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A COMPARISON OF THE SHORT-TERM EFFECTS OF TWO FUEL TREATMENTS ON CHAPARRAL COMMUNITIES IN SOUTHWEST OREGON

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ABSTRACT

Fuel treatments to reduce fire danger are being applied to public lands throughout the western United States, affecting significant acreage at considerable expense. This study compares the short term effects of two fuel treatment methods, shrub mastication and "hand cut, pile, and burn" (HPB), on chaparral communities in southwestern Oregon. *Ceanothus cuneatus* dominated the study sites where permanent paired plots were established on either side of treatment-control boundaries. Over a two year period, we recorded all vascular plant species within each treatment or control plot, along with an abundance class for each species. The effects of treatment on species composition and abundance of forbs and graminoids, overall as well as by plant trait group, were surprisingly small. Time since treatment, 1 yr or 2 yr, had a stronger effect on species composition than did treatment method. Species abundance and richness were greater in the first year after treatments than in the second year or in controls. In the second year, after both types of treatments, species abundance and richness were reduced, while after mastication treatment, these measures were lower than in control areas. The HPB treatment had a greater effect on plant communities than did mastication, due, at least in part, to the presence of fire rings from burned piles. Compared to their surrounding treated plots, fire rings had greater proportions of both annuals (95% versus 71%) and introduced weeds (35% versus 21%) in the second year after treatment. *Ceanothus* germination was stimulated in fire rings but also occurred in most plots, including controls. Both types of treated areas had more *Ceanothus* seedlings than their controls. Short term evidence suggests that the HPB treatment may lead to an increase in weedy and exotic species and the mastication treatment may reduce species diversity. The HPB treatment may also increase native species diversity by allowing fire-cued species to establish. Additional monitoring over time is needed to assess longer term treatment consequences for these northern chaparral communities.

Key Words: Burn piles, *Ceanothus cuneatus*, chaparral, fire management, fuel reduction, shrub mastication.

Fuel 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 some forests and shrublands, potentially increasing spread and severity of fires when they occur. Recent large-scale wildfires and the expansion of housing in proximity to wildlands have resulted in loss of property and increased motivation to prevent further losses (Keeley 2002). The primary purpose of fuel treatment is to remove, reduce, or alter combustible plant material that can act as fuel for a fire, thus reducing wildfire risks. Other potential benefits include rejuvenating senescent shrubs, increasing forage for wildlife (Lillywhite 1977; Rogers et al. 2004), and improving conditions for forbs and grasses. Risks associated with fuel treatments include invasion or expansion of exotic plant species (e.g., Merriam et al. 2006; Perchemlides et al. 2008) and loss or reduction of native species caused by depleted or treatment-damaged seed banks, mycorrhizal communities, or resprouting success (e.g., Smith et al. 2004; Busse et al. 2005; LeFer and Parker 2005; Knapp et al. 2007). These potentially negative effects depend on the ecosystem being treated, fuel loads, historical fire regimes, recent fire-free intervals, and method of treatment (e.g., Keeley and Fotheringham 2001a; Veblen 2003; Knapp et al. 2007).

Mechanical methods of fuel treatment have become more common because of risks associated with prescribed fire and the effects of smoke on air quality in populated areas. Mechanical treatments are being used in fire-adapted plant communities, such as chaparral, though consequences for the ecosystems are not well understood (Knapp et al. 2007). Chaparral most commonly burns in stand-replacing fires that do not leave records such as fire scars, resulting in uncertainty about past fire regimes (Keeley and Fotheringham 2001b). Estimates of a fire return interval for chaparral in the pre-industrial United States range from 20–40 yr (Leenhouts 1998) to 50–80 yr (Zedler 1995). A range of regeneration strategies coexists in this diverse plant community, making chaparral resilient to fire and other

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disturbances (Lavorel 1999; Keeley et al. 2005b). While some chaparral plant species are long-lived (Keeley et al. 2005a), many have relatively short life spans, and are dependent on fire for reproduction or the creation of sites for establishment. Replacing fire with mechanical fuel treatment is likely to affect the regeneration of some species and the community in general.

Chaparral is found in regions with the hot dry summers and cool wet winters of a Mediterranean climate, and is often dominated by a single shrub species with small, evergreen leaves. *Ceanothus cuneatus* (Hook.) Nutt. (Rhamnaceae) 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 the most xeric vegetation type in southwestern Oregon, with *C. cuneatus* being the most drought and heat tolerant of the area's woody dominants (Detling 1961). In the Rogue Valley, *Ceanothus* chaparral is usually found adjacent to the more mesic habitat of *Quercus garryana* Dougl. ex Hook. (Fagaceae) oak woodlands, while *Arctostaphylos viscida* Parry (Ericaceae) is found overlapping the two adjacent zones (Detling 1961). *Ceanothus* stands are often interspersed with grass openings.

The Medford District of the Bureau of Land Management (BLM) in southwestern Oregon manages a checkerboard of public lands intermixed with private, inhabited lands. In recent years, the BLM has allotted substantial funds to treating fuels in both chaparral and oak woodlands using two methods: mastication and hand cut, pile, and burn (HPB). Mastication uses heavy machinery on caterpillar treads that is equipped with a rotating brush cutting disk on a moveable arm. The machine can shred woody material up to 30 cm in diameter. The resulting chips and coarsely shredded debris are left on site. In the HPB treatment, fuel removal is accomplished using chainsaws and the cut material is piled for later burning. Burn piles are generally about 3 m in diameter and up to 2 m high; a few sheets of plastic are incorporated into them to keep their centers relatively dry for burning during the rainy season.

