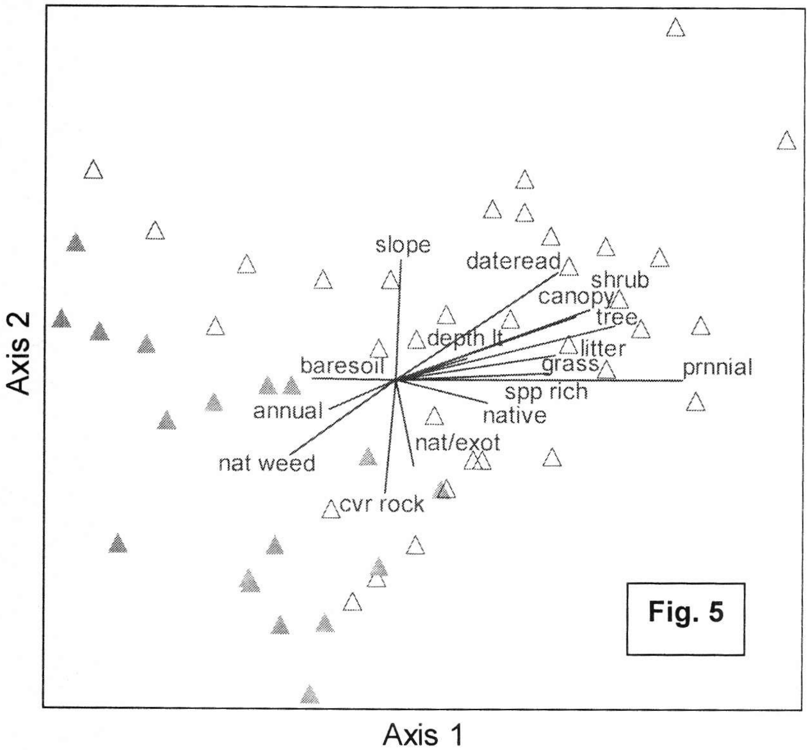


Table 5. Correlations of various attributes with ordination axes for plots in 2004, rotated to align with strongest attribute gradient. Acronyms listed are labels on joint plot (Figures 5-7).

		Correlation (r)		
Acronym	**Attribute**	**Axis 1**	**Axis 2**	**Axis 3**
spp rich	Species richness	0.542	-0.025	0.002
dateread	Date sampled	0.668	0.545	0.219
slope	Slope	0.113	0.577	0.264
baresoil	Bare soil cover	-0.480	0.062	-0.299
litter	Litter cover	0.663	0.267	0.060
depth lt	Maximum litter depth	0.444	0.243	0.313
cvr rock	Rock cover	-0.166	-0.559	0.003
canopy	Canopy	0.705	0.422	0.032
grass	Graminoids	0.648	0.143	0.180
shrub	Shrubs	0.732	0.442	-0.194
tree	Trees	0.777	0.390	0.128
annual	Annuals	-0.430	-0.285	-0.109
prnnial	Perennials	0.889	0.028	0.049
native	Natives	0.505	-0.251	-0.164
nat weed	Native weeds	-0.539	-0.456	-0.266
nat/exot	Ratio of natives to exotics	0.225	-0.486	-0.238

cover of bare soil and abundance of annuals and native weeds are negatively correlated with this axis. Axis 2 is primarily related to increasing slope and decreasing rock cover. The plots form distinct groups by establishment year (2003 or 2004), separating on a diagonal to Axes 1 & 2. The sampling date variable is aligned with the separation by plot establishment year because plots established in 2003 were resampled in 2004 before any new plots were established.

Figure 6 shows Axes 1 & 3 of the same ordination with the treatment and control plots coded by type of treatment. The lack of discernible pattern for these four groups was replicated in the other views of the ordination, not reproduced here. Figure 7 shows Axes 2 & 3 of the same ordination with the plots coded by soil type. There appears to be a relationship between species composition and soil type along both axes. The Carney clay soils, which have the slowest permeability, plot high on Axis 3. The other two types of soils show separation along Axis 2.

Figure 6. NMS ordination of all plots sampled in 2004 in species space (the same as in Figures 5 & 7). Solid triangles represent treatment plots. Open triangles represent control plots. Upright triangles represent mastication units. Point-down triangles represent HPB units. Joint plots, rotated to maximize relationship with abundance of perennials, show significantly correlated species traits and environmental variables ($r^2 > 0.20$). Acronyms for variables are given in Table 5.

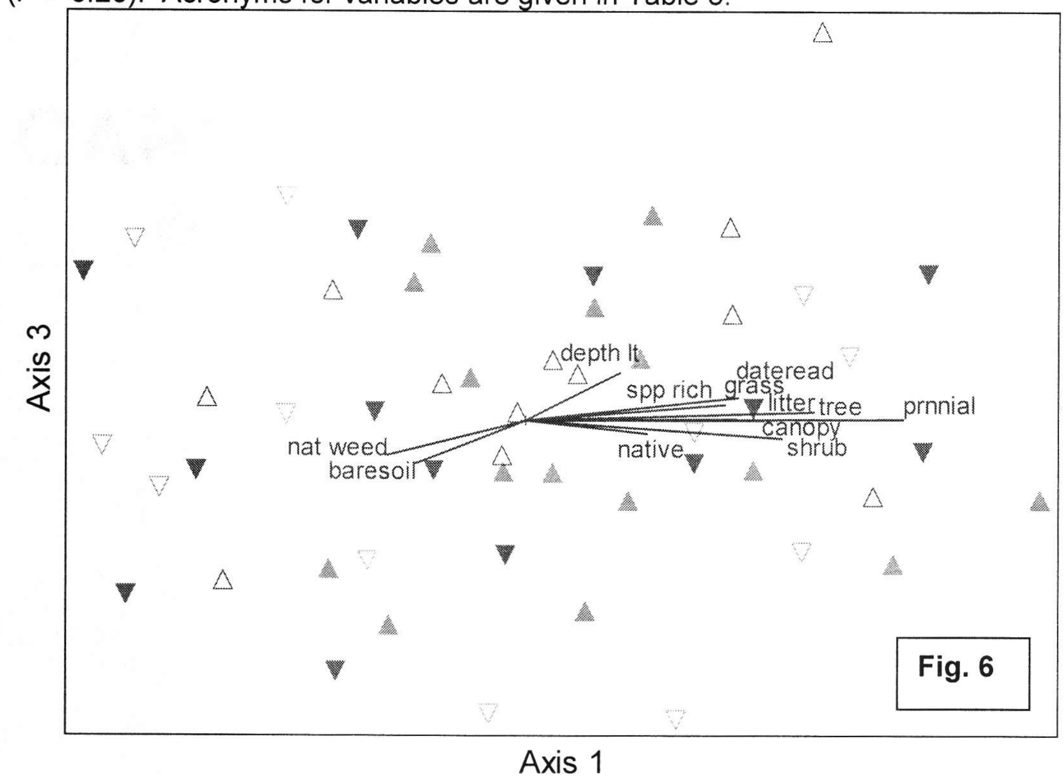
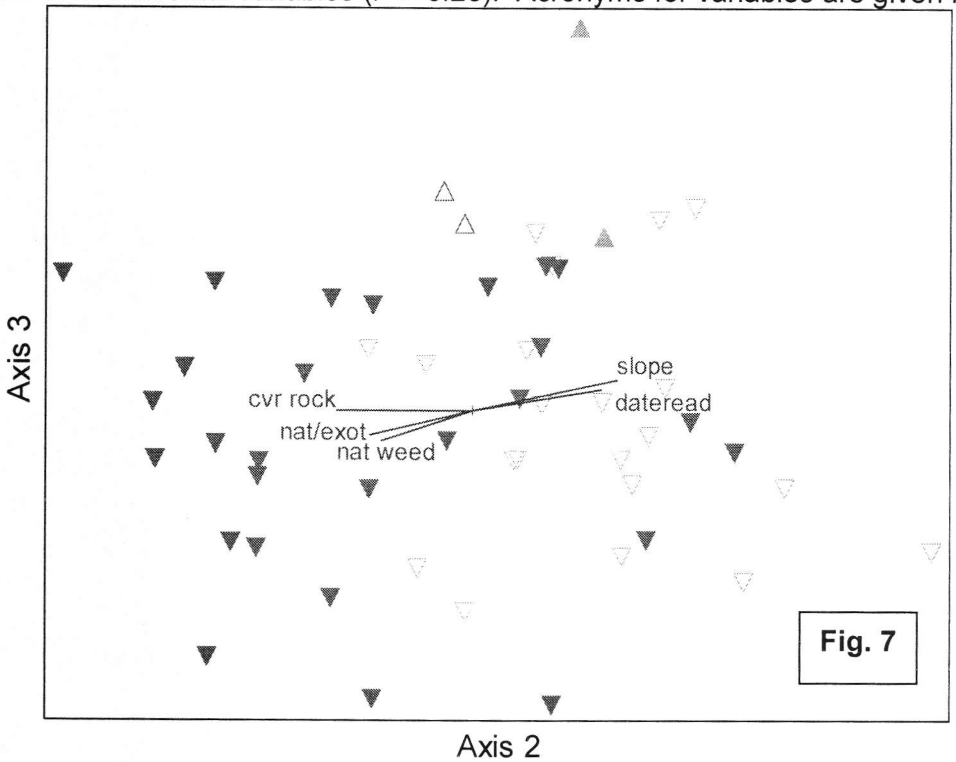


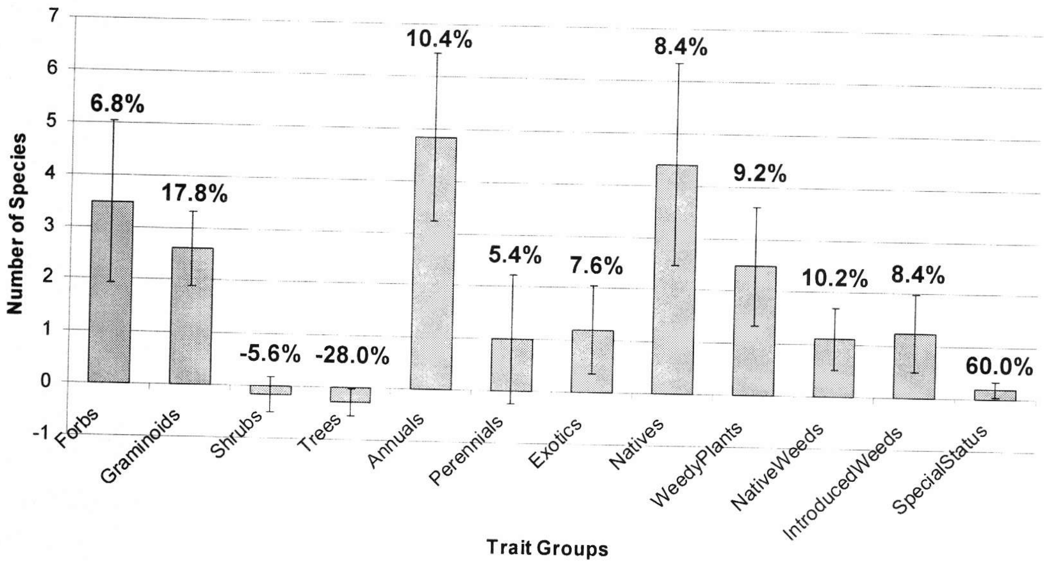
Figure 7. NMS ordination of all plots sampled in 2004 in species space (the same as in Figures 5 & 6) coded by soil type. Open upright triangles are plots on Carney clay. Solid upright triangles are plots on Carney cobbly clay. Open point-down triangles represent plots on McMullin-Rock outcrop complex. Solid point-down triangles represent plots on Medco-McMullin complex. Joint plots, rotated to maximize relationship with abundance of perennials, show significantly correlated species traits and environmental variables ($r^2 > 0.20$). Acronyms for variables are given in Table 5.



Differences between treatments and controls, 2004

The following trends were observed for differences between matched treatment and control plots in 2004 (calculated as treatment less control by pair). Species richness was greater by an average of 4.2 species in treatment plots than in control plots ($p = 0.039$, one-sample t-test; 95% confidence interval from 0.2 to 8.2 species). Overall there was greater species abundance in the treatment plots. The only trait groups that were less abundant in treated plots than in their controls were shrubs and trees. The following trait groups were significantly more abundant in treated than in control plots (mean of differences significantly greater than zero; $p < 0.05$ by one-sample t-test): forbs, graminoids, annuals, natives, and native weeds. The amount of change between treatments and controls is more comprehensible when trait groups were expressed as number of species that differ, using binary indicators for increase or decrease of abundance per species per plot rather than actual difference in abundance. The same patterns apply for average number of species that differ in abundance between treatment and control (Figure 8).

Figure 8. Differences in abundance between treatment and control plots by trait. Bars show average number of species that differ in abundance, with standard error. Numbers in bold show the difference in species as a percentage of the average number of species for that trait in control plots.



Trends in abiotic plot attributes included greater cover of bare soil in treatment plots, and significantly greater woody debris ($p = 0.001$, one-sample t-test; Figure 9). Maximum litter depth was the same in the treatment and control groups. Cover of litter and rocks decreased in treatments compared to controls (Figure 9). Tree basal area increased in treatment plots while tree canopy cover decreased. *Ceanothus* seedlings were more abundant in treatment plots than in control plots (Figure 10). Uncut mature *Ceanothus* were substantially less abundant (Figure 10) as was cover of standing dead *Ceanothus* in treated plots (both $p < 0.001$, one-sample t-test; Figure 9).

Comparing differences by treatment type, the HPB treatment had a larger increase in number of species and in species abundance than did the mastication treatment. Species abundance in all trait groups except trees increased in the HPB treated plots. In the mastication treatment, 4 out of 12 trait groups (shrubs, trees, perennials, and native weeds) decreased in species abundance in treated plots.

Figure 9. Mean change in substrate cover class units between treatments and controls (treatment minus controls), with standard error bars. Cover classes ranged from *0* for no cover to *3* for 25-50% cover.

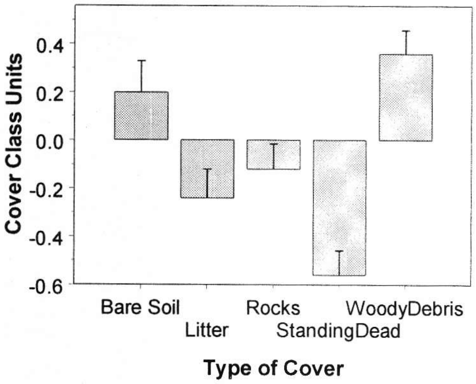
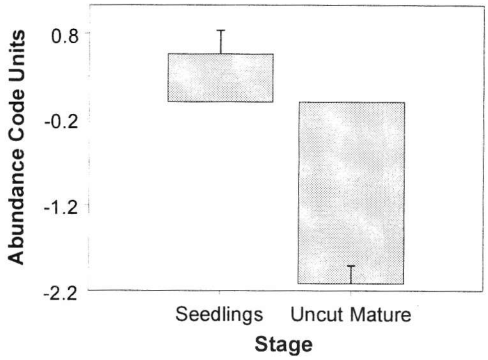


Figure 10. Mean change in abundance code units for *Ceanothus* between treatments and controls, with standard error bars. Abundance codes ranged from *0* for absent to *4* for >50% cover.



The HPB treatment had significantly greater difference in abundance of native weeds ($p = 0.013$, two-sample t-test) and shrubs ($p = 0.018$, two-sample t-test) between matched treatment and control plots, in comparison to the mastication treatment. The change in abundance of special status plants tended to be greater in the HPB treatment pairs than in the mastication treatments, though the difference was not significant ($p = 0.089$, two-sample t-test). These results are expressed in terms of number of species that show a difference in Table 6.

When differences were separated by time since treatment, the 1 yr since treatment pairs had a greater increase in species abundance than did the 2 yr since treatment pairs. Species abundance in all trait groups increased in the first year after treatment. In the second year since treatment, 4 out of 12 trait groups (shrubs, trees, perennials, and natives) decreased in species abundance. Those groups that differed significantly between ages ($p < 0.05$ by two-sample t-test) were graminoids, trees, perennials, and natives. These traits are expressed in terms of number of species that differed between treatment and control in Table 7.

Table 6. Average number of species that differ in abundance between matched treatment and control plots by trait (treatment minus control), with standard error. Statistical significance of differences between treatments is indicated by ** when p < 0.01, * when p < 0.05, by two-sample t-test. Differences are also given as a percentage of the average number of species by trait in control plots. See also Figures 11 & 12.

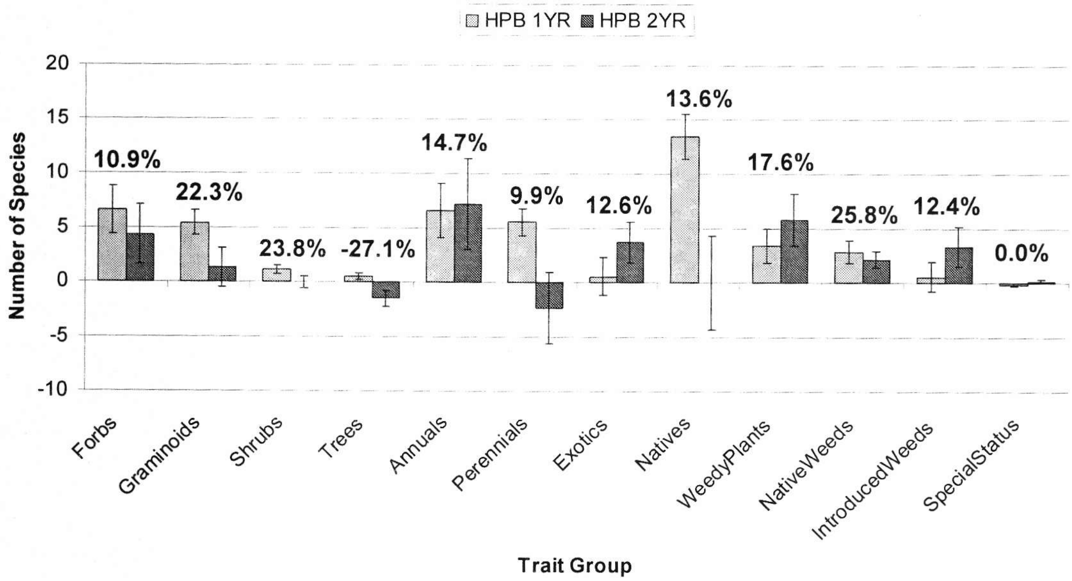
Traits	HPB mean	SE	% of control		Mastication mean	SE	% of control
Forbs	5.5	1.7	10.9		1.3	2.6	2.5
Graminoids	3.5	1.1	22.3		1.6	0.8	11.9
Shrubs	0.6*	0.3	23.8		-1.0*	0.5	-32.4
Trees	-0.4	0.5	-27.1		-0.2	0.2	-28.6
Annuals	6.8	2.2	14.7		2.7	2.2	5.7
Perennials	1.9	1.9	9.9		0.0	1.5	0.0
Exotics	2.0	1.3	12.6		0.3	1.0	2.1
Natives	7.2	2.9	13.6		1.3	2.3	2.6
Weedy Plants	4.5	1.4	17.6		0.3	1.7	0.9
Native Weeds	2.5**	0.6	25.8		-0.4**	0.8	-3.4
Introduced Weeds	1.8	1.1	12.4		0.6	0.9	4.0
Special Status	0.0	0.1	0.0		0.4	0.3	100.0

Table 7. Average number of species that differ in abundance between matched treatment and control plots by trait (treatment minus control), with standard error. Statistical significance of differences between time since treatment is indicated by ** when p < 0.01, * when p < 0.05, by two-sample t-test. Differences are also given as a percentage of the average number of species by trait in control plots. See also Figures 11 & 12.