Both treatment methods open the canopy, increasing light availability on the ground, but neither removes the litter layer or approximates the soil surface conditions found after fire (Kauffman 2004). Mastication does not remove fuels, but alters their structure, leaving woody debris on the soil surface. Little woody debris remains after HPB treatment. Small patches of the HPB treatment area are subjected to high intensity fire from burned brush piles.

There are concerns about the potential consequences of the management techniques being used on the plant communities in southwestern Oregon. While fuel management has been applied to thousands of hectares in this region since the mid-1990's (USDI 1999), effects of treatments on fuel conditions, subsequent fire behavior, or plant or animal communities have received little attention (but see Perchemlides et al. 2008). The overall objective of this study was to determine the short term effect of fuel treatments on plant communities in which *Ceanothus cuneatus* is the dominant shrub. Specifically, we addressed the following questions. Do fuel treatments enhance or reduce opportunities for native plant establishment? Do they increase the spread of non-native weeds? Do the two types of fuel treatments affect the plant community differently? How does the presence of fire rings from burned piles affect species composition? We analyzed differences in plant community response and *Ceanothus* regeneration one and two years after both types of treatments.

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 Bureau of Land Management (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 (Fig. 1). 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 and Dyrness 1973). At higher elevation, mixed conifer forest begins to dominate (Franklin and Dyrness 1973). Chaparral occurs in the driest areas up to 1100 m (Detling 1961).

Elevation at our study sites ranges from 500 to 920 m. Precipitation varies with elevation, averaging 47 cm per yr at 395 m in Medford, and 92 cm in Butte Falls at 762 m (NOAA 2004; Fig. 1). Only 20 percent of the annual precipitation falls between April and September (Johnson 1993). The normal average July temperature is 20.7°C at 482 m (Lost Creek Dam), and 22.6°C at Medford. The average January temperature is 3.2°C at 482 m, and is similar throughout the elevation range of our study sites (NOAA 2004).

Study sites have moderate (4%–31%) slopes, predominantly south-facing. Soils are 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 can cause areas

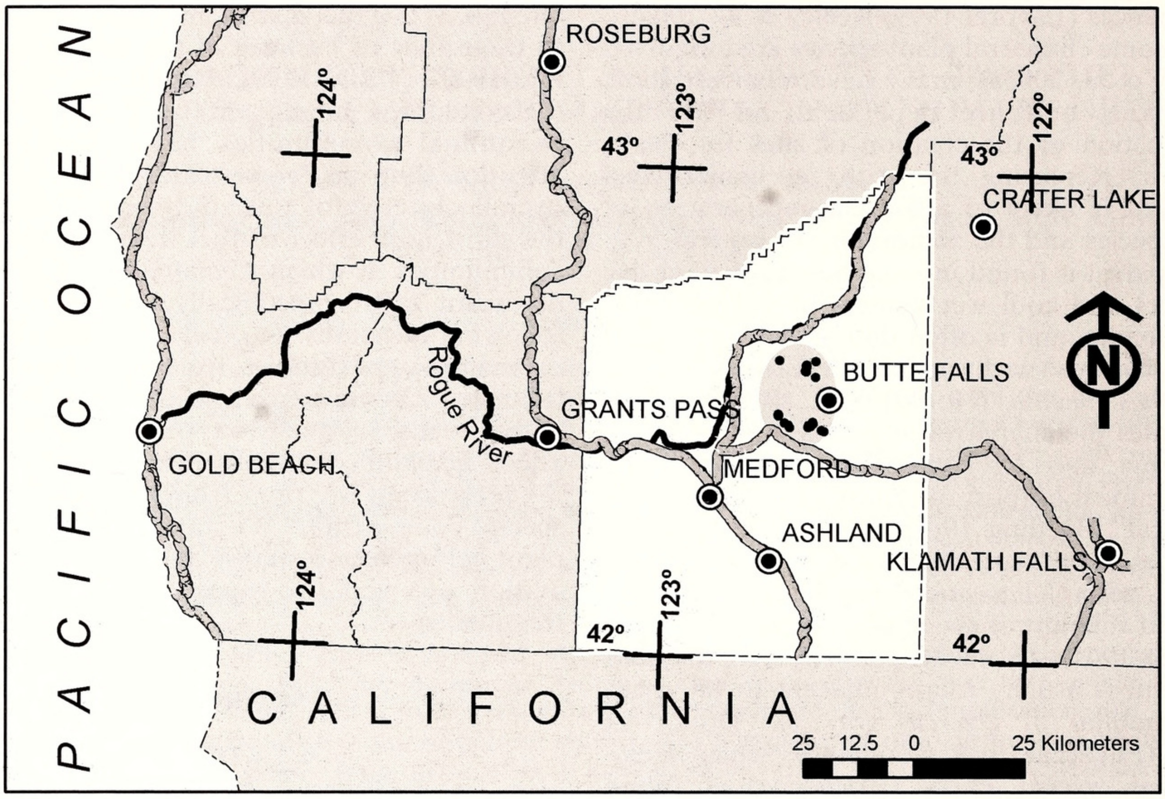

FIG. 1. Map of Jackson County, Oregon (in white), including major highway routes, significant population centers, and the Rogue River. Plot pairs are shown as dots within the grey oval in the vicinity of Butte Falls.

of standing water. The clay subsoil of the Medco soil keeps the water table high in winter and limits the effective rooting depth. McMullin soils are moderately permeable, but shallow, supporting mainly shrubs, grasses and forbs (Johnson 1993).

Ceanothus cuneatus was the dominant woody plant at all of the study sites. *Quercus garryana* (Oregon white oak) was common, with some oak canopy occurring in about half of the plots sampled. Herbaceous vegetation was dominated by annuals, especially non-native annual grasses. 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 Prescriptions

In both HPB and mastication treatments, approximately 75% of the shrub cover was removed (M. Wineteer, BLM, personal communication). 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 to produce a space between clumps of approximately 7.6 m (Fig. 2). Large healthy trees (i.e., conifers >18 cm diam. at breast height (DBH) and hardwoods >25 cm DBH) were not removed. Smaller trees were retained with a 7.6 m spacing.

Management units were contiguous areas with a single treatment prescription carried out by a single entity. Units varied in size from 1.2 to 76 ha. Treatment conditions often varied from place to place within a unit because treatments could take days or months to be completed. All burning of hand piles for the HPB treatment occurred in November or December of 2002 (2 yrs before sampling) or 2003 (1 yr before sampling). Cutting and piling, however, began as early as November of the previous year. Mastication treatments were completed in April, May, or June of 2002 and in November or December of 2003.

Field Methods

Plot establishment. Pairs of permanent plots (n = 26 plots) were established in summer 2003 near the boundaries of fuels reduction units slated to be treated the following fall or winter. We selected areas where the plant community and environment appeared similar on both sides of the boundary. Plot pairs were randomly located on either side of such boundaries. Random locations were rejected and redetermined if they resulted in conditions that clearly differed in community or aspect within a pair. Each plot was 50 m × 1 m (50 m^2) and was at least 15 m from the marked treatment boundary. Nine of the pairs established pre-treatment were subsequently treated, and re-sampled in 2004. One of the nine

FIG. 2. Photos of a management area before and after mastication treatment. Pre-treatment photo (top) was taken in July 2003, post-treatment (bottom) in May 2004.

pairs, an HPB treatment and control, was only partially treated, having been cut and piled, while the piles remained unburned.

Additional plots were established in 2004 near the boundaries of fuel management units that had already been treated. Some were 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 treatments had not occurred. One plot was matched with an existing control, resulting in five additional pairs sampled 1 yr since treatment. Other plots (n = 25) established in 2004 were in areas that had been treated in 2002. In locating retrospective plots, we reduced the required distance between plot and boundary to 7.5 m since we no longer needed to allow for potential differences between the boundary flagged for treatment and the actual treatment boundary.

Species abundance data reported here were collected between May 5 and July 28, 2004 in 52 plots, including six pairs for each treatment type and year and four additional unpaired plots that were sampled 1 yr post-treatment. Pair members were sampled on or near the same day. Plots were distributed across 13 land survey sections (each 2.59 sq km in area) and 21 management units. Some units were adjacent to each other, so that some pairs in different units were closer together than pairs in the same unit. The distances between pairs ranged from approximately 100 m to 20 km (Fig. 1), with the average being 870 m.

Sampling method. Physical data recorded for each 50 m^2 plot included cover estimates for bare soil (including gravel-sized rock), rock, litter, woody debris, and standing dead *C. cuneatus* shrubs (rooted in plot and >30 cm in height), with the following cover classes: 0 = none; 1 = < 10% cover; 2 = 10–25% cover; 3 = 25–50% cover; 4 = > 50% cover.

We recorded numbers of *Ceanothus* in each of four life stages: seedlings (plants that germinated that year with little or no woody tissue); immatures (plants with some woodiness ≤30 cm tall); living stumps (plants that had been cut but still had green foliage); and matures (uncut mature plants >30 cm tall). Living stumps did not show vigorous resprouting as observed in other woody species; it was unclear whether these plants would survive.

The abundance of each vascular plant taxon rooted in the plot was recorded. We used the following broad abundance classes, adapted from the Forest Health Monitoring protocol for lichens (USDA/FS 2002): 0 = Absent (not present in plot), 1 = Rare (1–3 individuals in the plot), 2 = Uncommon (4–10 individuals present), 3 = Common (more than 10 individuals but less than 50% cover), 4 = Abundant (greater than 50% cover). Most taxa were recorded as species; however, some that were not reliably distinguished in the field were combined with other species in their genus. Samples of unknown or uncertain species were collected for later identification or verification. See Sikes (2005) for a full species list and a list of vouchers accessioned by OSC (Sikes 45A-232).

Because the species composition of fire rings resulting from burned piles appeared different from the rest of an HPB treated plot, and because the proportion of the plot covered by fire rings was small, we adopted an additional sampling procedure to capture their species composition. A circular plot with an area of 0.25 m^2 was placed at the visually estimated center of each of three fire rings that most overlapped the HPB treated plot, or were closest to it. Species abundance within the circular plot was recorded using the same abundance codes given above. Percent cover of rock, litter, and woody debris was also estimated using the substrate cover codes described above. Forty-two fire rings were sampled in June or July 2004, three 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.

Data Analysis

Multivariate analyses were performed using PC-ORD for Windows, Version 4.25 (McCune and Mefford 1999). Other statistical tests were performed using S-PLUS 6.2 for Windows. Data included abundance codes for all taxa in four categories of post-treatment plots, distinguished by treatment type and time since treatment, and their paired controls. Number of taxa present in each plot was included as a plot attribute. Before analyses, we deleted species that occurred in fewer than four plots, to reduce heterogeneity in the data in order that the more rare species did not overly influence our results and to strengthen overall species composition patterns. We also deleted all woody plants from the primary dataset since we were interested in the treatment effects on other components of the plant communities.