Traits	1 YR (n =13) mean	SE	% of controls		2 YR (n =12) mean	SE	% of controls
Forbs	6.6*	1.6	13.6		0.1*	2.4	0.2
Graminoids	4.2*	0.9	36.5		0.8*	0.9	4.7
Shrubs	0.2	0.5	10.7		-0.6	0.5	-16.7
Trees	0.3*	0.2	184.6		-0.9*	0.4	-50.0
Annuals	7.2	1.5	15.2		2.3	2.8	5.1
Perennials	3.5*	1.2	28.1		-1.8*	1.9	-7.2
Exotics	0.8	1.0	5.7		1.6	1.4	9.5
Natives	10.3**	1.9	22.2		-2.0**	2.4	-3.4
Weedy Plants	3.3	1.0	12.3		1.6	2.1	5.9
Native Weeds	2.1	0.6	17.9		0.1	1.0	0.8
Introduced Weeds	0.8	0.8	6.0		1.7	1.3	10.8
Special Status	0.3	0.3	61.5		0.1	0.1	50.0

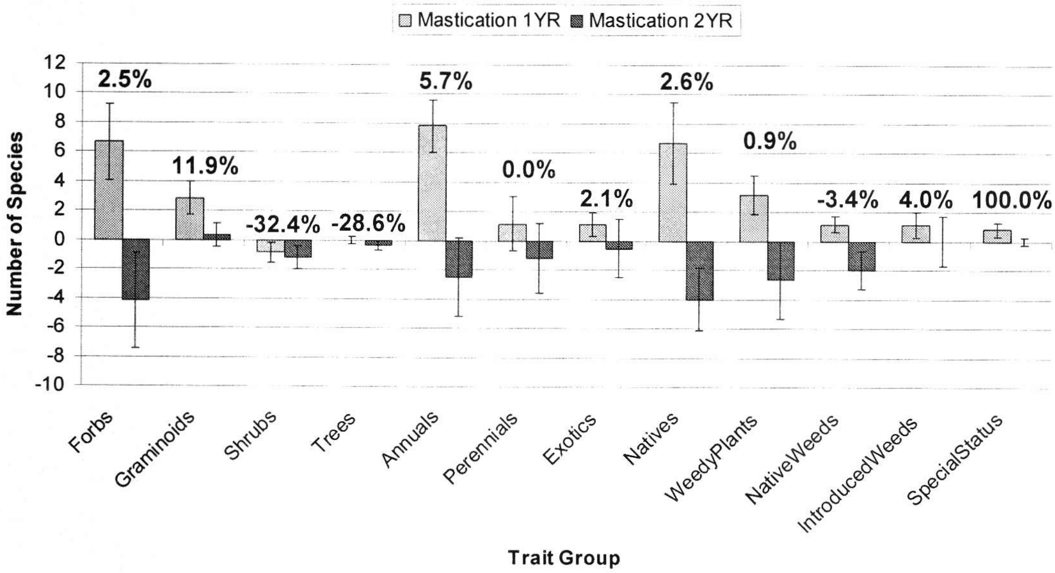
Difference in species richness between treatment and control differed between time-since-treatment groups (p = 0.004, two-sample t-test). The difference was positive in the 1 yr since treatment group, but negative in the 2 yr since treatment group. That is, there were fewer species in treated plots than control plots after 2 yrs. For first year plots, the average species richness was 74.1 in treated plots and 62.5 in the controls. In second year plots, it was 73.6 in treated plots and 76.0 for controls. The mean change in species richness was 10.5 species greater in the first year than in the second (95% confidence interval from 3.7 to 17.3 species). Dividing the two treatment types into their year-since-treatment groups, I found that species richness is more reduced in the second year of the mastication treatment than in the corresponding year of the HPB treatment. The HPB second year since treatment plots have greater species richness than their control plots. The same trend is illustrated in the number of species that show a change in species abundance (Figures 11 & 12).

Figure 11. Change in species traits for HPB pairs separated by years since treatment. Bars show average number of species that differ in abundance between treatment and control plots by trait, with standard error. Numbers in bold show the mean species difference across years-since-treatment as a percentage of the average number of species for that trait in HPB control plots (see Table 6).



Groups of pairs established in 2003 versus 2004 did not differ in overall species composition changes (MRPP: A = 0.003, p = 0.188), nor did treatment types (MRPP: A = 0.002, p = 0.211). There were, however significant differences in composition changes between pairs treated two years versus one year ago (MRPP: A = 0.008, p = 0.009), and between pairs grouped by geographic section (MRPP: A = 0.026, p = 0.033).

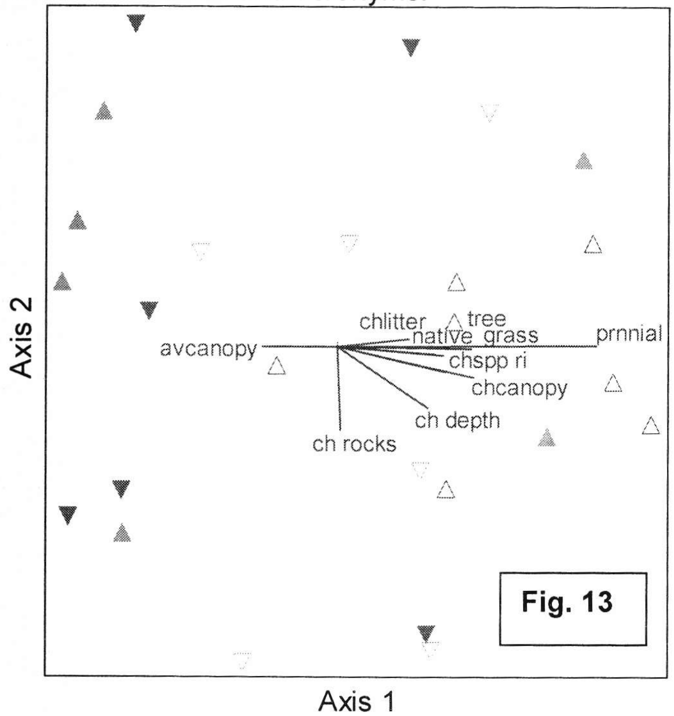
Figure 12. Change in species traits for mastication pairs separated by years since treatment. Bars show average number of species that differ in abundance between treatment and control plots by trait, with standard error. Numbers in bold show the mean species difference across years-since-treatment as a percentage of the average number of species for that trait in mastication control plots (see Table 6).



The NMS ordination of differences in 2004 species composition between treatments and controls (Figures 13-15) was rotated 160° on axes 1 and 2, and 60° on axes 1 and 3, to align Axis 1 with increasing difference in abundance of perennials, maximizing variance along the strongest attribute gradient. The first axis represented 18.1% of the variance, the second axis 14.1%, the third axis 15.0% with a cumulative variance explained of 47.1%. Simple linear correlations between various attributes and ordination scores are provided in Table 8.

In addition to differences in abundance of perennials, Axis 1 is positively correlated with increased differences between treatment and control in species richness; abundance of trees, grasses and natives; and overstory canopy cover. Axis 1 is negatively correlated with each pair's average canopy cover. These relationships between Axis 1 and canopy cover variables suggest that differences in trait group abundances are more related to tree canopy differences than to other environmental variables. Average canopy cover was significantly greater in both the 2 yr since treatment group versus 1 yr ($p = 0.004$, two-sample t-test) and in plots established in 2004 versus 2003 ($p = 0.011$, two-sample t-test). The year-since-treatment groups also tend to separate along Axis 1, with 2 yr-since-treatment plots associated with greater canopy cover.

Figures 13 & 14. NMS ordination of community differences between treatments and controls sampled in 2004. Plots are shown with open triangles if 1 year since treatment, and solid triangles if 2 years. Upright triangles represent HPB treatment while point-down triangles represent mastication. Joint plot, rotated to maximize relationship with differences in abundance of perennials, shows significantly correlated species traits and environmental variables ($r^2 > 0.20$). See Table 8 for ordination coefficients and acronyms.



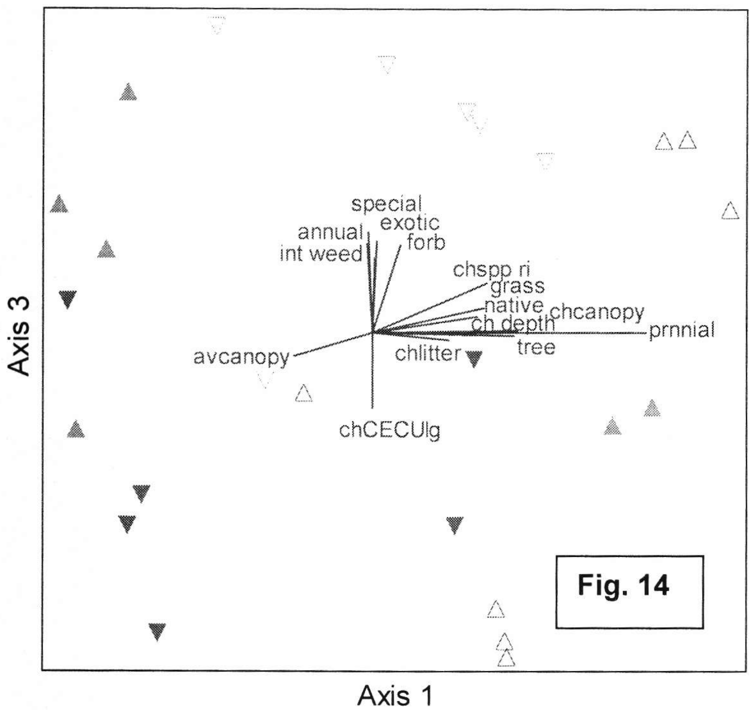


Figure 15. NMS ordination of community differences between treatments and controls sampled in 2004. Plots are shown with open triangles if 1 year since treatment, and solid triangles if 2 years. Upright triangles represent HPB treatment while point-down triangles represent mastication. Joint plot, rotated to maximize relationship with differences in abundance of perennials, shows significantly correlated species traits and environmental variables ($r^2 > 0.20$). See Table 8 for ordination coefficients and acronyms.

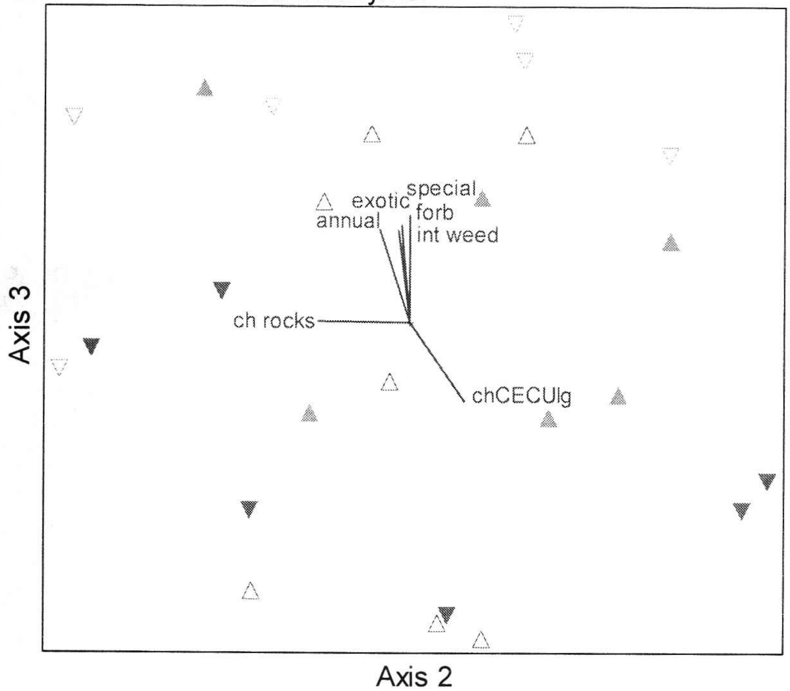


Table 8. Correlations of various attributes with ordination axes for the difference between paired treatment and control plots in 2004 (difference is treatment minus control), rotated to align with strongest attribute gradient. Acronyms listed are labels on joint plot (Figures 13-15).

		Correlation (r)		
Acronym	**Attribute**	**Axis 1**	**Axis 2**	**Axis 3**
chspp ri	Difference in species richness	0.586	-0.279	0.384
chlitter	Difference in litter cover	0.476	0.153	-0.147
ch depth	Difference in maximum litter depth	0.536	-0.439	0.056
ch rocks	Difference in rock cover	0.103	-0.510	0.062
chcanopy	Difference in canopy cover	0.658	-0.311	0.088
avcanopy	Average canopy cover	-0.484	-0.016	-0.264
chCECUlg	Difference in uncut *Ceanothus* abundance	-0.030	0.395	-0.473
forb	Difference in forbs	0.288	-0.184	0.510
grass	Difference in graminoids	0.579	-0.167	0.274
tree	Difference in trees	0.651	-0.082	-0.091
annual	Difference in annuals	-0.130	-0.292	0.512
prnnial	Difference in perennials	0.905	0.057	-0.006
exotic	Difference in exotics	0.117	-0.152	0.522
native	Difference in natives	0.560	-0.152	0.218
int weed	Difference in introduced weeds	0.080	-0.147	0.472
special	Difference in special status plants	-0.112	-0.003	0.547

The strongest correlation with Axis 2 is the difference in rockiness between treatment and control plots. Rockiness affects species composition, but not any particular trait group. Axis 3 is related to the difference in uncut mature *Ceanothus*. The more negative the difference between treatment and control, that is, the more *Ceanothus* removed by treatment, the more positive the difference is for annuals, forbs, exotics, introduced weeds, and special status plants. In Figure 14, year since treatment groups tend to separate along a diagonal to Axes 1 & 3, the same diagonal as difference in species richness, illustrating the differences in species richness that were indicated by two-sample t-test. No patterns are visible in the positions of HPB pairs to mastication pairs on any of the ordination graphs (Figures 13-15). In addition, there was no discernable pattern when pairs were coded by soil type (ordinations not shown).

Overstory relationships to community composition

Community composition differed significantly between plots that had more than 10% tree canopy (23 plots) and those that did not (29 plots; MRPP: $A = 0.069$, $p < 0.001$). Treated and control plots were about evenly divided in the two groups, with 11 controls and 12 treatments in the >10% canopy category. Fifty-two species were significant indicators for one of the two groups, 16 of these indicating plots with <10% canopy. The most significant indicators are in Table 9, while a full list, including all significant indicators, is given in Appendix 2. The two groups of species vary most in the ratio of annuals and perennials, with 6.3% of the species in the minimal canopy group being perennial, and 70.6% being perennial in the >10% canopy group. They also vary in the proportion of the species that are graminoids: 6.3% in the <10% canopy group and 19.4% in the >10% canopy group.

Plots that were established in 2003 differed in amount of canopy cover from those established in 2004 ($p < 0.001$, two-sample t-test). The mean difference in cover class was 1.24 cover class units (where 1 = <10% cover and 2 = 10-25% cover) greater in plots established in 2004 (Figure 16; 95% confidence interval: 0.60 - 1.89 units). Only 16.7% of the plots established in 2003 had >10% canopy, while 58.8% of the plots established in 2004 did. Similarly, only 18.5% of the plots associated with units treated in 2003 (1 yr since treatment) had >10% canopy. There is some overlap of these groups since the majority of plots treated in 2003 were also established in 2003. Fully 72% of the plots associated with units treated in 2002 (2 yrs since treatment) had >10% canopy. Plots belonging to units treated in 2002 had more canopy cover than plots of units treated in 2003 ($p < 0.001$, two-sample t-test). The mean difference in cover class was 1.34 units (Figure 17: 95% confidence interval: 0.76 – 1.93 units). Canopy cover also differed among management units (Figure 18; $p < 0.001$, one-way ANOVA) and geographic sections (Figure 19; $p < 0.001$, one-way ANOVA).

Table 9. Indicator values for species (percentage of perfect indication) for groups based on percent tree canopy in 2004. Only the most statistically significant indicator species are listed here ($p = 0.001$), the rest are included in Appendix 2. Boldface indicates non-native species.

Less than 10% tree canopy	IV	Greater than 10% tree canopy	IV
Clarkia gracilis	61.1	*Agoseris grandiflora*	60.9
Lomatium utriculatum	58.6	*Calochortus tolmiei*	67.7
Pectocarya pusilla	57.9	*Centaurium muehlenbergii*	51.6
Scleranthus annuus	60.2	*Euphorbia spathulata*	42.0
		Horkelia daucifolia	52.9
		Iris chrysophylla	39.1
		Ranunculus occidentalis	72.2
		Sanicula crassicaulis	54.4
		Torilis arvensis	67.4
		Bromus carinatus	56.5
		Elymus spp.	55.3
		Festuca idahoensis	64.5
		Koeleria macrantha	63.9
		Luzula comosa	75.2
		Lonicera interrupta	65.0

Figure 16. Average canopy cover class for plots established in 2003 versus 2004, with standard error. Cover classes ranged from *0* for no cover to *4* for >50% cover.

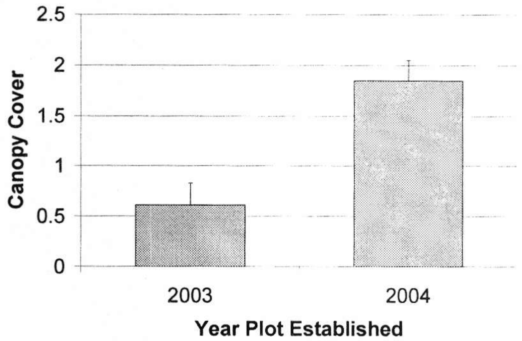


Figure 17. Average canopy cover class for plots associated with units treated 1 yr and 2 yrs previously, with standard error. Cover classes ranged from *0* for no cover to *4* for >50% cover.

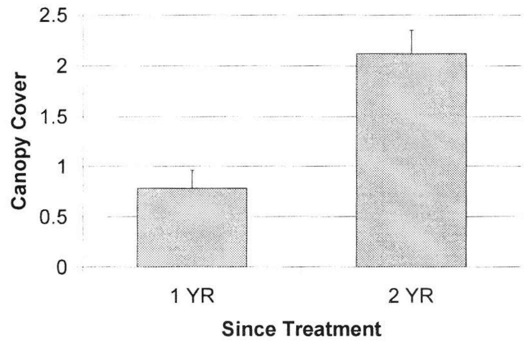


Figure 18. Average canopy cover class for plots by management unit, with standard error. Cover classes ranged from *0* for no cover to *4* for >50% cover.