We used blocked multi-response permutation procedures with Euclidean distances (McCune and Grace 2002) to compare species composition between treatment and control groups. Multi-response permutation procedure (MRPP) is a nonparametric test for multivariate difference between two or more pre-defined groups (Zimmerman et al. 1985). The test provides a P-value based on randomized group reassignments and 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 and Grace 2002). Blocking by pair focused the analysis on differences within the pair, while accounting for variation among pairs. We tested for differences between treatment and controls in all matched pairs, HPB treatments (including one partially treated pair), mastication treatments, plots treated the year prior to sampling, plots treated 2 yr prior to sampling, and the four subgroups made up of each treatment type by each treatment year.

We also created a data matrix to represent the treatment effect on each pair. For each matched pair, species abundance values for the control were subtracted from those for the treatment. The full set of 229 taxa, including rare and woody species, was used. We used this difference matrix to calculate a matrix of changes in species traits for assessing treatment effects on life forms (forb, graminoid, shrub, tree, annual, or perennial), geographic origin (native or non-native), and weediness of community constituents. Traits were determined from Hickman (1993) and the Plants database (USDA, NRCS 2004). Weedy plants (characteristic of disturbed habitats) were also categorized as either native or introduced weeds. Plants accorded special status by the Medford District BLM (M. Wineteer, personal communication) were also indicated. The change in species trait by pair matrix was created by multiplying the difference matrix described above by a matrix of traits by species.

Within-pair differences in each trait were tested for differences from zero using one-sample t-tests. Two-sample t-tests tested whether changes in traits differed between treatment types and between time-since-treatment groups. We also tested for changes in species richness and canopy cover between these groups. T-tests were chosen because a normal distribution was approximated; however, P-values should be treated with caution because multiple comparisons were made and geographic proximity between some pairs probably compromised their independence.

To compare how many species of each trait had changed in treated versus control plots, a second change of species trait by pair matrix was created by converting each change in species

TABLE 1. TESTS FOR DIFFERENCES IN COMMUNITY COMPOSITION BETWEEN FUEL-TREATED PLOTS AND MATCHED CONTROLS, MRPP BLOCKED BY PAIR (WOODY SPECIES DELETED). The larger the A-value the less likely the similarity within the pre-defined groups can be expected by chance.

Group	Number of plots	Within-group agreement (A)	Probability (P)
All matched pairs	48	0.007	0.024
HPB treatment	24	0.012	0.036
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	0.029	0.038
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

abundance by pair in the difference matrix to 1 or −1 (increase or decrease) such that the subsequent multiplication resulted in a matrix containing the number of species changed rather than the quantitative changes in abundance values. We then used the full 229 taxa dataset to make a presence-absence matrix for control plots and calculate the number of species per trait in these plots. Percent of species changed by trait was calculated using the average number of species differing between treatment and control for a given trait divided by the average number of species with that trait in the control plots.

We also tested differences in treatment effects on species composition between the following groups using MRPP: plots established before treatment in 2003 versus post-treatment in 2004, time since treatment (1 yr or 2), and mastication versus HPB. These analyses used the matrix of treatment – control differences in species abundances, from which we deleted all woody taxa, resulting in 208 taxa for 25 pairs of plots. The relatively even distribution of data in this matrix made it unnecessary to delete rare species to reduce skewness or coefficients of variation. The matrix contained negative numbers, so MRPPs were based on Euclidean distances (McCune and Grace 2002).

Effect of treatment on Ceanothus. We tested for differences in abundances of *Ceanothus* life stages collectively between the two treatments and between years since treatment using MRPP on treated plots only, and for differences in abundance of each individual stage between treatments and controls using a two-sample t-test. We compared within-pair differences in abundance of each *Ceanothus* stage (treated plots minus paired controls) between years since treatment and between treatment types using two-sample t-tests.

Fire rings of HPB treatment. We tested for differences in the species composition of fire rings between the first year and the second year after burning, and between the two ages of HPB treated plots in their entirety using MRPP (Sørenson distance; McCune and Grace 2002). All species were retained to better represent species richness. We also compared the average proportions of species by trait found in the two ages of fire rings and their associated treatment plots.

We used Indicator Species Analysis (ISA; Dufrêne and Legendre 1997) to look at species differences between the two ages of fire rings, and the two ages of HPB plots. ISA produces indicator values that combine relative abundance and relative frequency of each species by group to represent faithfulness and exclusivity to that group. A Monte Carlo test of 1000 randomizations (McCune and Grace 2002) was used to test the significance of species indicator values.

RESULTS

Pair members established pre-treatment in 2003 did not differ in species composition before treatment (blocked MRPP: P = 0.175; Sikes 2005). In 2004, after treatments had occurred, the average species richness per plot (*alpha* diversity) was 71.8, and *beta* diversity (the total number of species in all plots/*alpha*) was 3.19. 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 on differences between matched treatment and control plots in 2004; 95% confidence interval from 0.2 to 8.2 species). In addition, treated plots had greater mean species abundance compared to control plots, averaging an increase in abundance for 5.6 species (P = 0.017, one sample t-test; 95% confidence interval from 1.1 to 10.2 species). There was significantly more cover of woody debris in both types of treatment plots than in control plots (P = 0.001, one-sample t-test on treatment minus control); the mean cover class for control plots was 0.96 (almost all with <10% cover woody debris) and 1.36 for treated plots (with about one third recorded as 10–25% cover of woody debris). No other abiotic plot attributes differed significantly after treatment.