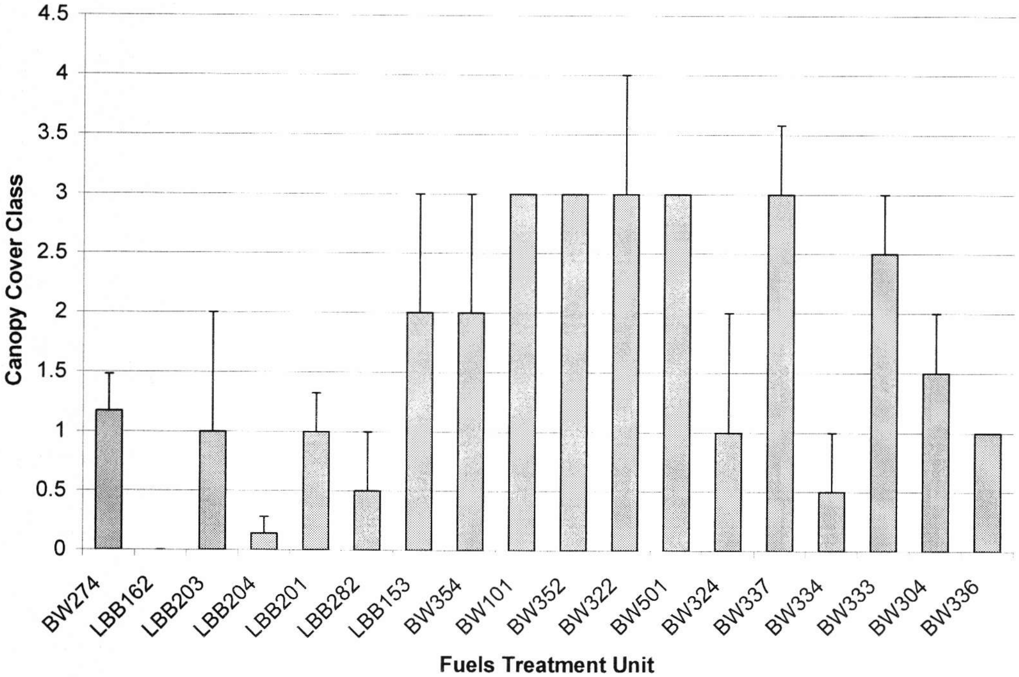
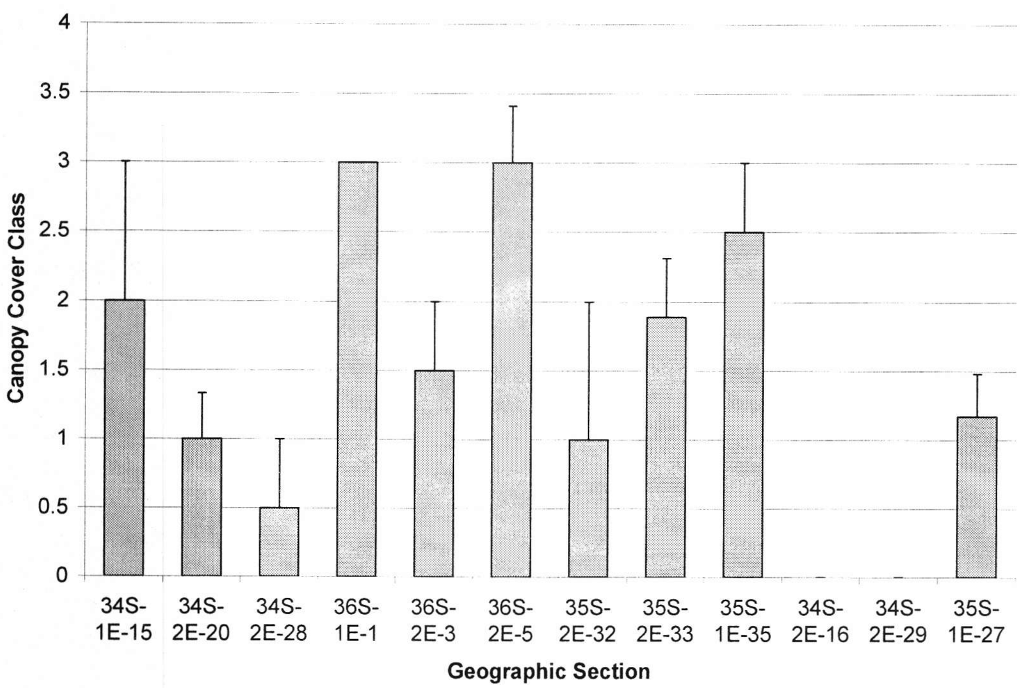


Figure 19. Average canopy cover class for plots by geographic section, with standard error. Cover classes ranged from *0* for no cover to *4* for >50% cover.



There was a direct inverse relationship between the cover class of tree canopy and the abundance of mature *Ceanothus* (r = -0.423). Species composition differed between control plots sampled in 2004 that had abundant mature *Ceanothus* (more than 10 plants > 0.3 m tall) and those with more sparse *Ceanothus* (MRPP: A = 0.021, p = 0.049). Fifteen taxa were significant indicators for one of the two groups of plots (Table 10). One species, *Lomatium utriculatum* (common lomatium), indicated both <10% canopy and >10 mature *Ceanothus* groups. Two species, *Iris chrysophylla* and *Cynosurus echinatus* (hedgehog dogtail grass) indicated both >10% canopy and fewer mature *Ceanothus* groups. Again, more overstory was associated with a higher proportion of perennial indicators; 75.0% of the abundant *Ceanothus* group and 33.3% of the sparse *Ceanothus* group were perennials. None of the abundant *Ceanothus* indicators were weedy, while more than half of the sparse *Ceanothus* indicators were, including the invasive weed *Centaurea solstitialis* (yellow star-thistle), the only plant found in any of my plots that occurs on the Medford District BLM's noxious weed survey list.

Table 10. Indicator values for species (percentage of perfect indication) for groups based on abundance of mature *Ceanothus* in 2004 samples of controls. Statistically significant indicator species only (p < 0.05). Boldface indicates non-native species.

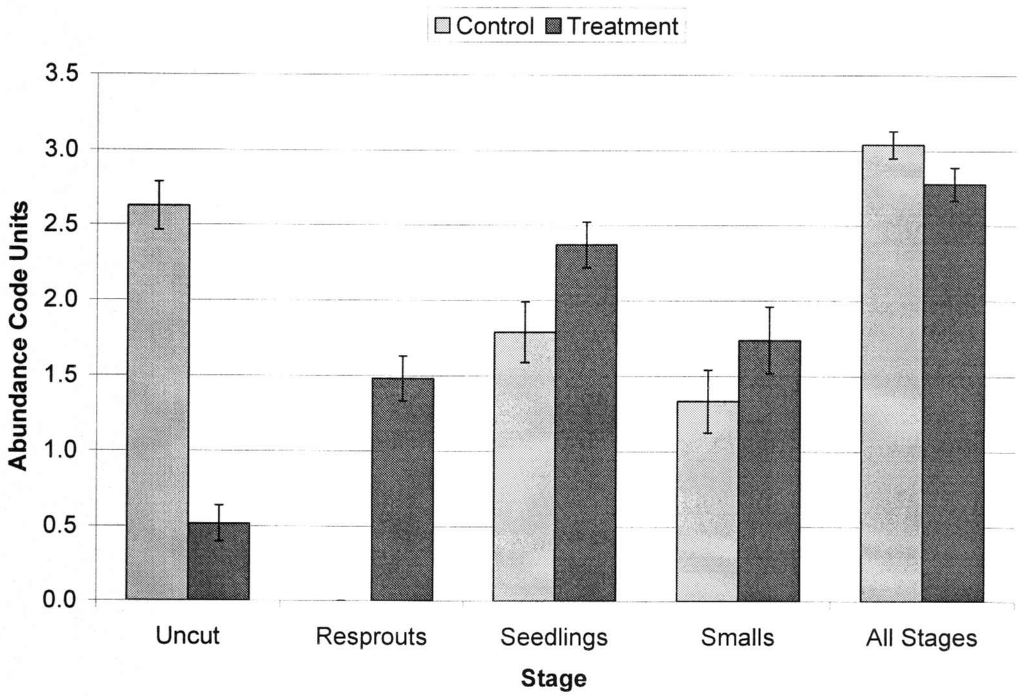
Greater than 10 mature *Ceanothus* per plot	IV	10 or fewer mature *Ceanothus* per plot	IV
Achnatherum lemmonii	60.2	*Arctostaphylos viscida*	36.4
Collinsia parviflora	38.5	***Bromus sterilis***	56.2
Fragaria vesca	49.8	***Centaurea solstitialis***	36.4
Githopsis specularioides	69.3	***Cynosurus echinatus***	67.2
Juncus spp.	53.0	***Hypochaeris glabra***	49.0
Lithophragma spp.	46.2	*Iris chrysophylla*	36.4
Lomatium utriculatum	61.7	*Plagiobothrys nothofulvus*	36.4
Potentilla glandulosa	59.4		

Effect of treatment **on Ceanothus**

As expected, the difference in *Ceanothus* stages in 2004 between treatments and controls was highly significant (MRBP: A = 0.349, p < 0.001). Comparing treated plots only, stage abundances differed between 1 yr and 2 yr since treatment (MRPP: A = 0.068, p = 0.004), but not between the two types of treatment (MRPP: A = 0.021, p = 0.129).

Average abundance of stages by treatment and control is shown in Figure 20. Seedlings were significantly more abundant in treatments than in controls ($p = 0.024$, two-sample t-test), as were resprouts, which did not occur at all in control plots ($p < 0.001$, two-sample t-test). Uncut *Ceanothus* were more abundant in controls ($p < 0.001$, two-sample t-test). The abundance of small *Ceanothus* did not differ between treatments and controls ($p = 0.19$, two-sample t-test).

Figure 20. Average abundance of *Ceanothus* stages in 2004, control versus treatment plots, with standard error. Abundance codes ranged from *0* for absent to *4* for >50% cover.



Comparing differences between matched treatment and control plots (treatment minus control), the combined stages of *Ceanothus* were less abundant after treatment ($p = 0.030$, one-sample t-test). The abundance of the species across stages was more reduced by the mastication treatment than it was by the HPB treatment (Figure 21; $p = 0.050$, two-sample t-test). The difference in small *Ceanothus* for matched pairs was greater in the second year than in the first year after treatment (Figure 22; $p = 0.003$, two-sample t-test). None of the changes in other stages differed significantly between year-since-treatment nor between types of treatment.

Figure 21. Average difference in abundance classes for *Ceanothus* stages by treatment type (treatment minus control), with standard error. Abundance codes ranged from *0* for absent to *4* for >50% cover.

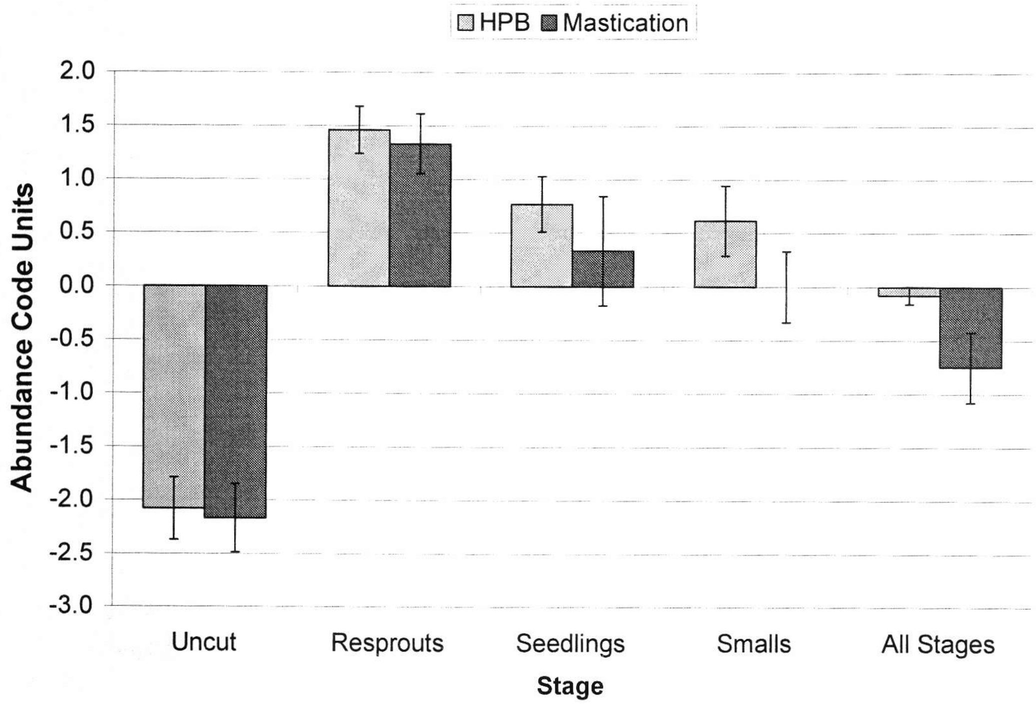
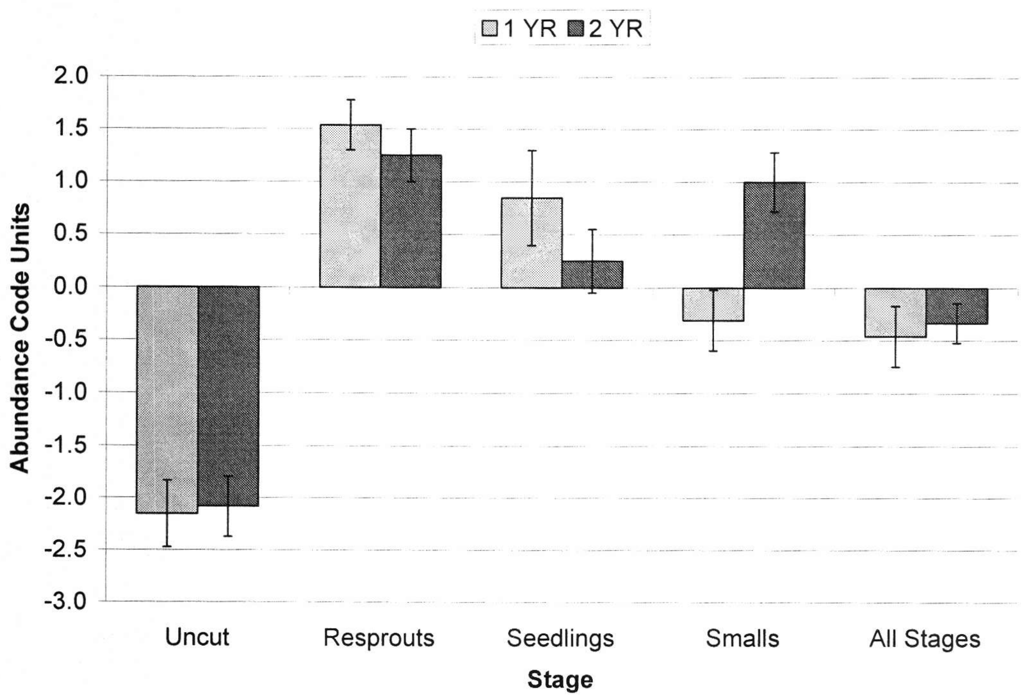


Figure 22. Average difference in abundance code units for *Ceanothus* stages by year since treatment (treatment minus control), with standard error. Abundance codes ranged from *0* for absent to *4* for >50% cover.



Fire rings of HPB treatment

Fire rings differed in species composition and abundance between 1 yr since treatment and 2 yrs (MRPP: A = 0.108, $p < 0.001$). Only one species was a significant indicator for first year fire rings: *Brodiaea elegans,* a native perennial (ISA: IV = 50.0). Twenty species were significant indicators for second year fire rings, all of them annuals, most of them weedy, and about half being exotic (Table 11).

Table 11. Indicator values for species in percentage of perfect indication for second year fire rings. Statistically significant indicator species only ($p < 0.05$). Boldface indicates non-native species.

Annuals, not noted as weedy	IV	Weedy annuals	IV
Agoseris heterophylla	66.7	***Aira caryophyllea***	48.9
Cardamine oligosperma	42.9	***Bromus japonicus***	38.1
Clarkia purpurea ssp. quadrivulnera	34.6	***Bromus tectorum***	33.3
Cryptantha torreyana	28.6	***Cerastium glomeratum***	33.3
Linanthus bicolor	37.0	*Epilobium* spp.	85.7
Madia exigua	39.7	***Galium* spp.**	76.2
Madia spp.	38.1	***Gastridium ventricosum***	42.9
Phlox gracilis	28.6	***Lactuca serriola***	28.6
		Lotus humistratus	33.3
		Myosotis discolor	28.6
		Veronica spp.	29.6
		Vulpia myuros	47.6

The HPB treatment plots in entirety also differed in species composition and abundance between the 1 yr and 2 yr since treatment groups (MRPP: A = 0.082, $p = 0.005$). Twelve species were significant indicators for one of the two groups (Table 12). Only five of them are annuals, and only two of these are considered weedy. The only species to appear as an indicator for both fire rings and HPB treatment plots was *Myosotis discolor* (changing forget-me-not).

Table 12. Indicator values for species in percentage of perfect indication for HPB treatment plots by year since treatment. Statistically significant indicator species only ($p < 0.05$). Boldface indicates non-native species.

1 yr since treatment	IV	2 yr since treatment	IV
Clarkia gracilis	76.9	*Agoseris grandiflora*	100.0
Elymus elymoides	68.0	*Calochortus tolmiei*	81.8
Lomatium utriculatum	82.6	*Galium porrigens*	100.0
Myosotis discolor	76.5	*Hesperolinon micranthum*	61.2
Plagiobothrys cognatus	71.4	*Horkelia daucifolia*	85.7
Poa secunda	75.0		
Trifolium willdenowii	80.0		

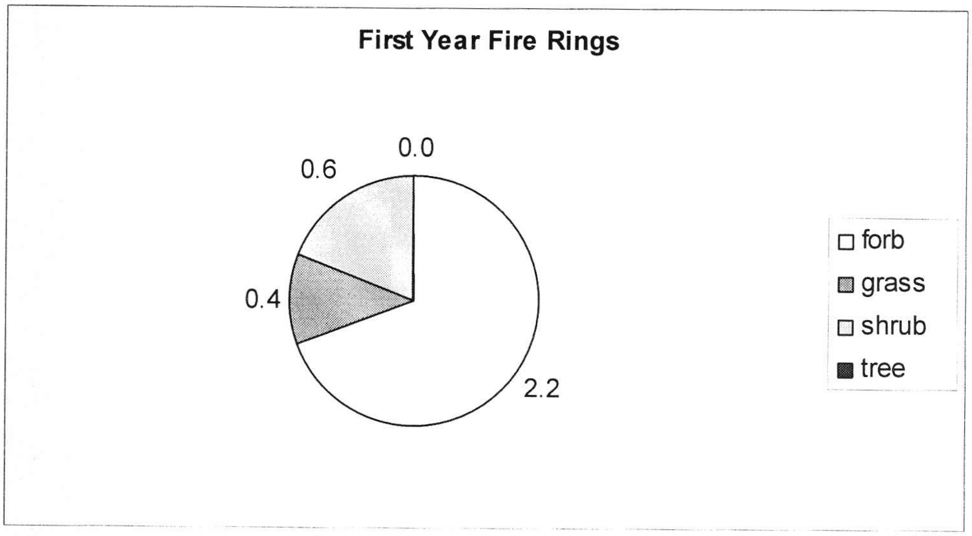
Average number of species per fire ring differed greatly according to ring age. The first year fire rings averaged 3.2 species per ¼-m^2 plot, while the second year rings averaged 14.2 species. Average abundance value per species was 1.1 in the first year, indicating that almost all species had only 1-3 individuals present. The average abundance class was 1.5 in the second year fire rings, which would be the value if half the species had 1-3 individuals and half had 4-10. The HPB treatment plots in entirety, with 200 times the area of the fire ring plots, had an average of 77.6 species per plot. The average abundance value was 2.5, with most species having more than 10 individuals.

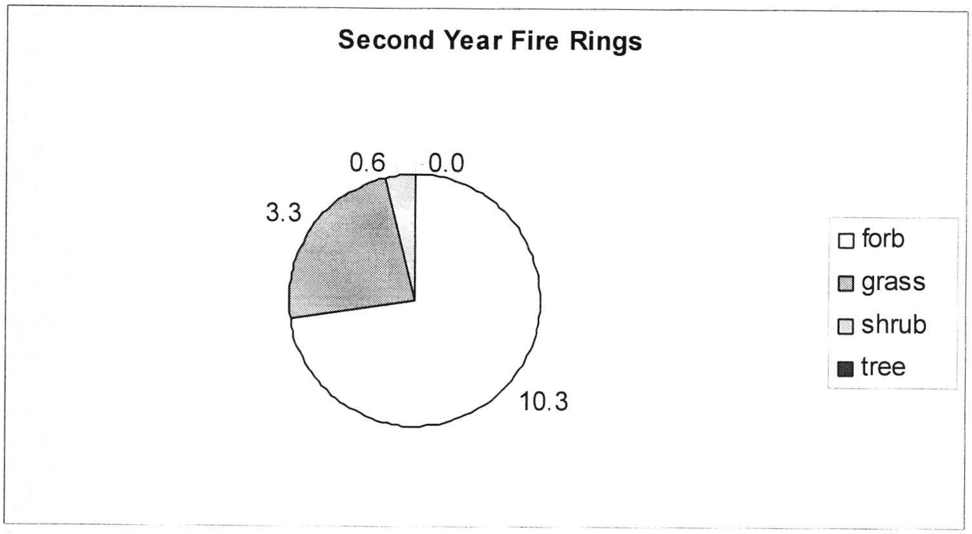
The proportions of species by trait in first year fire rings, second year fire rings, and HPB treatment plots are shown in Figures 23-26. Though species composition differed between 1 yr and 2 yr since treatment groups for HPB treatments (MRPP), the proportions of species traits did not differ appreciably between these groups, therefore years are combined for HPB treatment plots in Figures 23-26. The only trait whose number of species present differed significantly between years for the HPB treatment plots was native weeds ($p = 0.013$, two-sample t-test). There was an average of 3.7 more native weed species in the first year since treatment plots than in the second year (with a 95% confidence interval between 1.0 and 6.5 species). In contrast, the number of species present for all traits except shrubs and perennials differed significantly between the two ages of fire rings ($p < 0.001$, two-sample t-test).