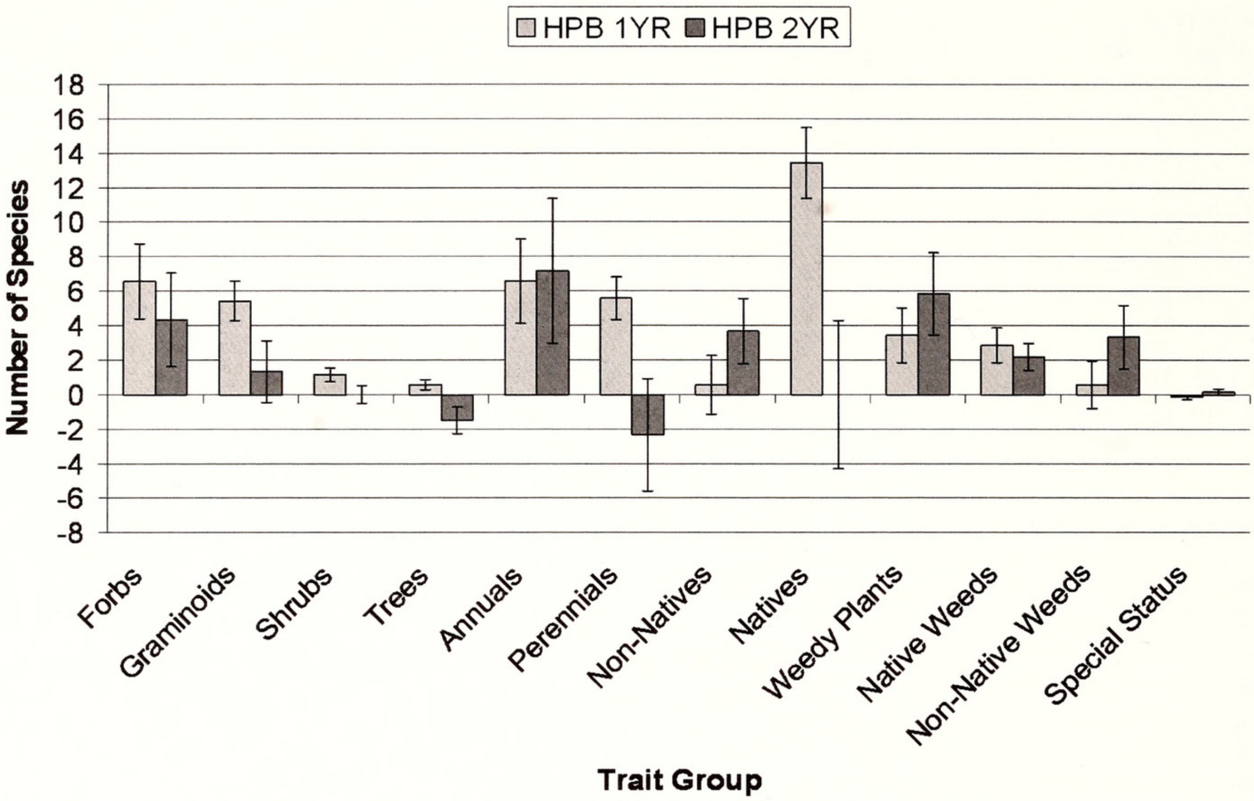

FIG. 3. Change in species traits for "hand cut, pile, and burn" (HPB) treatment and control pairs 1 and 2 yr since treatment. Bars show average number of species that increased or decreased in abundance between treatment and control plots by trait, with standard errors. See Table 2 for means across years and the percentage change by trait group.

Treatment Effect on Species Composition

After treatments were applied, herbaceous plant communities differed ($P < 0.05$) between treatments and controls when all treatments were compared to all controls (blocked MRPP on 48 plots; Table 1). The species composition of the HPB-treated plots also differed from their controls, whereas the mastication treatment plots and controls did not differ (Table 1). This difference in results between treatment types indicates that the HPB treatment, in the short term, had a greater effect on species composition than did mastication. In all cases, however, the effect sizes were modest, as judged by small measures of within-group agreement (A).

When treatments and controls were grouped by time since treatment, communities in 1-yr-since-treatment plots differed from their controls, while the 2-yr-since-treatment plots did not (Table 1). Within each treatment type, communities also differed between treatment and controls 1 yr after treatment, while neither differed 2 yr post-treatment (Table 1). These results suggest that treatment effects on community composition were greatest immediately after treatment.

Differences in species abundance between matched treatment and control plots in 2004 provided a measure of community composition change presumably due to treatment. Pairs established in 2003 versus 2004 did not differ in overall species composition changes (MRPP on matrix of treatment-control differences: $A = 0.002$, $P = 0.206$), indicating that results were not affected by whether plot establishment occurred before or after treatment. Species composition changes also did not differ between pairs grouped by treatment type (MRPP: $A = 0.002$, $P = 0.223$). There were, however, significant differences in composition changes between pairs sampled 1 yr versus 2 yr post-treatment (MRPP: $A = 0.008$, $P = 0.015$).

The HPB treatment resulted in larger increases in species richness and abundance than did the mastication treatment (Figs. 3, 4; Table 2). Species abundance in all trait groups except trees was greater in HPB treated plots across years than in untreated plots (Fig. 3, Table 2). Compared to mastication, the HPB treatment caused significantly greater increases in abundance of native weeds and shrubs ($P \leq 0.018$, $df = 23$, two-sample t-test). An increase in shrub abundance after treatment that removed shrubs was possible due to seedling recruitment post-treatment. Both treatments tended to favor annuals more than perennials. Differences between responses of non-natives and natives were not as pronounced.

When differences were separated by time since treatment, the 1 yr since treatment pairs generally

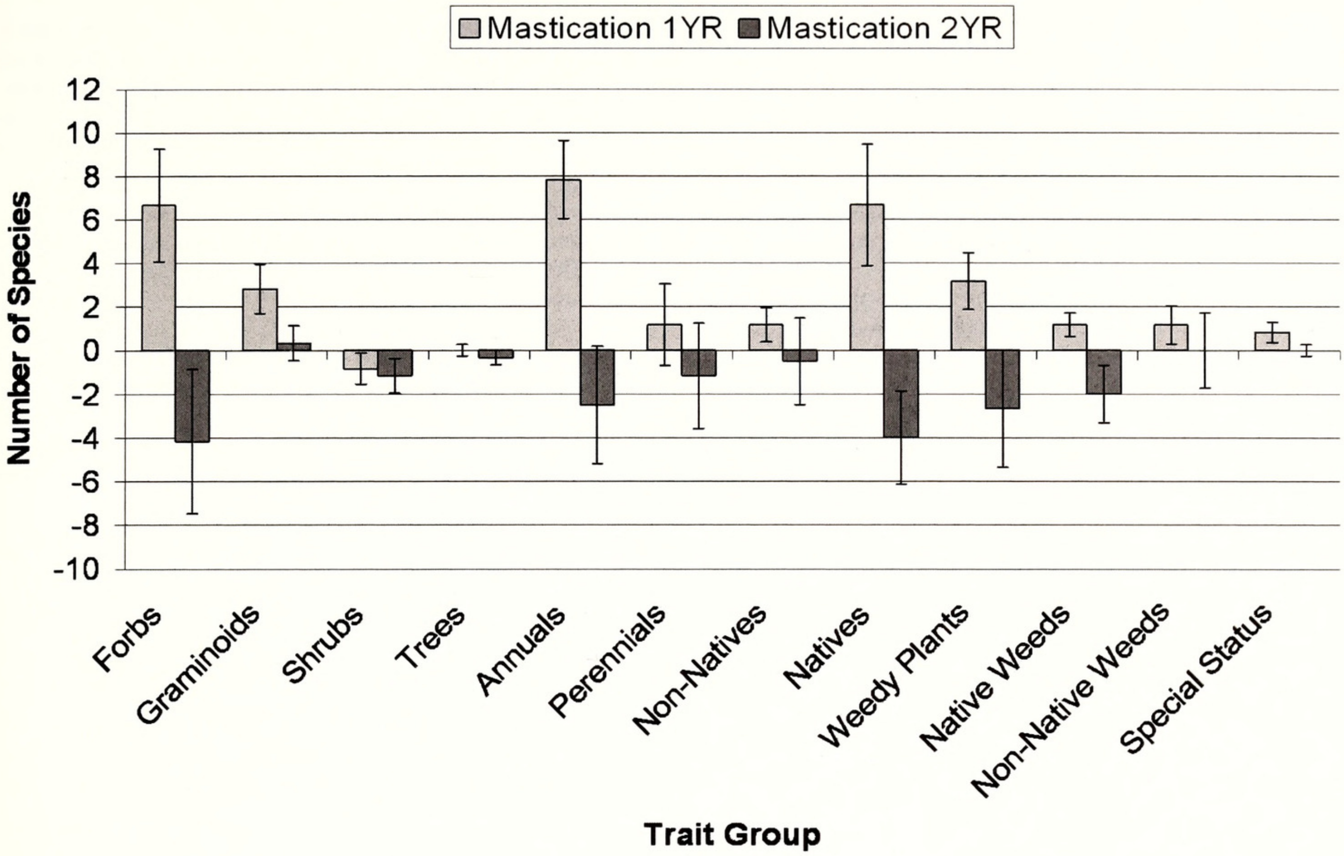

FIG. 4. Change in species traits for mastication treatment and control pairs 1 and 2 yr since treatment. Bars show average number of species that increased or decreased in abundance between treatment and control plots by trait, with standard errors. See Table 2 for means across years and the percentage change by trait group.

showed greater increases in species abundance than did the 2 yr since treatment pairs (Figs. 3, 4; Table 2). More graminoids, trees, perennials, and natives had increased abundance in treated relative to untreated plots 1 yr after treatment than did those groups in plots sampled 2 yr post-treatment ($P < 0.05$ by two-sample t-test). Of these trait groups, all but graminoids were less abundant in treated than in untreated plots 2 yr after treatment.

While species richness increased in the first year after fuel treatments, there were fewer species in treated than in control plots after 2 yr. For first year plots, the average species richness was 74.1 species in treated plots and 62.5 species in the controls. In second year plots, the

TABLE 2. CHANGES IN NUMBERS OF SPECIES PER TRAIT GROUP BY TREATMENT TYPE, ACROSS YEARS SINCE TREATMENT, EXPRESSED AS THE MEAN NUMBER OF SPECIES THAT INCREASED OR DECREASED (TREATMENT MINUS CONTROL; AS SHOWN IN FIGS. 3, 4). The percentage change in the trait group is calculated as the mean change in species divided by the average number of species for that trait in the control plots.