The first year fire rings show a disproportionate amount of shrub species compared to second year fire rings and HPB treatment plots (Figure 23). However, very few species are present in first year fire rings, such that shrub species represent a large proportion of those present. In fact, first year and second year fire rings have the same average number of shrub species, 0.6 per ring plot. *Ceanothus* seedlings are commonly found in fire rings of both ages, at a density greater than elsewhere, affirming the oft-cited fact that their germination is stimulated by fire. By the second year the distribution of life forms in the fire rings is similar to that of the HPB treatment plots in entirety (Figure 23).

The proportions of herbaceous annuals and perennials (excluding woody species) differ greatly in all three cases (Figure 24). The first year fire rings have slightly more perennials than annuals. Most are probably individuals that survived the

Figure 23. Average number of species (indicated by bordering numbers) per plot by life form.





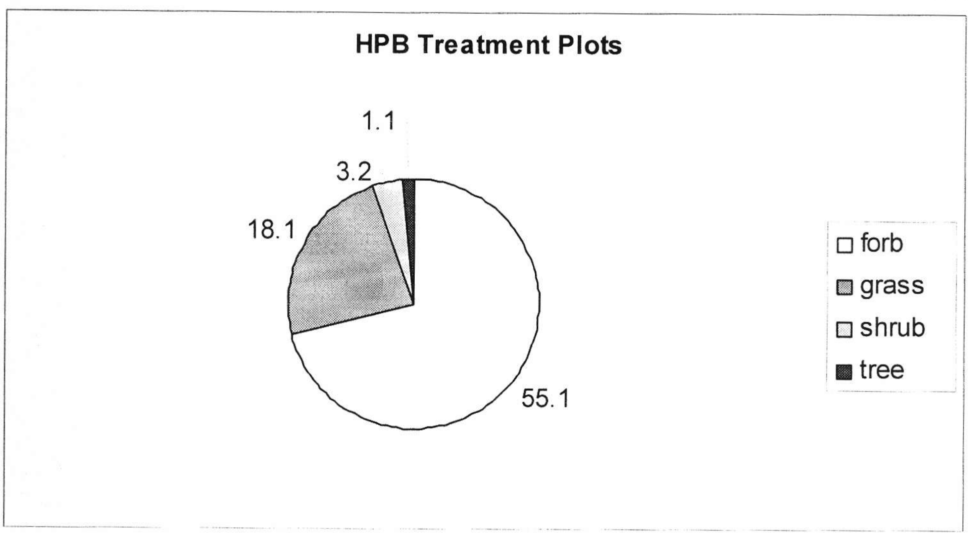
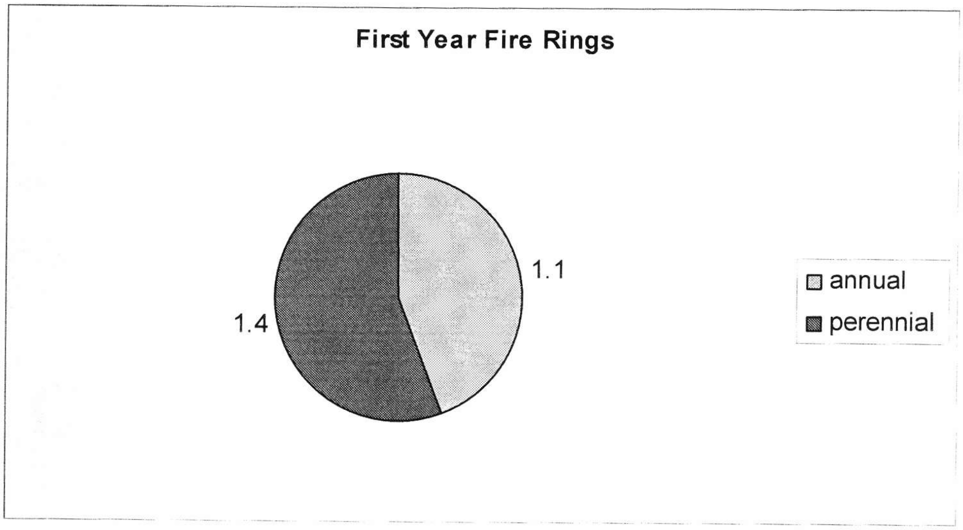
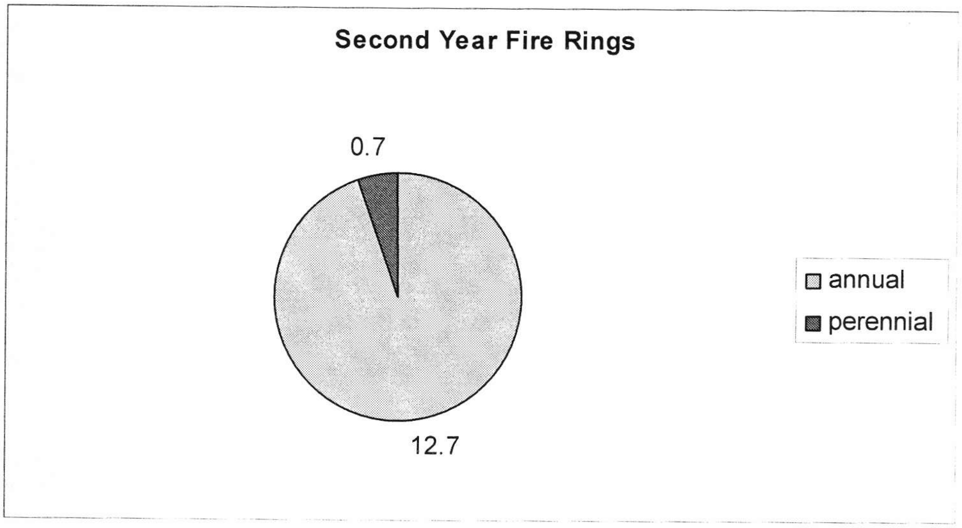
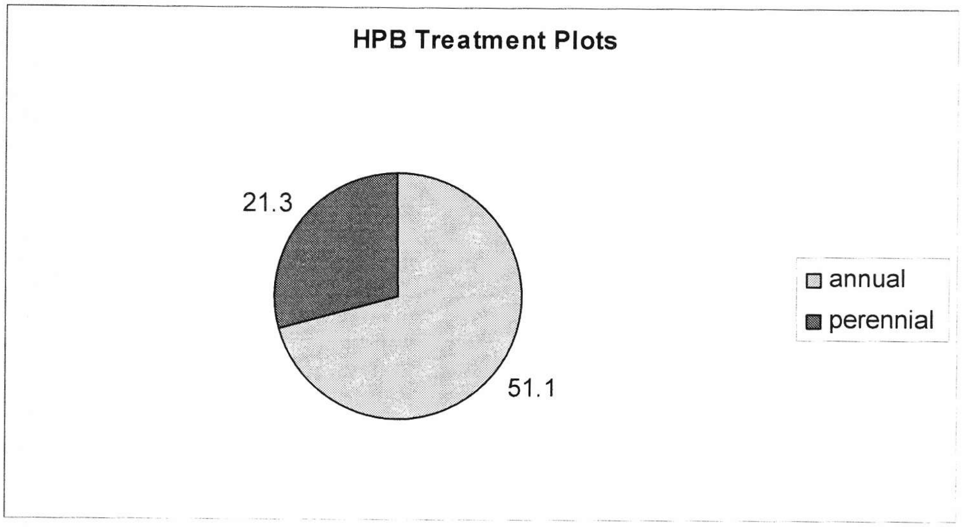


Figure 24. Average number of herbaceous species per plot (indicated by bordering numbers), annuals versus perennials.







fire. Recall that the only indicator species for the first year rings was the liliaceous *Brodiaea,* which has an underground bulb. Very little colonization of the newly open area has occurred by year one. In the second year, colonization is taking place, as evidenced by the high proportion of annual species as a whole, and as indicators (Table 11).

Exotics and natives (Figure 25) show patterns similar to annuals and perennials, in that the two ages of fire rings show two extremes, while the entire treatment plots lie between the two. In the first year, natives comprise the majority of species in fire rings. By the second year, the proportion of exotic species has increased dramatically relative to the number of natives.

The proportion of native weeds compared to the sum of introduced weeds and non-weedy plants remains relatively constant in all three cases (Figure 26). It is the introduced weeds that vary strikingly. There are comparatively few in the first year fire rings, but they increase to twice the number of native weeds in the second year. The HPB treatment plots show an intermediate proportion of introduced weeds.

In comparing the species lists for fire rings, HPB treatments and HPB controls, I found several species that occurred in one but not another. However, even species that appeared to be stimulated by fire, such as *Ceanothus* seedlings, were present in control plots as well as in treated areas. Three species occurred in more than one treatment plot but not in any control plots, and all of these appeared to grow in association with burn piles. One of them, *Lactuca serriola* (prickly lettuce), an annual introduced weed, was an indicator species for the second year fire rings (Table 11). The other two species are native annuals, *Stephanomeria virgata* (rod wirelettuce) and *Gnaphalium palustre* (western marsh cudweed). Though only the *Gnaphalium* was considered weedy by my sources, the *Stephanomeria* is a composite with wind-dispersed seeds.

Figure 25. Average number of exotic and native species (indicated by bordering numbers) per plot.

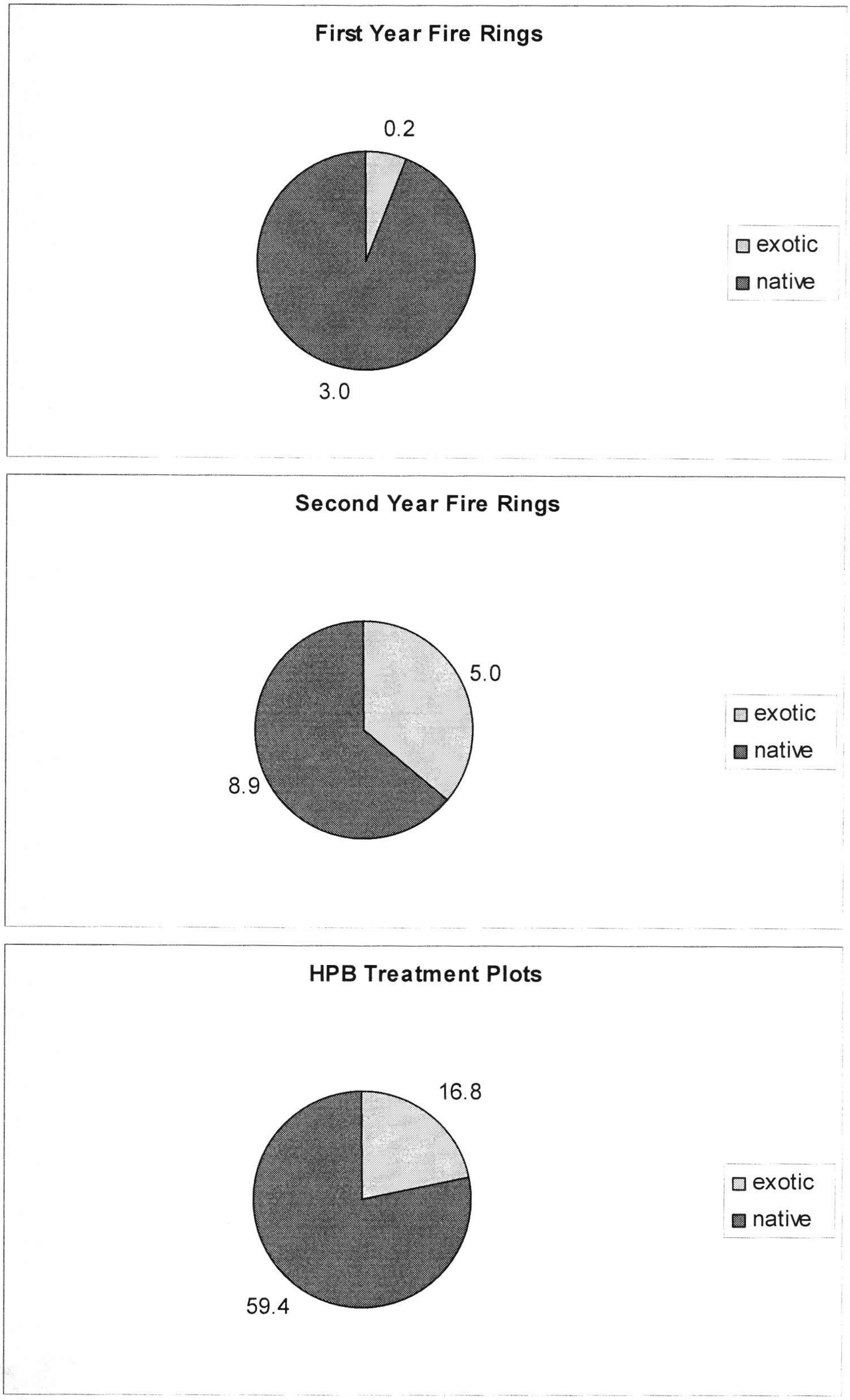
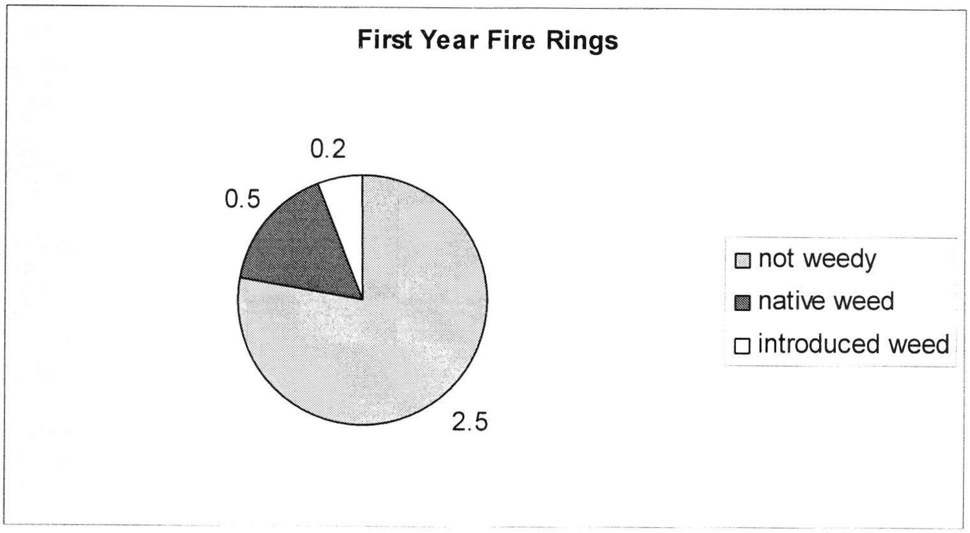
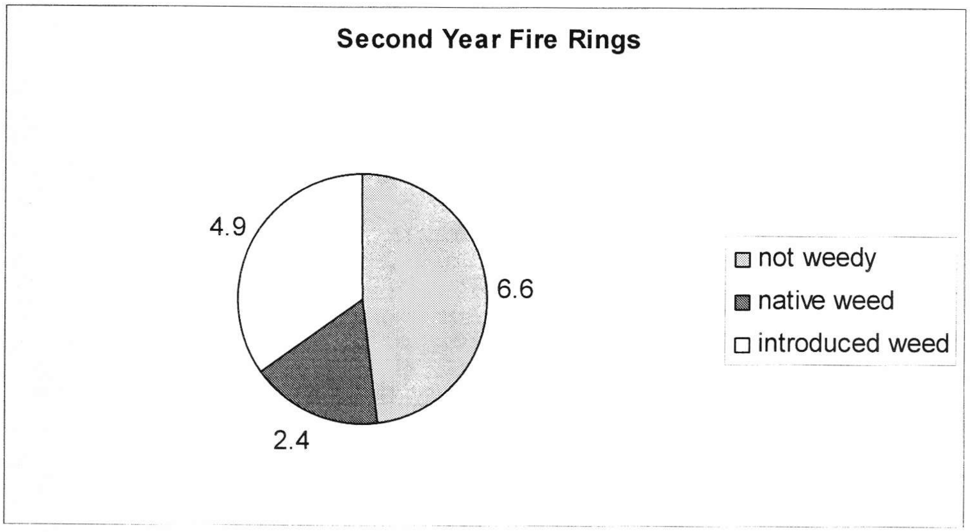
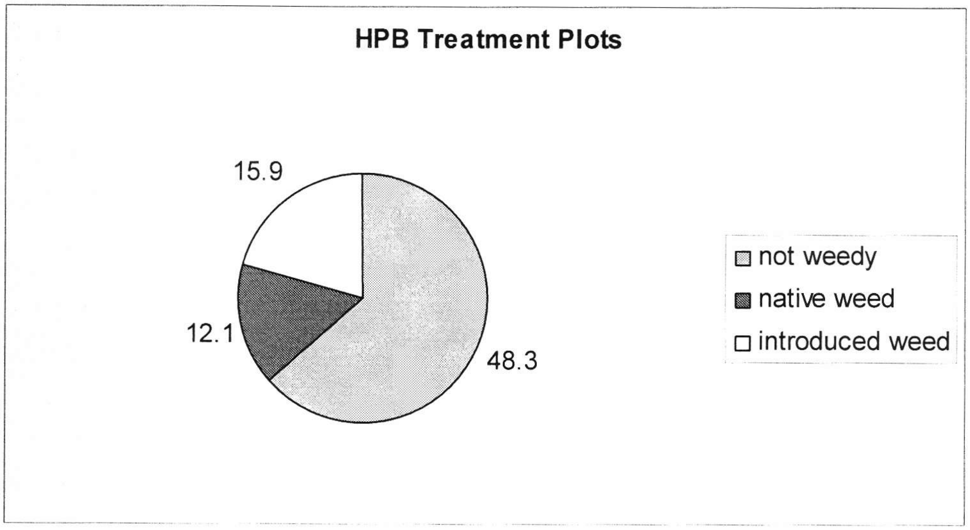


Figure 26. Average number of species by plot (indicated by bordering numbers), categorized by weediness and place of origin.







DISCUSSION

Community Composition

Much of the difference in pre-treatment species composition detected between groups of plots assigned to the two different treatments can probably be ascribed to environmental differences. Assignments to treatment type are made by managers using considerations of terrain (e.g. rockiness, steepness of slopes) and accessibility, which may in turn affect species composition. The stronger difference in composition detected among management units than between treatment assignments demonstrates a spatial correlation in species composition. Since plots within management units were generally closer to each other than they were to plots in other units, this spatial correlation may explain much of the pre-treatment difference shown between plots assigned to different treatments.

Variability of species composition across the study area increases the importance of pairing in the study design. Comparing community differences between treatment and control of matched pairs lessens the impact of site variability. For example, the significant difference in species composition between plots established in 2003 versus those established in 2004, when sampled in 2004, related strongly to the difference in oak canopy. However, when the species composition differences between paired treatment and control plots were compared between establishment years across the same plots, they did not differ. This result met my expectation, because the species composition variability caused by different environmental conditions ideally would not affect differences between matched plots, which should be attributable largely to the effect of treatment. However, the pairs grouped by management unit differed in species composition changes between treatment and control, which was unexpected.