	Hand, pile, and burn		Mastication	
Trait group	Species differed	Percent change in trait group	Species differed	Percent change in trait group
Forbs	5.5	10.9	1.3	2.5
Graminoids	3.5	22.3	1.6	11.9
Shrubs	0.6	23.8	−1.0	−32.4
Trees	−0.4	−27.1	−0.2	−28.6
Annuals	6.8	14.7	2.7	5.7
Perennials	1.9	9.9	0.0	0.0
Non-natives	2.0	12.6	0.3	2.1
Natives	7.2	13.6	1.3	2.6
Weedy plants	4.5	17.6	0.3	0.9
Native weeds	2.5	25.8	−0.4	−3.4
Non-native weeds	1.8	12.4	0.6	4.0
Special status	0.0	0.0	0.4	100.0

TABLE 3. INDICATOR VALUES (IV) FOR SPECIES IN PERCENTAGE OF PERFECT INDICATION FOR SECOND YEAR FIRE RINGS. Perfect indication (100%) occurs when the species is always present in that group and never occurs in other groups. Only statistically significant indicator species are listed (P < 0.05). Boldface indicates non-native species.

Annuals, not noted as weedy	IV	Weedy annuals	IV
Agoseris heterophylla (Nutt.) Greene (Asteraceae)	66.7	***Aira caryophyllea* L.** (Poaceae)	48.9
Cardamine oligosperma Nutt. (Brassicaceae)	42.9	***Bromus japonicus* Thunb. ex Murr.** (Poaceae)	38.1
Clarkia purpurea ssp. *quadrivulnera* (Dougl. ex Lindl.) H.F. & M.E. Lewis (Onagraceae)	34.6	***Bromus tectorum* L.** (Poaceae)	33.3
Cryptantha torreyana (Gray) Greene (Boraginaceae)	28.6	***Cerastium glomeratum* Thuill.** (Caryophyllaceae)	33.3
Madia exigua (Sm.) Gray (Asteraceae)	39.7	***Galium* L. spp.** (Rubiaceae)	76.2
Linanthus bicolor (Nutt.) Greene (Polemoniaceae)	37.0	*Epilobium* L. spp. (Onagraceae)	85.7
Madia Molina spp. (Asteraceae)	38.1	***Gastridium ventricosum* (Gouan) Schinz & Thellung** (Poaceae)	42.9
Phlox gracilis (Hook.) Greene (Polemoniaceae)	28.6	***Lactuca serriola* L.** (Asteraceae)	28.6
		Lotus humistratus Greene (Fabaceae)	33.3
		***Myosotis discolor* Pers.** (Boraginaceae)	28.6
		Veronica L. spp. (Scrophulariaceae)	29.6
		***Vulpia myuros* (L.) K.C. Gmel.** (Poaceae)	47.6

average was 73.6 species per plot in treated plots and 76.0 species for controls. This difference in species richness between treatment and control of the time-since-treatment groups was statistically significant (P = 0.004, df = 23, two-sample t-test). 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). When the year-since-treatment groups were further divided into their treatment groups, we found that species richness was more reduced in the second year after mastication than in the corresponding year after HPB treatment. In fact, 2 yr after treatment, the HPB treated plots supported more species than their control plots. The same trends are visible in the number of species per trait that changed in abundance (Figs. 3, 4; Table 2).

Effect of Treatment on *Ceanothus*

When all treatments were compared to all controls, seedlings were significantly more abundant in treatments (P = 0.024, two-sample t-test). However, the abundance of immature *Ceanothus* did not differ between treatments and controls (P = 0.19, two-sample t-test across treatment types and years). Comparing within-pair differences, uncut mature *Ceanothus* were substantially less abundant after treatment (see Fig. 2), as was cover of standing dead *Ceanothus* (both P < 0.001, one-sample t-test). The combined life stages of *Ceanothus* were less abundant after treatment (P = 0.030, one-sample t-test).

Abundances of *Ceanothus* life stages in treated plots differed between years since treatment (MRPP: A = 0.068, P = 0.004), but not between the treatment types (MRPP: A = 0.021, P = 0.129). The abundance of the species across stages was more reduced by mastication than by the HPB treatment (P = 0.050, two-sample t-test comparing within-pair differences). The within-pair difference in the abundance of immature *Ceanothus* was greater in the second year than in the first year after treatment (P = 0.003, two-sample t-test); abundance of this life stage was reduced in year one, but in year two, immatures were more abundant in treated than in untreated plots. Other life stages did not differ significantly in their within-pair difference between years-since-treatment nor treatment types.

Fire Rings of HPB Treatment

Eighty-nine species were recorded in fire rings, despite their relatively small area, which was about half of the 184 species found in the associated HPB treatment plots. Species composition in fire rings differed between 1 yr and 2 yr post-treatment groups (MRPP: A = 0.108, P < 0.001). Only one species was a significant indicator for first year fire rings: *Brodiaea elegans* Hoover (Liliaceae), a native perennial geophyte (ISA: IV = 50.0). Twenty species were significant indicators for second year fire rings. All of these were annuals, most were weedy, and about half were non-native (Table 3).

Species composition and abundance in the HPB treatment plots in entirety also differed 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 4). Only five were annuals, and only two of these were considered weedy (*Myosotis discolor* Pers., Boraginaceae; *Trifolium wildenowii* Spreng., Fabaceae). The non-native *Myosotis* was the only species that indicated both fire rings and HPB treatment plots.

TABLE 4. INDICATOR VALUES (IV) FOR SPECIES IN PERCENTAGE OF PERFECT INDICATION FOR "HAND CUT, PILE, AND BURN" (HPB) TREATMENT PLOTS BY YEAR SINCE TREATMENT. Perfect indication (100%) occurs when the species is always present in that group and never occurs in other groups. Only statistically significant indicator species are listed ($P < 0.05$). Boldface indicates non-native species.