Data taken in 2003, before treatments occurred, included only 166 taxa in 26 plots. Fewer species were seen in 2003 than in 2004 because fewer sites were sampled and sampling occurred later in the season. The general increase in species richness and abundance in 2004 for plots that had been sampled in 2003 can be attributed mainly to sampling earlier in the year, before more ephemeral spring flora had dried up and disappeared. Abundance of graminoids, shrubs, and trees changed little between years because these groups were not susceptible to this difference in sampling date, being as easy to detect in mid-summer as in May. The numbers of species detected may also have increased in 2004 because I was a more experienced sampler.

Changes in species composition of the same plots sampled in 2003 and 2004 had more to do with sampling date than with treatment. Indeed, the A-value from MRBP was very close to zero when the year-to-year changes in species composition were compared between treatment versus control plots, meaning that the heterogeneity within the groups was almost that expected by chance (McCune & Grace 2002). Similarly, only a small effect of treatment was detected using ordination of the same data. The change in cover class of standing dead *Ceanothus* was correlated with Axis 2, which explained less than 10% of the variance present.

The strong influence of phenology on community changes between years was substantiated by the first axis of the ordination of changes in plot species composition between 2003 and 2004, which explained over half of the variance present in the data and was highly correlated with difference in date of sampling, 2004 sampling date, and elevation. Elevation can affect the presence of a species on a given date, and therefore its relationship to sampling date supports the conclusion that phenology was responsible for most of the community composition differences between the two years. The important influence of sampling date provides another reason to base conclusions about treatment effects on differences between matched pairs, which were sampled on or near the same day.

The effect of sampling date was compounded when comparing composition from two different years, since the sampling date in both years had an effect. Phenology is probably not as important in results for a single year. Elevation was not correlated with the ordination of 2004 post-treatment data. Here sampling date was correlated with the ordination because of variation across sites that coincided with my moving from area to area to sample. It was also related to the year of plot establishment, since the sampling dates for plots established in 2003 were all earlier than those for 2004 plots. In turn, sampling date correlated with species composition because the plots established in 2004 happened to have a greater oak influence than those established in 2003.

The ordination of 2004 species composition provides some basic information about how community composition and the environment were related. For example, the variables of slope and rock cover were negatively correlated with each other.

Slope can vary from place to place within a plot, but some of the moisture-loving species tended to concentrate in the flatter plots, where water might remain for longer periods. Species that tended to occur primarily in relatively rocky flat plots (lower end of Axis 2 in the ordination) are *Epilobium densiflorum* (denseflower willowherb), *Mimulus guttatus* (seep monkeyflower), *Camassia quamash* (small camas), and *Juncus bufonius* (toad rush), which are all associated with relatively moist habitats (Hickman 1993). Some species were associated with rock outcrops, such as *Minuartia douglasii* (Douglas' stitchwort) and *Pentagramma triangularis* (goldback fern), which I found growing only in the shade of large rocks. The rockiest flattest plots were generally on Medco-McMullin complex soil map units.

Other environmental variables that correlated with community composition were probably influenced by oak canopy cover. Cover and depth of litter were strongly affected by the presence of oak canopy, since white oaks produce much more abundant litter than *Ceanothus* or any other plant found in the plots. Cover of bare soil was, in turn, directly affected by the amount of litter. Rock cover was negatively associated with oak canopy, in part because rock may be hidden by fallen leaves, but also because surface rock is indicative of thin soils which cannot support oak trees.

Cover of oak canopy had the strongest detectable influence on species composition of any measured environmental attribute. One indication of the strength of this influence is the large number of strong indicator species for the group of plots with >10% canopy. Perennial species dominated the list of indicators of more canopy and the perennial trait had the strongest correlation with overall species composition of any species trait. At the same time, canopy cover had the strongest correlation of any environmental attribute with the same ordination axis. Areas under oaks were much more likely to support native perennial forbs and graminoids than other areas that I sampled.

Another indication of the importance of tree canopy is that groups that differed in community composition also differed in canopy cover (e.g. plots grouped by establishment date ignoring treatment and control). Amount of canopy probably drove the differences in species composition between first year and second year HPB treated plots. Recall that the difference in community composition between HPB treatments and controls was quite small compared to the difference between first and second year HPB treatments. Also note that the species that were indicators of the second year HPB treatments were all indicators of >10% canopy, while more than half of the first year group were indicators of <10% canopy. Though HPB treatment plots are just a subset of the plots shown in Figure 17, they follow the same trend of greater canopy cover in the group that was treated 2 yrs ago.

Presence and abundance of shrubs was clearly related to canopy cover. The shrub species trait was positively associated with the canopy cover variable on the ordination of species composition in 2004. The correlation of shrubs to canopy was driven largely by species other than *Ceanothus* that tend to grow in association with oak. Species scored as shrubs on the list of indicators for >10% oak canopy include *Toxicodendron diversilobum* (poison oak) and *Lonicera interrupta* (chaparral honeysuckle), a woody vine. Other woody species that grew under oaks but were too rare in my plots to be significant indicators are *Berberis aquifolium* (hollyleaved barberry), *Rosa californica,* and *Symphoricarpos albus* (common snowberry). *Arctostaphylos viscida* (manzanita) also grows in closer association with oaks than *Ceanothus* does (Detling 1961). Manzanita appeared as an indicator of plots with fewer *Ceanothus.*

Ceanothus was rarely found under oaks. *Ceanothus* are adapted to high light environments such as those that occur after fire and don't do as well under shaded conditions (Keeley 1992a). The fact that one indicator of plots with <10% tree canopy in 2004 was also an indicator of 2003 plots with *Ceanothus* regeneration (*Pectocarya pusilla*, a weedy native annual), illustrates that open areas without much shade provide suitable microhabitats for *Ceanothus* regeneration. By the same token, *Cynosurus echinatus*, a weedy exotic annual grass, was an indicator of >10% canopy in 2004 samples (its association with oaks has already been documented; Riegel et al. 1992) as well as lack of small *Ceanothus* in 2003 plots. Species that grow in areas with more tree canopy also tended to occur in areas with less *Ceanothus,* and vice versa, as suggested by the fact that *Cynosurus* was also an indicator of few mature *Ceanothus* in 2004 control plots. When values of canopy cover class and abundance of mature *Ceanothus* were compared for the 2004 control plots, the inverse relationship between them was confirmed.

Plots with abundant mature *Ceanothus* are more likely to show *Ceanothus* regeneration. Therefore it is not surprising that *Potentilla glandulosa* (sticky cinquefoil), a native perennial, was a significant indicator of 2003 plots with small *Ceanothus*, as well as of 2004 control plots with abundant mature *Ceanothus*. The other significant indicator species for the plots with some *Ceanothus* regeneration, whose communities differed from those without such regeneration, were all natives, including two annual forbs, a shrub, and another perennial forb. The indicators for the plots without small *Ceanothus* were two exotic grasses (one annual and one perennial) and a native annual clover. All of the latter are considered characteristic of disturbed habitats, while only one of the other group's indicators has that distinction. These patterns suggest that *Ceanothus* regeneration occurs under conditions that also benefit other native species, or that more disturbed areas are less conducive to *Ceanothus* regeneration.

Regeneration is not limited by seed dispersal to close proximity to existing shrubs, since *Ceanothus cuneatus* can cast its seed as far as 9 m away from the parent plant (Evans et al. 1987). Studies have found that competition for water has the strongest influence on seedling survival for shrub species such as *C. cuneatus,* and that recent fire assists establishment by reducing that competition (Schultz et al. 1955). For seedlings of the congener *C. impressus* after fire, there was a significant positive relationship between distance of nearest neighbor and both survivorship and relative growth rate (Tyler & D'Antonio 1995).

The indicator species for groups distinguished by mature *Ceanothus* abundance demonstrate that the canopy of this shrub provides good habitat for many native plant species. Perennial species were often indicators of plots with mature *Ceanothus* abundance as well as those with relatively abundant tree canopy cover. Though shrub canopy may suppress some natives, open grassy areas are more likely to harbor exotic species, as evidenced by the weedy plants which were indicators of low mature *Ceanothus* abundance and lack of *Ceanothus* regeneration. "Where shrubland and grassland meet, there is often a striking segregation of native and introduced grasses" with exotics dominating the open areas (Wells 1962, p. 80). Though heavy grazing is considered one of the main factors that allowed native grassland to be invaded by exotic grasses in the West, reducing or removing that disturbance does not permit such areas to return to native dominance (Seabloom et al. 2003).

Shrub facilitation of safe-sites for plant establishment is well documented for harsh environments (Dunne & Parker 1999). Oak recruitment is often facilitated by shrubs, perhaps due to more intense competition with grasses than shrubs, or to higher animal dispersal rates under shrubs (Callaway & Davis 1998). Additionally, shrub cover may protect oak and shrub seedlings from grazing. Grazing exclosures established in Sierra Nevada foothill chaparral after clearing showed that areas open to deer had reduced brush regrowth (including *Ceanothus cuneatus*), which reached 48% cover after 16 years compared to 79% cover for the area with complete grazing exclusion. Areas open to grazing by both deer and cattle had even less regrowth (Johnson & Fitzhugh 1990).

Oak and *Ceanothus* were influential in the plant community because of their size. By number of individuals, it was exotic annual grasses that dominated the plots. Their influence should not be discounted. Studies have repeatedly documented a negative correlation between cover of exotic grass and the success of native species (D'Antonio & Vitousek 1992). They are effective competitors for resources, especially for moisture. Areas dominated by exotic annual grasses are somewhat resistant to native shrub establishment (Keeley & Fotheringham 2003). Once introduced grasses have come to dominate a site, native forbs may continue to decline because of limited seed production (Seabloom et al. 2003). At least one species present in my plots (*Avena fatua* or wild oat), though uncommon, has been labeled allelopathic for inhibiting herb germination (Tinnin & Muller 1971). As exotic annual grasses come to dominate the groundlayer they undoubtedly alter the ecology of the site from the type of conditions experienced by members of the native community in the past. The fine dry fuel of annual grasses is also flammable. Invasions of medusahead, perhaps the most abundant species in my plots, and *Bromus tectorum* (cheat grass; less common in my plots) are believed to promote more frequent fire (D'Antonio & Vitousek 1992).

Effects of Treatment

Results showed that the pre-treatment and control plots were well matched because they did not differ significantly in species composition. The lack of pre-treatment difference in community composition between control plots and those destined for treatment meant that differences between treatment and control in subsequent post-treatment data could more confidently be ascribed to the effect of treatment. However, evidence for treatment effects was slight. When data from two years of sampling on the same plots were combined, species composition did not differ between treated and untreated plots. Nor did the change in composition between 2003 and 2004 vary between treatment and control. As mentioned earlier, less than 10% of the variance in the ordination of change in composition between years was correlated with a variable associated with treatment.

The most direct test of treatment effect on overall species composition was achieved using the 2004 abundance data, focusing the analysis on differences within each pair (MRBP). Though significant differences were detected, the effect size was quite small. When data were reduced to the mastication units or those that were treated two years ago, treatment and control did not differ. It was somewhat surprising that when the year-since-treatment groups were divided by treatment type they showed similar MRBP results; both types of 1 yr-since-treatment differed while both types of 2 yr-since-treatment did not differ. This result might be explained by the fact that the first year and second year effects of the mastication treatment were quite different, and therefore tended to cancel each other out when combined. No pattern related to treatment was discernible in the ordination of the 2004 data.

In general, the effect of time since treatment was stronger than the effect of which treatment method was used. Taking all species composition differences between matched treatment and control plots into account, I found that the two types of treatment did not differ, while the year-since-treatment groups did differ. The ordination of these data, differences between matched pairs, illustrates this finding. Plot pairs group by year since treatment, while no patterns can be seen in the positioning of the two treatment types. This result parallels that found for comparisons between trait group abundance differences, in which twice as many trait groups differed by time since treatment as by treatment type.

There was a general increase in species abundance and species richness with treatment, based on the difference between matched treatment and control plots. It is somewhat surprising that overall shrub abundance did not decrease with treatment, since the abundance of mature uncut *Ceanothus* was so reduced by it. While total *Ceanothus* abundance was significantly decreased after treatment, the effect was smaller than anticipated because of resprouting from cut stems and an increase in *Ceanothus* seedlings with treatment. In general, seedlings and resprouts were significantly more abundant in treatments than in controls. Overall shrub abundance was also affected by the occurrence of other species. Other shrub species either were not removed because of low density, or tended to resprout more than *Ceanothus*. Woody species that appeared to resprout without fail after cutting included poison oak, Oregon white oak, *Amelanchier alnifolia* (serviceberry), and *Prunus subcordata* (Klamath plum).

Whereas shrub abundance did not differ overall between treatment and control, I found there was a significant difference in the change in shrub abundance between types of treatment. The HPB treatment appeared to cause more increase in abundance of shrubs than did the mastication treatment. At the same time, abundance of all stages of *Ceanothus* was apparently more reduced by mastication than by the HPB treatment. The fire rings of the HPB treatment were responsible for the difference between treatment types. Though *Ceanothus* seedlings occurred in the majority of plots in 2004, whether control or treatment, their affinity for burned areas was evident. Differences in seedling abundance between treatment and control were greater in the HPB treatment than the mastication treatment, though not significantly.

The results concerning *Ceanothus* stages included two surprising findings considering the literature on the species. Previous work has emphasized that *C. cuneatus* is an obligate seeder that requires fire for seedling establishment and will not resprout after fire (e.g. Keeley 1992a). Therefore it was unexpected to find so many *Ceanothus* seedlings outside of the fire rings and in so many control plots where disturbance was minimal. Seed dormancy in *C. cuneatus* is due to a hard impermeable seed coat that may be cracked by the heat of fire or by scarification (mechanical breakage; Keeley 1991). The seed coat may also deteriorate with time to allow germination (Quick & Quick 1961). In addition, it is not unusual for some fraction of seed produced by fire-recruiting species, including *Ceanothus* species, to lack dormancy (Keeley 1991). These alternative situations, which allow germination of *Ceanothus* in the absence of fire, appeared to be in operation throughout my study area.

Because seedlings were generally more abundant in mastication treatments than in controls, there is some indication that the treatment may have increased germination of *Ceanothus*, either through causing scarification or through improving microhabitats. Another species, *C. greggii,* showed increased germination a year following the clearing of standing chaparral by clipping at about 10 cm from the ground (Moreno & Oechel 1991). Though often described as having no significant germination in the absence of fire, in a previously disturbed site that was quite open and invaded by exotic annual grasses, conditions similar to my research sites, Keeley (1992b) found both seedlings and uneven-aged shrubs of *C. cuneatus.*

The presence of resprouting *Ceanothus* was also surprising. In the chaparral literature *C. cuneatus* is sometimes referred to as a "non-sprouting species" in association with its status as an obligate seeder after fire (e.g. Keeley 1992a). The resprouting that I observed in *Ceanothus* occurred from above-ground buds that would not have survived burning. In older unburned stands it rarely initiates new stems after the first decade since fire, unlike resprouting shrub species (Keeley 1992b). Certainly *Ceanothus* is not as prone to resprouting as some other woody species that were found in my plots (mentioned above), and its resprouts are less vigorous than theirs. The average abundance for resprouts in treatment plots was 1.5 abundance code units, which would be the average if half of the treatment plots contained 1-3 resprouts and half had 4-10 resprouts. The abundance of resprouts can be roughly compared to the mean decrease in uncut *Ceanothus* between matched treatment and control plots, which was -2.1 abundance code units. This value indicates that more than 10 individual *Ceanothus* plants were cut down to the base in the average treatment plot. I would estimate from these values and my observations that less than half of the *Ceanothus* plants that were removed during treatment resprouted. However, resprouted individuals may or may not recover to produce significant crowns or to reach a reproductive stage.

The other species trait group that was significantly more abundant in HPB treatments than mastication treatments was native weeds. The mean number of increasing native weed species in HPB treatments relative to controls was 26%, while the corresponding change was a 3% decrease in mastication plots. Unlike the findings for *Ceanothus,* the proportion of native weeds was not greater in fire rings compared to the larger treatment plot, so fire rings were not responsible for the difference between treatments. Only 2 of the 20 indicator species of second year fire rings were native weeds, *Epilobium* spp. and *Lotus humistratus* (foothill deervetch). On the other hand, the only trait group to vary significantly between the years-since-treatment groups of HPB treated plots was native weeds. More native weed species were present in the first year after treatment than the second, even though only one of seven indicator species for first year HPB treatments (versus second year) was a native weed, *Trifolium willdenowii* (tomcat clover).

Why would native weeds be particularly favored in the first year after an HPB treatment occurred? Weedy plants would be expected to increase after either type of treatment, especially in the first year, because of the availability of newly exposed ground for establishment. Treatments did not differ in the amount of bare ground or woody debris, so these are unlikely to cause a difference between treatments in native weed species abundance. Perhaps differences in the distribution pattern of woody debris could help explain the difference in native weed abundance between types of treatment. Mastication leaves pieces of debris distributed fairly evenly over the area while HPB leaves debris clustered at the edge of burn piles. Another difference in the two treatments that might have an effect is degree of soil disturbance, which was not measured. Both treatments cause disturbance, but mastication causes some areas to be compacted by the treads of heavy equipment and leaves other areas of soil undisturbed. HPB treatment lightly compacts and disturbs most areas of the soil with foot traffic.

As with composition differences between groups separated by time since treatment (Table 4), there were greater increases in species abundance for treated plots compared to controls in the first year than in the second year after treatment. In the first year, more light and resources had just been made available, so the groundlayer responded accordingly. One might hypothesize that the increases would be disproportionately large for weedy plants, since they specialize in colonizing new and disturbed habitats. However, the weedy trait groups were only somewhat more abundant, not significantly, in the first year than the second year, and their percentage change was not large in comparison to that observed for other trait groups.

Species richness seemed to be increased by the disturbance of treatment. However by the second year after treatment, species richness was actually less in treatment plots than in controls. When the two treatments were each divided into year-since-treatment groups, only mastication treatments had less species richness than their controls in the second year. In the HPB pairs, species richness was also lower in the second year compared to the first year after treatment, but treated plots were still more diverse than control plots. Though these data address only short term changes, it appears that mastication treatments may reduce species richness over the long term.

Unlike species richness, total species abundance in treated plots remained greater than in controls in both years after treatment. However, when the two treatments were analyzed individually by years since treatment, it was apparent that species abundance was reduced in the second year of mastication treatment in comparison to controls, unlike in the HPB pairs.

Another way that plots differed between the first and second year after treatment was in the overall *Ceanothus* stage composition. Apparently the small *Ceanothus* stage was responsible for the difference. Small *Ceanothus* plants tended to be removed during treatment, and could not be replaced within the first year since new seedlings and resprouts had their own stage categories.

In addition, differences in treatment and control seemed to vary with environmental variables. The ordination of species composition differences between matched treatment and control plots did not correlate with the same static environmental variables as the ordination of the total species composition of the plots. Slope and date of sampling were related to total species composition but not to the treatment effect. Cover of bare soil was correlated with overall species composition but the difference in bare soil between matched plots was not correlated with their difference in species composition.

The abiotic variables that were correlated to both ordinations (overall species composition and paired treatment less control differences) were maximum litter depth and cover of litter and rocks. This is somewhat surprising because the sum of differences in maximum litter depth across pairs was zero, and while cover of both litter and rocks tended to decrease with treatment the decrease was small. Treatment might decrease litter by reducing overstory. It could affect the cover of rocks only if debris or other types of cover hid rocks from view. The only cover variable that differed significantly from zero was the change in woody debris between treatment and control. However this difference, which would be expected, was not correlated with the difference in species composition. Woody debris increased in both types of treatment because mastication generates it and HPB always leaves debris rings around incompletely burned piles.

Cover of tree canopy was correlated with both ordinations. Both average canopy and difference in canopy were correlated with the difference in species composition between matched plots, but negatively correlated with each other. The correlation between the variables of average canopy and change in canopy was $r = -0.281$. The greater the average canopy cover for a plot pair, the more likely it is that a large difference exists between the canopies of the two plots in the matched pair. Since treatment plots tended to have less canopy cover than their controls, the larger difference under conditions of relatively large canopy cover was a more negative number.

As already noted, greater tree canopy cover was associated with greater species richness and more abundant perennials, natives, and grasses. Why then was it negatively correlated with differences in abundance of perennials as well as the difference in canopy cover between matched treatments and controls that was explained above? Again, the negative numbers associated with subtracting control values from treatment values can cause confusion. Imagine a pair of plots that has no canopy and the same abundance of perennials in both plots, this pair would lie near the middle of the ordination (Figure 13). Imagine a second pair of plots with greater canopy cover in the control plot than in the treatment plot. The difference in canopy between them would be a negative number and therefore this pair would have a lower position along Axis 1. Because there would tend to be more perennials in the plot with more canopy cover, the control plot in this case, the difference in perennials would also tend to be a negative number. In this way, difference in canopy is positively correlated with difference in perennials ($r = 0.522$), as well as difference in species richness, graminoids, and natives. One can imagine that the correlation between difference in perennials and difference in canopy would hold true for any plot pairs that differed in amount of tree canopy cover, whether they had undergone shrub removal or not. Unfortunately, this theory could not be tested using my pre-treatment plots sampled in 2003, since only four pairs had any canopy cover.

The expectation is that canopy cover would be reduced by treatment, but this was not necessarily the case. In fact, on average, canopy cover was somewhat lower in treatments than in controls, but tree basal area was greater in treated than in control plots. Though neither difference was significant, these trends contradict each other and belie the assumption that differences in canopy cover were an effect of treatment. Since trees tended to be well-spaced in these sites before treatment occurred, few trees were removed. Additionally, the random location of the plots, along with the spacing of the trees, meant that abundance of trees or canopy cover was not reliably associated with treatment.

As in the ordination of the composition changes between 2003 & 2004, one correlated variable in the ordination of differences between treatment and control was clearly an effect of treatment. The change in abundance of large uncut *Ceanothus* shrubs was correlated with an axis that represented 15% of the variance in the data. The change in abundance of several species traits was negatively correlated with this variable. The more *Ceanothus* that was removed by treatment, the more positive was the difference for annuals, forbs, exotics, introduced weeds, and special status plants. In general, a goal of management is to reduce, not promote, the annual, exotic, and introduced weed trait groups. It is not surprising that these types of plants would react positively to the disturbance of treatment. On the other hand, increased abundance of forbs and special status plants would be considered a beneficial effect.

Does the removal of shrub cover actually promote special status plants? A review of relevant information gathered will help to address this question. The data include only a few species in a fraction of the plots, since it was intended that treatment boundaries exclude special status plants. Most of these plants are small and inconspicuous, while large areas were surveyed for their presence. Special status plants' abundance differed between treatment and control plots in 9 out of 25 matched pairs that make up the ordination. The five taxa that differ in these nine pairs are *Microseris laciniata* ssp. *detlingii* (Detling's silverpuffs), *Minuartia californica* (California sandwort), *Plagiobothrys glyptocarpus* (sculptured popcornflower), *P. greenei* (Greene's popcornflower), and *Scribneria bolanderi* (Scribner's grass). All are forbs except *Scribneria,* which is an annual grass of dry disturbed areas (Hickman 1993). I found *Scribneria*, as well as the diminutive *M. californica* and *P. greenei,* in open flat areas that probably held water early in the season, often with bare cracked soil. *Plagiobothrys glyptocarpus* was found in only one plot, which had more moisture than the average plot. All of the special status plants are annual natives of open habitats except for the perennial *Microseris,* which I found to be associated with oaks and which was an indicator of >10% tree canopy.

Some special status plants were found in one or more plots but could not be indicated as such in the dataset. *Ranunculus austrooreganus* (Southern Oregon buttercup) was confirmed in one control plot but it could not be reliably distinguished from *R. occidentalis* (western buttercup) in most stages of growth. Two special status *Navarretia* were collected and identified, *N. subuligera* (awl-leaf pincushionplant) and *N. tagetina* (marigold pincushionplant), but were combined with all *Navarretia* in the dataset because they had not been reliably distinguished in the field. *Navarretia tagetina* was confirmed in three plots in 2004, one treated and two controls, while *N. subuligera* was confirmed in 14 plots in 2004, half treated and half control.

In some cases I have pre-treatment data to compare with the post-treatment abundance. However, because of the earlier post-treatment sampling dates, I may have been able to locate more individuals of the special status plants in 2004 regardless of their actual occurrence, so the comparison between years may not be accurate. For *Minuartia californica,* all four of the plots in which it was found in 2004 were also sampled in 2003, pre-treatment. In one mastication treatment plot its abundance was the same in both years. In another mastication treatment plot, its abundance was lower after treatment. The other two plots were a matched mastication treatment and control, which both had more than 10 individuals of *M. californica* in 2004, while none were detected in 2003. For this plant, there is a very slight indication that mastication treatment decreased abundance.

Scribneria bolanderi was recorded in 4 plots in 2003 and 12 plots in 2004 (7 of which had been sampled in 2003). In 2003 it was found in two mastication pre-treatment plots, and one pair of HPB pre-treatment and control. In 2004, post-treatment, the abundance of *Scribneria* was the same as in 2003 in all except one plot that had been masticated, where it was not detected. Other plots that contained *Scribneria* in 2004, which had been established and sampled in 2003 without finding it, included three mastication treatment plots and one of their control plots. Plots with *Scribneria* that were newly established in 2004 and had no pre-treatment data included three HPB treatment plots and two control plots. In this case, the plant was found more often in treated plots, helping to increase the positive correlation between the change in special status plant abundance and treatment.

Since this grass is found in disturbed habitats it is not surprising that treatment might promote it. However, the information from plots sampled in both years is ambiguous, in that one plot appeared to have lost *Scribneria* after treatment, while it was newly detected in more treatment than control plots.

The three other taxa of special status plants that were included in the ordination occurred equally in treated and control plots. Collectively, they were present in four treated plots and three control plots. Though the positive correlation between special status plant abundance and treatment gives some indication that special status plants were promoted by fuel reduction treatment, the information added by comparing 2003 data gives little reason to believe that such a generalization can be made. On the other hand, there is no real evidence that special status plants were impacted negatively by treatment.

The fire rings that result from HPB treatment deserve individual consideration. The footprints of burn piles provide sites for invasive or weedy plants to gain a foothold (Korb et al. 2004). Though previous research has focused on larger slash piles that result from forest thinning operations, the principles invoked in those cases also apply in our situation. Human-condensed fuel piles burn hotter than almost any naturally occurring fuel load and therefore cause increased negative effects on the substrate, such as water repellency (e.g. MacDonald & Huffman 2004), altered soil chemistry and structure (Shea 1993), seed mortality, and mycorrhizal sterilization (Korb et al. 2004). Soil heating of a longer duration, such as would be expected in a burn pile, causes more damage than shorter duration heating as well (DeBano et al. 1979).

Some factors relevant to fire effects are unique to the chaparral ecosystem and to the circumstances of the burn piles I sampled. Chaparral has a thinner litter layer than forests and therefore soils are less insulated from heat (DeBano et al. 1979). Waxy-leaved shrublands are especially susceptible to water repellency of soil after fire, because of hydrophobic substances produced by chaparral plants (Beschta et al. 2004). On the other hand, shrubland wildfires naturally produce higher soil temperatures than forest fires because of their low, single stratum stature (Christensen 1985). Therefore burn pile temperatures in chaparral may be closer to naturally occurring temperatures than would be the case in a forest system. Natural fires would usually occur at drier times of the year, rather than in the moist conditions that are purposely chosen for burning piles to reduce fire danger. Moist soil may reduce most impacts of heating during fire, though damage to soil micro-organisms can be increased (DeBano et al. 1979). These differences in timing could affect the response of chaparral plants as well.

Though chaparral plants are thought to have evolved in association with regular fire events and high intensity fire, it is not likely that natural conditions would produce such concentrated fuel loads and resultant heat as occur in burn piles. Would any part of the soil under a burn pile experience temperatures that are similar to those caused by a fire occurring in standing, living chaparral? In standing chaparral, seed recruitment after a fall fire tends to be concentrated in areas that were gaps in the pre-burn vegetation, reflecting the high temperatures that result from concentrated areas of vegetation (Odion & Davis 2000). Odion & Davis (2000) moved existing fuels to test whether fire intensity or pre-burn distribution determined the spatial arrangement of seedlings and resprouts and found a positive correlation between soil temperature during fires and presence of pre-burn fuels that affects both subsequent resprouting and germination and therefore affects plant distribution. Soil depth of seed and buds also determines which of these survive.

Though *Ceanothus* germination was promoted in the fire rings, soil temperatures still may have been outside the range of normal. Odion & Davis (2000) found occasional *Ceanothus* (*C. cuneatus var.* fascicularis) and *Arctostaphylos* seedlings in the highest fire intensity areas in their study. No other taxa survived to germinate or resprout at those temperatures. A study that experimentally increased fire intensity by adding brush to an old existing chaparral stand before burning, found that another species of *Ceanothus, C. greggii,* had increased germination with moderately increased fuel load, and similar germination to the control at the highest intensities (Moreno & Oechel 1991). On average, the other chaparral species present in that study showed decreased seedling production with increased fire intensity, but reactions varied by species. Without the diversity of fire intensity inherent in a fire through standing chaparral, species diversity is likely to be reduced.

The indicator species of the second year fire rings are mostly weedy and half of the species are exotic. The almost entirely different set of indicators for the associated HPB treatment plots demonstrates the radical difference between the environments of the fire rings and the surrounding treatment area. Many seeds in the seed bank were likely killed by high soil temperatures in the fire rings. The plants that inhabit the fire rings tend to be either fire tolerant species or efficiently dispersing colonizers. It is apparent that many exotic species do well as colonizers of these open sites. Several species that have been noted as having fire cue-stimulated germination were present in the vicinity of fire rings, including *Arctostaphylos viscida* (Fried et al. 2004), poison oak, *Ceanothus cuneatus, Gilia capitata* (bluehead gilia), *Stephanomeria virgata, Clarkia purpurea* (Keeley 1991), *Trifolium microcephalum* and *Juncus bufonius* (Odion 2000). For *Ceanothus* and *Stephanomeria,* the association with fire rings was particularly evident, as discussed previously. Other genera present in the vicinity of fire rings share species that have been noted as fire-recruiters, including *Calystegia, Lotus, Cryptantha, Gnaphalium, Galium, Collinsia* (Keeley 1991), and *Navarretia* (Odion 2000).

The changes in proportion of species by trait in Figures 23-26 give an interesting snapshot of fire ring colonization. By the second year, the distribution of life forms appears to have achieved similar proportions as those of the larger associated HPB plots. However, the distribution of other species attributes in the fire rings does not show the same convergence. It would be interesting to sample the fire rings over subsequent years and determine if and when they return to similar species composition as the surrounding area.

CONCLUSION

My research did not show a large effect of fuel reduction treatments on the existing plant community in the short term. Perhaps my relatively coarse abundance data were insufficiently detailed to capture changes that occurred. Alternatively, treatments may have caused little alteration of the community because of a history of disturbance and the already extensive occurrence of introduced species.

Other factors, such as presence of oak canopy, had a stronger influence on species composition than did treatment. Both oaks and *Ceanothus* provide important habitat for natives and perennials. While open areas were overrun by exotic annual grasses, they also supported several native annuals that are of special interest to the BLM.

Both treatments increased species richness initially, probably because they both caused disturbance and increased resource availability. The groups that showed increased abundance in response to treatment included the special status plants, annuals, forbs, exotics, and introduced weeds. The greatest effect was detected in the first year after treatment. By the second year species abundance was generally lower in the mastication treatment plots than in associated controls. In the HPB treatment, species abundance was lower in the second year than the first year, but was still higher than in their controls.

The effects of treatment on overall species composition were stronger for the HPB treatment than the mastication treatment. The evidence suggests that the primary factor responsible for the difference between the two treatments is the fire rings that remain after piles are burned in the HPB treatment. Though soil heating may be greater under the burn piles than it would be during a chaparral fire, this treatment introduces the element of fire into a community that is adapted to it. This treatment may allow fire-adapted species in the community to persist that would otherwise be lost.

It is likely that other factors beyond the presence of fire rings have contributed to the differential effects of the two treatments. Levels and distribution of soil disturbance and woody debris are markedly different in the two treatments, and these factors should influence species composition. However, the importance of such factors cannot be assessed based on the data that I collected.

With the evidence at hand it appears that neither fuel reduction treatment is a definite detriment to the plant community. Short term data suggest that the HPB treatment may lead to an increase in weedy and/or exotic species. At the same time, it may increase diversity by promoting species with fire-cued germination. In contrast, the mastication treatment appears to reduce species diversity. Both treatments tended to promote *Ceanothus* germination.

Based on these results, it is difficult to provide a clear recommendation concerning future management. Fortunately, the BLM is collecting additional data from these permanent plots in 2005, and perhaps in subsequent years. Further monitoring should increase our understanding of the effects of these fuel reduction treatments on plant communities. In the meantime, managers should be aware of the potential negative effects that are linked to either treatment. On the positive side, since many areas are left untreated, these treatments will increase overall heterogeneity in chaparral areas that would otherwise be more homogeneous because of fire exclusion.

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APPENDICES

Appendix 1. Species list for all plots sampled in this study, including species traits, known fire association, and identity in analysis. Taxa are sorted by life form and then alphabetically by scientific name. Vouchered taxa are listed in Appendix 3.

Taxon name: Introduced species are indicated in boldface type. Scientific names according to Hickman 1993.
Family: Plant family of taxon.
Ann. Per: Herbaceous plants are coded ***a*** = annual and ***p*** = perennial.
Type of Weed: Plants known to occur in disturbed habitats (fide Hickman 1993; USDA, NRCS 2004) are marked as weedy: ***nw*** = native weed, ***iw*** = introduced weed, or ***nox*** = Oregon state-listed noxious weeds.
Special Status: Plants of special status according to the Medford District BLM (M. Wineteer, pers. comm.) are noted using the following codes: **BAO** = Assessment, **BSO** = Sensitive, **BTO** = Tracking.
Fire Status: Known fire association (Fried et al. 2004; Keeley 1991; Odion 2000) is noted as: ***a*** = a fire-recruiting species, ***g*** = the genus contains some fire-recruiting species, ***r*** = fire resilient, germination is independent from fire.
Common Name: According to USDA, NRCS 2004.
Analyzed As: Column gives the identification used for taxa that were lumped with other species for analysis.

FORBS -- Taxon name	Family	Ann. Per.	Type of Weed	Special Status	Fire Status	Common Name	Analyzed As
Achillea millefolium L.	Asteraceae	p	nw			common yarrow	
Achyrachaena mollis Schauer	Asteraceae	a				blow wives	
Agoseris grandiflora (Nutt.) Greene	Asteraceae	p				bigflower agoseris	
Agoseris heterophylla (Nutt.) Greene	Asteraceae	a			r	annual agoseris	
Allium acuminatum Hook.	Liliaceae	p			r	tapertip onion	
Allium amplectens Torr.	Liliaceae	p			r	narrowleaf onion	
Amsinckia menziesii (Lehm.) A.Nels. & J.F.Macbr.	Boraginaceae	a	nw			Menzies' fiddleneck	
Ancistrocarphus filagineus Gray	Asteraceae	a				false neststraw	
***Anthemis cotula* L.**	Asteraceae	a	iw			stinking chamomile	
***Anthriscus caucalis* Bieb.**	Apiaceae	a				burr chervil	
Aphanes occidentalis (Nutt.) Rydb.	Rosaceae	a				field parsley piert	
***Arabidopsis thaliana* (L.) Heynh.**	Brassicaceae	a	iw			mouseear cress	
***Arenaria serpyllifolia* L.**	Caryophyllaceae	a	iw			thymeleaf sandwort	
Athysanus pusillus (Hook.) Greene	Brassicaceae	a				common sandweed	

Appendix 1 (Continued)

FORBS -- Taxon name	Family	Ann. Per.	Type of Weed	Special Status	Fire Status	Common Name	Analyzed As
Blepharipappus scaber Hook.	Asteraceae	a				rough eyelashweed	
Brodiaea elegans Hoover	Liliaceae	p			r	harvest brodiaea	
Calochortus tolmiei Hook. & Arn.	Liliaceae	p			r	Tolmie star-tulip	
Calycadenia truncata DC.	Asteraceae	a				Oregon western rosinweed	
Calystegia occidentalis (Gray) Brummitt	Convolvulaceae	p			g	chaparral false bindweed	
Camassia quamash (Pursh) Greene	Liliaceae	p				small camas	
Campanulaceae?	Campanulaceae?	a					
Cardamine oligosperma Nutt.	Brassicaceae	a				little western bittercress	
Castilleja attenuata (Gray) Chuang & Heckard	Scrophulariaceae	a				attenuate Indian paintbrush	
Castilleja tenuis (Heller) Chuang & Heckard	Scrophulariaceae	a				hairy Indian paintbrush	
Centaurea solstitialis **L.**	Asteraceae	a	iw / nox			yellow star-thistle	
Centaurium muehlenbergii (Griseb.) W.A.Wight ex Piper	Gentianaceae	a			g	Muhlenberg's centaury	
Cerastium glomeratum **Thuill.**	Caryophyllaceae	a	iw			sticky chickweed	
Chamaesyce serpyllifolia (Pers.) Small	Euphorbiaceae	a				thymeleaf sandmat	
Chamomilla suaveolens **(Pursh) Rydb.**	Asteraceae	a	iw			disc mayweed	
Cichorium intybus **L.**	Asteraceae	p	iw			chicory	
Clarkia gracilis (Piper) A.Nels. & J.F.Macbr.	Onagraceae	a				slender clarkia	
Clarkia purpurea ssp. *quadrivulnera* (Dougl. ex Lindl.) H.F.& M.E.Lewis	Onagraceae	a			a	four spot, winecup clarkia	
Clarkia rhomboidea Dougl. ex Hook.	Onagraceae	a				diamond clarkia	
Claytonia L.	Portulacaceae	a	nw			springbeauty	
Claytonia exigua Torr. & Gray	Portulacaceae	a	nw			serpentine springbeauty	*Claytonia*
Claytonia parviflora Dougl. ex Hook.	Portulacaceae	a	nw			streambank springbeauty	*Claytonia*
Claytonia perfoliata Donn ex Willd.	Portulacaceae	a	nw			miner's lettuce	*Claytonia*
Collinsia linearis Gray	Scrophulariaceae	a			g	narrowleaf blue eyed Mary	

Appendix 1 (Continued)

FORBS -- Taxon name	**Family**	**Ann. Per.**	**Type of Weed**	**Special Status**	**Fire Status**	**Common Name**	**Analyzed As**
Collinsia parviflora Lindl.	Scrophulariaceae	a			g	maiden blue eyed Mary	
Collinsia sparsiflora Fisch. & C.A. Mey.	Scrophulariaceae	a	nw		g	spinster's blue eyed Mary	
Collomia grandiflora Dougl. ex Lindl.	Polemoniaceae	a				grand collomia	
Conyza canadensis (L.) Cronq.	Asteraceae	a	nw			Canadian horseweed	
Cordylanthus tenuis Gray ssp. *viscidus* (T.J.Howell) Chuang & Heckard	Scrophulariaceae	a				slender bird's beak	
***Crepis pulchra* L.**	Asteraceae	a				smallflower hawksbeard	
Cryptantha Lehm. ex G. Don	Boraginaceae	a			g		
Cryptantha flaccida (Dougl. ex Lehm.) Greene	Boraginaceae	a			g	weakstem cryptantha	
Cryptantha torreyana (Gray) Greene	Boraginaceae	a			r	Torrey's cryptantha	*Cryptantha* spp
Cynoglossum grande Dougl. ex Lehm.	Boraginaceae	p				Pacific hound's tongue	
Daucus pusillus Michx.	Apiaceae	a			r	American wild carrot	
Delphinium nuttallianum Pritz. ex Walp.	Ranunculaceae	p	nw			twolobe larkspur	
***Dianthus armeria* L.**	Caryophyllaceae	a	iw			Deptford pink	
Dichelostemma Kunth	Liliaceae	p					
Dichelostemma capitatum (Benth.) Wood	Liliaceae	p				bluedicks	*Dichelostemma* spp.
Dichelostemma congestum (Sm.) Kunth	Liliaceae	p				ookow	*Dichelostemma*
Dodecatheon hendersonii Gray	Primulaceae	p				mosquito bills	
Draba verna L.	Brassicaceae	a	nw			spring draba	
Epilobium L.	Onagraceae	a	nw				
Epilobium brachycarpum K. Presl	Onagraceae	a	nw			tall annual willowherb	*Epilobium* spp.
Epilobium densiflorum (Lindl.) Hoch & Raven	Onagraceae	a				denseflower willowherb	
Epilobium foliosum (Torr. & Gray) Suksdorf	Onagraceae	a	nw				*Epilobium* spp.
Epilobium torreyi (S. Wats.) Hoch & Raven	Onagraceae	a				Torrey's willowherb	

Appendix 1 (Continued)

FORBS -- Taxon name	Family	Ann. Per.	Type of Weed	Special Status	Fire Status	Common Name	Analyzed As
Eremocarpus setigerus (Hook.) Benth.	Euphorbiaceae	a	nw			dove weed, turkey mullein	
Erigeron philadelphicus L.?	Asteraceae	p					
Eriophyllum lanatum (Pursh) Forbes	Asteraceae	p			g	common woolly sunflower	
***Erodium botrys* (Cav.) Bertol.**	Geraniaceae	a	iw			longbeak stork's bill	
***Erodium cicutarium* (L.) L'Hér. ex Ait.**	Geraniaceae	a	iw			redstem stork's bill	
Erythronium hendersonii S. Wats.	Liliaceae	p				Henderson's fawnlily	
Euphorbia spathulata Lam.	Euphorbiaceae	a	nw			warty spurge	
Fragaria vesca L.	Rosaceae	p				woodland strawberry	
Fragaria virginiana Duchesne	Rosaceae	p	nw			Virginia strawberry	
***Galium* L.**	Rubiaceae	a	iw		g		
Galium aparine L.	Rubiaceae	a	nw		g	stickywilly	
***Galium divaricatum* Pourret ex Lam.**	Rubiaceae	a	iw		g	Lamarck's bedstraw	*Galium* spp.
***Galium parisiense* L.**	Rubiaceae	a	iw		g	wall bedstraw	*Galium* spp.
Galium porrigens Dempster	Rubiaceae	p			g	graceful bedstraw	
Geranium L.	Geraniaceae	a					
Geranium carolinianum L.	Geraniaceae	a	nw			Carolina geranium	*Geranium* spp.
***Geranium dissectum* L.**	Geraniaceae	a	iw			cutleaf geranium	*Geranium* spp.
***Geranium molle* L.**	Geraniaceae	a	iw			dovefoot geranium	*Geranium* spp.
Gilia capitata Sims	Polemoniaceae				a	bluehead gilia	
Githopsis specularioides Nutt.	Campanulaceae	a				common bluecup	
Gnaphalium palustre Nutt.	Asteraceae	a	nw		g	western marsh cudweed	
Hemizonia congesta DC.	Asteraceae	a	nw			hayfield tarweed	
Hemizonia fitchii Gray	Asteraceae	a	nw			Fitch's tarweed	
Hesperolinon micranthum (Gray) Small	Linaceae	a				smallflower dwarf-flax	
Heterocodon rariflorum Nutt.	Campanulaceae	a				rareflower heterocodon	
Hieracium L.	Asteraceae	p				orange hawkweed	
Horkelia daucifolia (Greene) Rydb.	Rosaceae	p				carrotleaf horkelia	
***Hypericum perforatum* L.**	Clusiaceae	p	iw / nox			common St. Johnswort	

Appendix 1 (Continued)

FORBS -- Taxon name	Family	Ann. Per.	Type of Weed	Special Status	Fire Status	Common Name	Analyzed As
***Hypochaeris glabra* L.**	Asteraceae	a	iw			smooth catsear	
Idahoa scapigera (Hook.) A.Nels. & J.F.Macbr.	Brassicaceae	a				oldstem idahoa	
Iris chrysophylla T.J. Howell	Iridaceae	p				yellowleaf iris	
***Lactuca serriola* L.**	Asteraceae	a	iw			prickly lettuce	
Lactuca tatarica (L.) C.A. Mey. ssp. *pulchella* (Pursh) Stebbins	Asteraceae	p	nw			blue lettuce	
Lagophylla ramosissima Nutt.	Asteraceae	a				branched lagophylla	
Lilium columbianum hort. ex Baker	Liliaceae	p	nw			Columbian lily	
Limnanthes floccosa T.J. Howell ssp. *floccosa*	Limnanthaceae	a				woolly meadowfoam	
Linanthus bicolor (Nutt.) Greene	Polemoniaceae	a				true babystars	
Linanthus bolanderi (Gray) Greene	Polemoniaceae	a				Bolander's linanthus	
Lithophragma (Nutt.) Torr. & Gray	Saxifragaceae	p				woodland-star	
Lithophragma parviflorum (Hook.) Nutt. ex Torr. & Gray	Saxifragaceae	p				smallflower woodland-star	*Lithophragma* spp.
***Logfia arvensis* (L.) Holub**	Asteraceae	a				field cottonrose	
Lomatium nudicaule (Pursh) Coult. & Rose	Apiaceae	p				pestle lomatium	
Lomatium triternatum (Pursh) Coult. & Rose	Apiaceae	p				nineleaf biscuitroot	
Lomatium utriculatum (Nutt. ex Torr. & Gray) Coult. & Rose	Apiaceae	p				common lomatium	
Lotus L.	Fabaceae	a	nw		g		
Lotus humistratus Greene	Fabaceae	a	nw		g	foothill deervetch	
Lotus micranthus Benth.	Fabaceae	a	nw		g	desert deervetch	*Lotus* spp.
Lotus purshianus F.E. & E.G. Clem.	Fabaceae	a	nw		g	American bird's-foot trefoil	*Lotus* spp.
Lupinus bicolor Lindl.	Fabaceae	a	nw		g	miniature lupine	
Madia Molina	Asteraceae	a					

Appendix 1 (Continued)

FORBS -- Taxon name	Family	Ann. Per.	Type of Weed	Special Status	Fire Status	Common Name	Analyzed As
Madia citriodora Greene	Asteraceae	a				lemonscented madia	*Madia* spp.
Madia elegans D. Don ex Lindl.	Asteraceae	a				common madia	
Madia exigua (Sm.) Gray	Asteraceae	a				small tarweed	
Micropus californicus Fisch. & C.A.Mey.	Asteraceae	a				q tips	
Microseris laciniata (Hook.)Schultz-Bip. ssp. *detlingii* Chambers	Asteraceae	p		BSO?		Detling's silverpuffs	
Mimulus breviflorus Piper	Scrophulariaceae	a				shortflower monkeyflower	
Mimulus guttatus DC.	Scrophulariaceae	a				seep monkeyflower	
Mimulus pygmaeus A.L. Grant	Scrophulariaceae	a				Egg Lake monkeyflower	
Minuartia californica (Gray) Mattf.	Caryophyllaceae	a		BTO		California sandwort	
Minuartia douglasii (Fenzl ex Torr. & Gray) Mattf.	Caryophyllaceae	a				Douglas' stitchwort	
Moenchia erecta **(L.) P.G. Gaertn. et al.**	Caryophyllaceae	a	iw			upright chickweed	
Montia linearis (Dougl. ex Hook.) Greene	Portulacaceae	a	nw			narrowleaf minerslettuce	
Myosotis discolor **Pers.**	Boraginaceae	a	iw			changing forget-me-not	
Navarretia Ruiz & Pavòn	Polemoniaceae	a			g		
Navarretia divaricata (Torr. ex Gray) Greene	Polemoniaceae	a			g	divaricate navarretia	*Navarretia* spp.
Navarretia intertexta (Benth.) Hook	Polemoniaceae	a			g	needleleaf navarretia	*Navarretia* spp.
Navarretia pubescens (Benth.) Hook. & Arn.	Polemoniaceae	a			g	downy pincushionplant	*Navarretia* spp.
Navarretia subuligera Greene	Polemoniaceae	a		BTO	g	awl-leaf pincushionplant	*Navarretia* spp.
Navarretia tagetina Greene	Polemoniaceae	a		BTO	g	marigold pincushionplant	*Navarretia* spp.
Nemophila pedunculata Dougl. ex Benth.	Hydrophyllaceae	a	nw			littlefoot nemophila	
Orobanche uniflora L.	Orobanchaceae	a				oneflowered broomrape	

Appendix 1 (Continued)

FORBS -- Taxon name	Family	Ann. Per.	Type of Weed	Special Status	Fire Status	Common Name	Analyzed As
pappused composite	Asteraceae						
Pectocarya pusilla (A. DC.) Gray	Boraginaceae	a	nw			little combseed	
Penstemon roezlii Regel	Scrophulariaceae	p			g	Roezl's penstemon	
Pentagramma triangularis (Kaulfuss) Yatskievych et al.	Pteridaceae	p				goldback fern	
Perideridia oregana (S. Wats.) Mathias	Apiaceae	p				squaw potato	
***Petrorhagia* (Ser.) Link**	Caryophyllaceae	a	iw			pink	
Phacelia heterophylla Pursh	Hydrophyllaceae	p	nw		a	varileaf phacelia	
Phlox gracilis (Hook.) Greene	Polemoniaceae	a				slender phlox	
Piperia elegans (Lindl.) Rydb.	Orchidaceae	p				elegant piperia	
Plagiobothrys cognatus (Greene) I.M.Johnston	Boraginaceae	a				sleeping popcornflower	
Plagiobothrys glyptocarpus (Piper) I.M.Johnston	Boraginaceae	a		BAO		sculptured popcornflower	
Plagiobothrys greenei (Gray) I.M.Johnston	Boraginaceae	a		BAO		Greene's popcornflower	
Plagiobothrys nothofulvus (Gray) Gray	Boraginaceae	p				rusty popcornflower	
Plagiobothrys tenellus (Nutt. ex Hook.) Gray	Boraginaceae	a				Pacific popcornflower	
***Plantago lanceolata* L.**	Plantaginaceae	p	iw			narrowleaf plantain	
Plectritis congesta (Lindl.) DC.	Valerianaceae	a				shortspur seablush	
Polygonum L.	Polygonaceae	a					
Polygonum californicum Meisn.	Polygonaceae	a				California knotweed	*Polygonum* spp.
Polygonum douglasii Greene	Polygonaceae	a				Douglas' knotweed	*Polygonum* spp.
Polygonum parryi Greene	Polygonaceae	a				Parry's knotweed	
Polygonum polygaloides Wallich ex Meisn.	Polygonaceae	a				milkwort knotweed	*Polygonum* spp.
Potentilla glandulosa Lindl.	Rosaceae	p				sticky cinquefoil	

Appendix 1 (Continued)

FORBS -- Taxon name	Family	Ann. Per.	Type of Weed	Special Status	Fire Status	Common Name	Analyzed As
Prunella vulgaris L. var. *lanceolata* (W.Bart.) Fern.	Lamiaceae	p				lance selfheal	
Psilocarphus brevissimus Nutt.	Asteraceae	a				short woollyheads	
***Ranunculus arvensis* L.**	Ranunculaceae	a	iw			corn buttercup	
Ranunculus austrooreganus L. Benson	Ranunculaceae	p		BSO		Southern Oregon buttercup	*Ranunculus occidentalis*
Ranunculus hebecarpus Hook. & Arn.	Ranunculaceae	a				delicate buttercup	
Ranunculus occidentalis Nutt.	Ranunculaceae	p				western buttercup	
Rigiopappus leptocladus Gray	Asteraceae	a				wireweed	
***Rumex crispus* L.**	Polygonaceae	p	iw			curly dock	
Sanguisorba occidentalis Nutt.	Rosaceae	a	nw			western burnet	
Sanicula bipinnatifida Dougl. ex Hook.	Apiaceae	p				purple sanicle	
Sanicula crassicaulis Poepp. ex DC.	Apiaceae	p				Pacific blacksnakeroot	
Saxifraga integrifolia Hook.	Saxifragaceae	p				wholeleaf saxifrage	
***Scleranthus annuus* L.**	Caryophyllaceae	a	iw			German knotgrass	
Sclerolinon digynum (Gray) Rogers	Linaceae	a				digynum flax	
Scrophulariaceae indet.	Scrophulariaceae						
Scutellaria antirrhinoides Benth.	Lamiaceae	p				nose skullcap	
***Sherardia arvensis* L.**	Rubiaceae	a	iw			blue fieldmadder	
Silene antirrhina L.	Caryophyllaceae	a			g	sleepy silene	
Silene campanulata S. Wats.	Caryophyllaceae	p			g	Red Mountain catchfly	
Sisyrinchium bellum S. Wats.	Iridaceae	p				western blue-eyed grass	
***Sonchus asper* (L.) Hill**	Asteraceae	a	iw			spiny sowthistle	
Stellaria nitens Nutt.	Caryophyllaceae	a	nw			shiny chickweed	
Stephanomeria virgata Benth.	Asteraceae	a			a	rod wirelettuce	
sterile comp 1	Asteraceae						
sterile comp 2	Asteraceae						
Taraxacum G.H. Weber ex Wiggers?	Asteraceae						

Appendix 1 (Continued)

FORBS -- Taxon name	**Family**	**Ann. Per.**	**Type of Weed**	**Special Status**	**Fire Status**	**Common Name**	**Analyzed As**
Thysanocarpus curvipes Hook.	Brassicaceae	a				sand fringepod	
Tonella tenella (Benth.) Heller	Scrophulariaceae	a				lesser baby innocence	
***Torilis arvensis* (Huds.) Link**	Apiaceae	a	iw			spreading hedgeparsley	
***Tragopogon dubius* Scop.**	Asteraceae	a	iw			yellow salsify	
Trichostema lanceolatum Benth.	Lamiaceae	a	nw			vinegarweed	
Trifolium albopurpureum Torr. & Gray	Fabaceae	a	nw			rancheria clover	
Trifolium bifidum Gray	Fabaceae	a				notchleaf clover	
Trifolium ciliolatum Benth.	Fabaceae	a	nw		a	foothill clover	
***Trifolium dubium* Sibthorp**	Fabaceae	a	iw			suckling clover	
Trifolium eriocephalum Nutt.	Fabaceae	p				woollyhead clover	
***Trifolium hybridum* L.**	Fabaceae		iw			alsike clover	
Trifolium microcephalum Pursh	Fabaceae	a	nw		a	smallhead clover	
Trifolium willdenowii Spreng.	Fabaceae	a	nw			tomcat clover	
Triodanis perfoliata (L.) Nieuwl.	Campanulaceae	a	nw			Venus' looking-glass	
Triteleia hyacinthina (Lindl.) Greene	Liliaceae	p				white brodiaea	
unknown scapose							
Uropappus lindleyi (DC.) Nutt.	Asteraceae	a				Lindley's silverpuffs	
***Valerianella locusta* (L.) Lat.**	Valerianaceae	a				Lewiston cornsalad	
***Verbascum blattaria* L.**	Scrophulariaceae		iw			moth mullein	
Veronica L.	Scrophulariaceae	a					
***Veronica arvensis* L.**	Scrophulariaceae	a	iw			corn speedwell	*Veronica* spp
Veronica peregrina L.	Scrophulariaceae	a	nw			neckweed	*Veronica* spp
***Veronica persica* Poir.**	Scrophulariaceae	a	iw			birdeye speedwell	
***Vicia sativa* L.**	Fabaceae	a	iw			garden vetch	
Viola sheltonii Torr.	Violaceae	p				Shelton's violet	
Yabea microcarpa (Hook. & Arn.) K.-Pol.	Apiaceae	a				false carrot	
Zigadenus venenosus S. Wats.	Liliaceae	p	nw			meadow deathcamas	

Appendix 1 (Continued)

GRAMINOIDS -- Taxon name	**Family**	**Ann. Per.**	**Type of Weed**	**Special Status**	**Fire Status**	**Common Name**	**Analyzed As**
Achnatherum lemmonii (Swallen) Barkworth	Poaceae	p				Lemmon's needlegrass	
***Aira caryophyllea* L.**	Poaceae	a	iw			silver hairgrass	
***Apera interrupta* (L.) Beauv.**	Poaceae	a	iw				
***Avena fatua* L.**	Poaceae	a				wild oat	
***Briza minor* L.**	Poaceae	a				little quakinggrass	
Bromus carinatus Hook. & Arn.	Poaceae	p	nw			California brome	
***Bromus diandrus* Roth**	Poaceae	a	iw			ripgut brome	
***Bromus hordeaceus* L.**	Poaceae	a	iw			soft brome	
***Bromus japonicus* Thunb. ex Murr.**	Poaceae	a	iw			Japanese brome	
Bromus laevipes Shear	Poaceae	p				Chinook brome	
***Bromus madritensis* L.**	Poaceae	a	iw			compact brome	
***Bromus sterilis* L.**	Poaceae	a	iw			poverty brome	
***Bromus tectorum* L.**	Poaceae	a	iw			cheatgrass	
***Cynosurus echinatus* L.**	Poaceae	a	iw			bristly dogstail grass	
Danthonia californica Boland. var. *americana* (Scribn.) A.S.Hitchc.	Poaceae	p				California oatgrass	*D. californica*
Danthonia californica Boland. var. *californica*	Poaceae	p				California oatgrass	*D. californica*
Danthonia unispicata (Thurb.) Munro ex Macoun	Poaceae	p				onespike danthonia	
Deschampsia danthonioides (Trin.) Munro	Poaceae	a				annual hairgrass	
Elymus L.	Poaceae	p					
Elymus elymoides (Raf.) Swezey	Poaceae	p				squirreltail	
Elymus glaucus Buckl.	Poaceae	p			r	blue wildrye	*Elymus* spp.
Elymus trachycaulus (Link) Gould ex Shinners	Poaceae	p				slender wheatgrass	*Elymus* spp.
Festuca idahoensis Elmer	Poaceae	p				Idaho fescue	

Appendix 1 (Continued)

GRAMINOIDS -- Taxon name	**Family**	**Ann. Per.**	**Type of Weed**	**Special Status**	**Fire Status**	**Common Name**	**Analyzed As**
***Gastridium ventricosum* (Gouan) Schinz & Thellung**	Poaceae	a	iw			nit grass	
***Hordeum marinum* Huds. ssp. *gussonianum* (Parl.) Thellung**	Poaceae	a	iw			Mediterranean barley	
Juncus L.	Juncaceae	p				large rush	
Juncus bufonius L.	Juncaceae	a	nw		a	toad rush	
Juncus hemiendytus F.J. Herm. var. *hemiendytus*	Juncaceae	a				Herman's dwarf rush	an unknown annual forb
Koeleria macrantha (Ledeb.) J.A.Schultes	Poaceae	p				prairie Junegrass	
***Lolium multiflorum* Lam.**	Poaceae	a	iw			Italian ryegrass	
Luzula comosa E. Mey.	Juncaceae	p				Pacific woodrush	
Melica harfordii Boland.	Poaceae	p				Harford's oniongrass	
Panicum capillare L.	Poaceae	a	nw			witchgrass	
***Poa bulbosa* L.**	Poaceae	p	iw			bulbous bluegrass	
***Poa compressa* L.**	Poaceae	p	iw			Canada bluegrass	
***Poa pratensis* L.**	Poaceae	p	iw			Kentucky bluegrass	
Poa secunda J. Presl	Poaceae	p				Sandberg bluegrass	
Pseudoroegneria spicata (Pursh) A.Löve	Poaceae	p					*Elymus* spp.
Scribneria bolanderi (Thurb.) Hack.	Poaceae	a	nw	BTO		Scribner's grass	
sterile branched grass	Poaceae						
***Taeniatherum caput-medusae* (L.) Nevski**	Poaceae	a	iw / nox			medusahead	
Vulpia bromoides (L.) S.F. Gray	Poaceae	a	iw			brome fescue	included with *V.microstachys*
Vulpia microstachys (Nutt.) Munro	Poaceae	a	nw			small fescue	
***Vulpia myuros* (L.) K.C. Gmel.**	Poaceae	a	iw			rat-tail fescue	

Appendix 1 (Continued)

SHRUBS -- Taxon name	Family	Ann. Per.	Type of Weed	Special Status	Fire Status	Common Name	Analyzed As
Amelanchier alnifolia (Nutt.) Nutt. ex M. Roemer	Rosaceae					Saskatoon serviceberry	
Arctostaphylos viscida Parry	Ericaceae				a	sticky whiteleaf manzanita	
Berberis aquifolium Pursh	Berberidaceae					hollyleaved barberry	
Ceanothus cuneatus (Hook.) Nutt.	Rhamnaceae				a	buckbrush	
Cercocarpus betuloides Nutt.	Rosaceae				r	mountain mahogany	
Lonicera interrupta Benth.	Caprifoliaceae					chaparral honeysuckle	
Prunus subcordata Benth.	Rosaceae					Klamath plum	
Rosa californica Cham. & Schlecht.	Rosaceae					California wildrose	
Symphoricarpos albus (L.) Blake	Caprifoliaceae					common snowberry	
Toxicodendron diversilobum (Torr. & Gray) Greene	Anacardiaceae				a	Pacific poison oak	

TREES -- Taxon name	Family	Ann. Per.	Type of Weed	Special Status	Fire Status	Common Name	Analyzed As
Arbutus menziesii Pursh	Ericaceae					Pacific madrone	
Calocedrus decurrens (Torr.) Florin	Cupressaceae					incense cedar	
Pinus ponderosa P.& C. Lawson	Pinaceae					ponderosa pine	
Quercus garryana Dougl. ex Hook.	Fagaceae					Oregon white oak	
Quercus kelloggii Newberry	Fagaceae					California black oak	

Appendix 2. Indicator values for species (percentage of perfect indication) for the two groups of plots defined by percent tree canopy (<10% versus >10%) in 2004. Statistically significant indicator species are listed here ($p < 0.05$). Non-native species appear in boldface type. The most significant indicators were also listed in Table 9.

Less than 10% tree canopy	IV	Greater than 10% tree canopy	IV
Castilleja tenuis	47.2	*Agoseris grandiflora*	60.9
Claytonia spp.	40.6	*Allium acuminatum*	41.5
Clarkia gracilis	61.1	*Bromus carinatus*	56.5
Epilobium torreyi	50.2	*Bromus laevipes*	30.0
Erodium cicutarium	49.1	*Calochortus tolmiei*	67.7
Linanthus bicolor	54.8	*Centaurium muehlenbergii*	51.6
Lomatium utriculatum	58.6	*Collomia grandiflora*	60.0
Myosotis discolor	38.5	***Crepis pulchra***	58.4
Pectocarya pusilla	57.9	***Cynosurus echinatus***	57.9
Plagiobothrys cognatus	30.3	*Danthonia californica*	57.1
Psilocarphus brevissimus	29.3	*Dodecatheon hendersonii*	26.6
Sanguisorba occidentalis	43.7	*Elymus spp.*	55.3
Scleranthus annuus	60.2	*Erythronium hendersonii*	26.1
Scribneria bolanderi	27.5	*Euphorbia spathulata*	42.0
Trichostema lanceolatum	45.3	*Festuca idahoensis*	64.5
Trifolium albopurpureum	42.9	*Galium porrigens*	62.8
Yabea microcarpa	55.5	*Hesperolinon micranthum*	37.0
		Horkelia daucifolia	52.9
		Iris chrysophylla	39.1
		Koeleria macrantha	63.9
		Lomatium nudicaule	17.4
		Lonicera interrupta	65.0
		Luzula comosa	75.2
		Microseris laciniata ssp. detlingii	17.4
		Orobanche uniflora	23.4
		Penstemon roezlii	29.0
		Piperia elegans	17.4
		Poa pratensis	40.4
		Prunella vulgaris var. *lanceolata*	20.6
		Ranunculus occidentalis	72.2
		Sanicula crassicaulis	54.4
		Sisyrinchium bellum	17.4
		Torilis arvensis	67.4
		Toxicodendron diversilobum	63.8
		Tragopogon dubius	33.9
		Zigadenus venenosus	39.9

Appendix 3. List of taxa from my study sites with vouchers in the Oregon State University herbarium (OSC), in alphabetical order by taxon name. Non-native species appear in boldface type. Numbers listed are my collection numbers. For example, Sikes 45A is a specimen of *Aira caryophyllea*.

Taxon name	Number	Family
Achnatherum lemmonii (Swallen) Barkworth	148	Poaceae
Achyrachaena mollis Schauer	118	Asteraceae
Agoseris heterophylla (Nutt.) Greene	116	Asteraceae
***Aira caryophyllea* L.**	45A	Poaceae
Allium acuminatum Hook.	129	Liliaceae
Allium amplectens Torr.	123A	Liliaceae
Amsinckia menziesii (Lehm.) A.Nels. & J.F.Macbr.	102, 103	Boraginaceae
***Anthemis cotula* L.**	142	Asteraceae
***Anthriscus caucalis* Bieb.**	136	Apiaceae
***Apera interrupta* (L.) Beauv.**	153	Poaceae
Aphanes occidentalis (Nutt.) Rydb.	180	Rosaceae
***Arenaria serpyllifolia* L.**	124A	Caryophyllaceae
Athysanus pusillus (Hook.) Greene	177	Brassicaceae
***Avena fatua* L.**	169	Poaceae
***Briza minor* L.**	143	Poaceae
Brodiaea elegans Hoover	45E	Liliaceae
Bromus carinatus Hook. & Arn.	144	Poaceae
***Bromus diandrus* Roth**	116A	Poaceae
***Bromus hordeaceus* L.**	110	Poaceae
***Bromus japonicus* Thunb. ex Murr.**	196	Poaceae
Bromus laevipes Shear	197	Poaceae
***Bromus madritensis* L.**	191	Poaceae
***Bromus sterilis* L.**	113	Poaceae
***Bromus tectorum* L.**	116B	Poaceae
Calycadenia truncata DC.	50	Asteraceae
Calystegia occidentalis (Gray) Brummitt	228	Convolvulaceae
Camassia quamash (Pursh) Greene	45D	Liliaceae
Cardamine oligosperma Nutt.	85	Brassicaceae
Castilleja attenuata (Gray) Chuang & Heckard	98	Scrophulariaceae
***Centaurea solstitialis* L.**	52	Asteraceae
***Cerastium glomeratum* Thuill.**	86, 96	Caryophyllaceae
Chamaesyce serpyllifolia (Pers.) Small	64	Euphorbiaceae
***Cichorium intybus* L.**	226	Asteraceae
Clarkia gracilis (Piper) A.Nels. & J.F.Macbr.	114	Onagraceae
Clarkia purpurea ssp. *quadrivulnera* (Dougl. ex Lindl.) H.F.& M.E.Lewis	117	Onagraceae
Clarkia rhomboidea Dougl. ex Hook.	116C	Onagraceae
Claytonia exigua Torr. & Gray	126A	Portulacaceae
Claytonia parviflora Dougl. ex Hook.	95A	Portulacaceae
Claytonia perfoliata Donn ex Willd.	151	Portulacaceae
Collinsia linearis Gray	83	Scrophulariaceae
Collinsia parviflora Lindl.	81	Scrophulariaceae

Appendix 3 (Continued)

Taxon name	**Number**	**Family**
Collinsia sparsiflora Fisch. & C.A. Mey.	97	Scrophulariaceae
Conyza canadensis (L.) Cronq.	57	Asteraceae
Cordylanthus tenuis Gray ssp. *viscidus* (T.J.Howell) Chuang & Heckard	130	Scrophulariaceae
***Crepis pulchra* L.**	186	Asteraceae
Cryptantha flaccida (Dougl. ex Lehm.) Greene	188	Boraginaceae
***Cynosurus echinatus* L.**	137	Poaceae
Danthonia unispicata (Thurb.) Munro ex Macoun	170	Poaceae
Daucus pusillus Michx.	45C	Apiaceae
Delphinium nuttallianum Pritz. ex Walp.	82	Ranunculaceae
***Dianthus armeria* L.**	171	Caryophyllaceae
Dichelostemma congestum (Sm.) Kunth	122	Liliaceae
Elymus elymoides (Raf.) Swezey	178	Poaceae
Elymus glaucus Buckl.	199	Poaceae
Elymus trachycaulus (Link) Gould ex Shinners	231	Poaceae
Epilobium brachycarpum K. Presl	49, 67	Onagraceae
Epilobium densiflorum (Lindl.) Hoch & Raven	53	Onagraceae
Epilobium torreyi (S. Wats.) Hoch & Raven	152	Onagraceae
Eremocarpus setigerus (Hook.) Benth.	51	Euphorbiaceae
Erigeron philadelphicus L.?	172	Asteraceae
***Erodium botrys* (Cav.) Bertol.**	69	Geraniaceae
***Erodium cicutarium* (L.) L'Hér. ex Ait.**	68	Geraniaceae
Euphorbia spathulata Lam.	145	Euphorbiaceae
***Galium divaricatum* Pourret ex Lam.**	210	Rubiaceae
***Galium parisiense* L.**	211	Rubiaceae
Galium porrigens Dempster	139	Rubiaceae
***Gastridium ventricosum* (Gouan) Schinz & Thellung**	192	Poaceae
Geranium carolinianum L.	208	Geraniaceae
***Geranium dissectum* L.**	181	Geraniaceae
***Geranium molle* L.**	71	Geraniaceae
Gnaphalium palustre Nutt.	173	Asteraceae
Hemizonia congesta DC.	54	Asteraceae
Hemizonia fitchii Gray	46	Asteraceae
Hesperolinon micranthum (Gray) Small	225	Linaceae
Heterocodon rariflorum Nutt.	200	Campanulaceae
***Hordeum marinum* Huds. ssp. *gussonianum* (Parl.) Thellung**	182	Poaceae
Horkelia daucifolia (Greene) Rydb.	203	Rosaceae
***Hypericum perforatum* L.**	60	Clusiaceae
***Hypochaeris glabra* L.**	120	Asteraceae
Iris chrysophylla T.J. Howell	105	Iridaceae
Juncus bufonius L.	117A	Juncaceae
Juncus hemiendytus F.J. Herm. var. *hemiendytus*	163	Juncaceae
Juncus L.	126	Juncaceae
***Lactuca serriola* L.**	55	Asteraceae

Appendix 3 (Continued)

Taxon name	Number	Family
Lactuca tatarica (L.) C.A. Mey. ssp. *pulchella* (Pursh) Stebbins	58	Asteraceae
Lagophylla ramosissima Nutt.	47, 115	Asteraceae
Linanthus bicolor (Nutt.) Greene	70	Polemoniaceae
Linanthus bolanderi (Gray) Greene	229	Polemoniaceae
Lithophragma parviflorum (Hook.) Nutt. ex Torr. & Gray	78	Saxifragaceae
***Logfia arvensis* (L.) Holub**	227	Asteraceae
***Lolium multiflorum* Lam.**	62	Poaceae
Lomatium triternatum (Pursh) Coult. & Rose	204	Apiaceae
Lonicera interrupta Benth.	124B	Caprifoliaceae
Lotus humistratus Greene	131	Fabaceae
Lupinus bicolor Lindl.	76	Fabaceae
Luzula comosa E. Mey.	117B	Juncaceae
Madia citriodora Greene	183	Asteraceae
Madia elegans D. Don ex Lindl.	121	Asteraceae
Madia Molina	112, 119	Asteraceae
Microseris laciniata (Hook.)Schultz-Bip. ssp. *detlingii* Chambers	212, 218	Asteraceae
Mimulus breviflorus Piper	125	Scrophulariaceae
Mimulus guttatus DC.	61	Scrophulariaceae
Mimulus pygmaeus A.L. Grant	160	Scrophulariaceae
Minuartia californica (Gray) Mattf.	123C	Caryophyllaceae
***Moenchia erecta* (L.) P.G. Gaertn. et al.**	92	Caryophyllaceae
Montia linearis (Dougl. ex Hook.) Greene	164	Portulacaceae
***Myosotis discolor* Pers.**	117C	Boraginaceae
Navarretia divaricata (Torr. ex Gray) Greene ssp. *vividior* (Jepson & V. Bailey) Mason	187	Polemoniaceae
Navarretia subuligera Greene	129D	Polemoniaceae
Navarretia tagetina Greene	179	Polemoniaceae
Orobanche uniflora L.	93	Orobanchaceae
Panicum capillare L.	232	Poaceae
Pectocarya pusilla (A. DC.) Gray	84	Boraginaceae
Pentagramma triangularis (Kaulfuss) Yatskievych et al.	168	Pteridaceae
***Petrorhagia* (Ser.) Link**	59	Caryophyllaceae
Phlox gracilis (Hook.) Greene	79	Polemoniaceae
Piperia elegans (Lindl.) Rydb.	216	Orchidaceae
Plagiobothrys cognatus (Greene) I.M.Johnston	75	Boraginaceae
Plagiobothrys glyptocarpus (Piper) I.M.Johnston	146	Boraginaceae
Plagiobothrys greenei (Gray) I.M.Johnston	116F	Boraginaceae
Plagiobothrys nothofulvus (Gray) Gray	72	Boraginaceae
Plagiobothrys tenellus (Nutt. ex Hook.) Gray	73, 94	Boraginaceae
***Plantago lanceolata* L.**	220	Plantaginaceae
Plectritis congesta (Lindl.) DC.	88	Valerianaceae
***Poa compressa* L.**	219	Poaceae
***Poa pratensis* L.**	126B	Poaceae
Polygonum californicum Meisn.	129E	Polygonaceae
Polygonum douglasii Greene	66	Polygonaceae

Appendix 3 (Continued)

Taxon name	**Number**	**Family**
Polygonum parryi Greene	126C	Polygonaceae
Prunella vulgaris L. var. *lanceolata* (W.Bart.) Fern.	213	Lamiaceae
Pseudoroegneria spicata (Pursh) A.Löve	194	Poaceae
Ranunculus austrooreganus L. Benson	74	Ranunculaceae
Ranunculus hebecarpus Hook. & Arn.	117F	Ranunculaceae
Ranunculus occidentalis Nutt.	74A	Ranunculaceae
Rigiopappus leptocladus Gray	124	Asteraceae
***Rumex crispus* L.**	63	Polygonaceae
Sanguisorba occidentalis Nutt.	133	Rosaceae
Sanicula bipinnatifida Dougl. ex Hook.	209	Apiaceae
Saxifraga integrifolia Hook.	89	Saxifragaceae
Sclerolinon digynum (Gray) Rogers	166	Linaceae
Scribneria bolanderi (Thurb.) Hack.	158	Poaceae
Scutellaria antirrhinoides Benth.	65	Lamiaceae
***Sherardia arvensis* L.**	221	Rubiaceae
Silene antirrhina L.	195	Caryophyllaceae
Sisyrinchium bellum S. Wats.	217	Iridaceae
Stellaria nitens Nutt.	99	Caryophyllaceae
Stephanomeria virgata Benth.	224	Asteraceae
***Taeniatherum caput-medusae* (L.) Nevski**	134	Poaceae
Thysanocarpus curvipes Hook.	116G	Brassicaceae
Tonella tenella (Benth.) Heller	80	Scrophulariaceae
***Torilis arvensis* (Huds.) Link**	135	Apiaceae
***Tragopogon dubius* Scop.**	201	Asteraceae
Trichostema lanceolatum Benth.	48	Lamiaceae
Trifolium albopurpureum Torr. & Gray	77	Fabaceae
Trifolium bifidum Gray	106	Fabaceae
Trifolium ciliolatum Benth.	108	Fabaceae
***Trifolium dubium* Sibthorp**	117G	Fabaceae
***Trifolium hybridum* L.**	147	Fabaceae
Trifolium microcephalum Pursh	202	Fabaceae
Trifolium willdenowii Spreng.	167	Fabaceae
Triodanis perfoliata (L.) Nieuwl.	222	Campanulaceae
Triteleia hyacinthina (Lindl.) Greene	123D	Liliaceae
Uropappus lindleyi (DC.) Nutt.	111	Asteraceae
***Valerianella locusta* (L.) Lat.**	90	Valerianaceae
***Verbascum blattaria* L.**	56	Scrophulariaceae
***Veronica arvensis* L.**	157	Scrophulariaceae
Vulpia bromoides (L.) S.F. Gray	126D	Poaceae
***Vulpia myuros* (L.) K.C. Gmel.**	174	Poaceae
Yabea microcarpa (Hook. & Arn.) K.-Pol.	104	Apiaceae

Appendix 4. Contents of data CD.

NAME	FORMAT	CONTENTS
2003 plot photos	Folder of .jpg photos	110 photos taken in 2003
2004 plot photos	Folder of .jpg photos	210 photos taken in 2004
Photo Names.doc	Microsoft Word 2003	text explains naming convention for plot photos
Photo Names.rtf	rich text format	text explains naming convention for plot photos
plots physical.xls	Microsoft Excel 2003	spreadsheet gives location information and physical data for each plot
plots physical.csv	comma delimited format	spreadsheet gives location information and physical data for each plot
key to plots physical. doc	Microsoft Word 2003	text explains contents of **plots physical** spreadsheet
key to plots physical. rtf	rich text format	text explains contents of **plots physical** spreadsheet
spp comp.xls	Microsoft Excel 2003	spreadsheet gives the abundance code data for all species by plot
spp comp.csv	comma delimited format	spreadsheet gives the abundance code data for all species by plot
spp codes used.xls	Microsoft Excel 2003	spreadsheet lists full scientific name for each species code used in data files
spp codes used.csv	comma delimited format	spreadsheet lists full scientific name for each species code used in data files
all traits.xls	Microsoft Excel 2003	spreadsheet gives trait assignments for each species
all traits.csv	comma delimited format	spreadsheet gives trait assignments for each species
fire ring physical. xls	Microsoft Excel 2003	spreadsheet gives location information and physical data for each fire ring
fire ring physical.csv	comma delimited format	spreadsheet gives location information and physical data for each fire ring
key to fire ring physical.doc	Microsoft Word 2003	text explains contents of **fire ring physical** spreadsheet
key to fire ring physical.rtf	rich text format	text explains contents of **fire ring physical** spreadsheet
spp in fire rings.xls	Microsoft Excel 2003	spreadsheet gives the abundance code data for all species by fire ring plot
spp in fire rings.csv	comma delimited format	spreadsheet gives the abundance code data for all species by fire ring plot

The following corrections are included in the above files but were not included in the results reported in this thesis:

1. *Vulpia bromoides* was added to the data. A sample of this species was collected but lumped with the *V. microstachys* that also occurred in the plot. It was not correctly identified until after analysis occurred.

Appendix 4 (Continued)

2. *Ancistrocarphus filagineus* (species code "shortstar") was identified as a native species. During the analysis it was coded only as an annual forb.

3. *Juncus hemiendytus* (species code "smallest") was identified as a native annual graminoid. During the analysis it was coded as an annual forb.

4. A sample that I had identified as *Madia citriodora* representing 2 plants in plot LBB153-1T was corrected to *M. elegans*. Therefore an abundance code of "1" was added to the MAEL column in "spp comp.xls".