1 yr since treatment	IV	2 yr since treatment	IV
Clarkia gracilis (Piper) A. Nels. & J.F. Macbr. (Onagraceae)	76.9	*Agoseris grandiflora* (Nutt.) Greene (Asteraceae)	100.0
Elymus elymoides (Raf.) Swezey (Poaceae)	68.0	*Calochortus tolmiei* Hook. & Arn. (Liliaceae)	81.8
Lomatium utriculatum (Nutt. ex Torr. & Gray) Coult. & Rose (Apiaceae)	82.6	*Galium porrigens* Dempster (Rubiaceae)	100.0
***Myosotis discolor* Pers.** (Boraginaceae)	76.5	*Hesperolinon micranthum* (Gray) Small (Linaceae)	61.2
Plagiobothrys cognatus (Greene) I.M. Johnston (Boraginaceae)	71.4	*Horkelia daucifolia* (Greene) Rydb. (Rosaceae)	85.7
Poa secunda J. Presl. (Poaceae)	75.0		
Trifolium willdenowii Spreng. (Fabaceae)	80.0		

The age of the fire rings affected the average number of species present. First year fire rings averaged 3.2 species per 0.25 m^2 plot, while second year rings averaged 14.2 species. The average abundance value per species was 1.1 in the first year, indicating that almost all species were represented by only 1–3 individuals. The average abundance class was 1.5 in second year fire rings, which corresponds to half of the species having 1–3 individuals and half having 4–10. The HPB treatment plots in entirety, with 200 times the area of the fire ring plots, supported an average of 77.6 species per plot, with an average abundance value of 2.5; most species were represented by more than 10 individuals.

First year fire rings had significantly fewer species in all trait groups except shrubs and perennials than did second year fire rings ($P < 0.001$, two-sample t-test). In contrast, native weeds was the only trait group for which species numbers differed significantly between years in HPB treatment plots overall ($P = 0.013$, two-sample t-test). There was an average of 3.7 more native weed species in 1 yr than in 2 yr since treatment plots (95% confidence interval between 1.0 and 6.5 species). Since the proportions of species representing various trait groups did not differ appreciably between years since treatment, we combined ages for HPB treatment plots overall for comparisons with fire rings.

In HPB treatment plots overall, the herbaceous species were mostly annuals (70.6%). First year fire rings, however, had slightly more perennials than annuals (56% versus 44%); most appeared to have resprouted after the fire. Very little colonization of the newly open area had occurred by year one. In the second year fire rings, 94.8% of the species were annuals. The higher proportion of annual species and greater species richness in year two indicate that more colonization had occurred after two years.

The proportion of native weeds, compared to the sum of non-native weeds and non-weedy plants, was similar in the two ages of fire rings and HPB treatment plots overall, ranging between 15.2 and 17.3% of the species present. The proportions of non-native weeds, however, varied strikingly among these plot types. There were comparatively few non-native weeds in the first year fire rings (6.1%), but non-native weed species increased to twice the number of native weeds in the second year, accounting for 35.3% of the species present. The HPB treatment plots showed an intermediate proportion of non-native weeds (20.8% of the species present).

Several species occurred in only one of three situations: fire rings, HPB treatments, or HPB controls. Even species that appeared to be stimulated by fire, however, such as *Ceanothus* seedlings, were present in control plots as well as in treated areas. Three species occurred in more than one HPB plot but not in any control plots, and all of these grew in association with burn piles. One of them, *Lactuca serriola* L. (Asteraceae), an annual non-native weed, was an indicator species for the second year fire rings (Table 3). The other two species are native annuals, *Stephanomeria virgata* Benth. (Asteraceae) and *Gnaphalium palustre* Nutt. (Asteraceae). Though only *Gnaphalium* was categorized as weedy, *Stephanomeria* is a composite with wind-dispersed seeds.

DISCUSSION

The pre-treatment and control plots did not differ significantly in species composition (Sikes 2005), thus post-treatment differences between treatment and control can most probably be interpreted as treatment effects. The short term treatment effects on the herbaceous community that we detected were generally small. Other factors, such as presence of oak canopy, had a stronger influence on species composition than did treatment (Sikes 2005). Both oaks and *Ceanothus* provided important habitat for natives and perennials. While open areas were dominated by non-native annual grasses such as *Taeniatherum caput-medusae* (L.) Nevski (Poaceae), they also supported several native annuals that are of special interest to the BLM including *Navarettia subuligera* Greene (Polemoniaceae) and *Plagiobothrys greenei* (Gray) I.M. Johnston (Boraginaceae).

While treatment-induced differences in herbaceous plant communities were smaller than expected, given the dramatic reductions in shrub cover, communities in treated areas did differ from those in controls. Treatment effects were larger for HPB than for mastication treatments, in general, and also tended to be stronger in the first year after treatment than in the second year. First year communities of both treatments differed from controls, while communities did not differ in the second post-treatment year of either treatment. The lack of mastication treatment effects across years probably occurred because the general increase of species abundance in the first year after mastication was counterbalanced by a general decrease in the second year.

In general, the effect of time since treatment was stronger than the effect of treatment type. Taking all species composition differences between matched treatment and control plots into account, we found that the two types of fuel treatment did not differ, consistent with community responses 4 to 7 yr after treatment in a nearby study area (Perchemlides et al. 2008), while the year-since-treatment groups did differ. These findings parallel those for comparisons between trait group abundances, in which twice as many trait groups differed by time since treatment as by treatment type.

There was a general increase in plant species abundance and species richness with treatment, with the exception that both were lower in treated areas than in controls 2 yr after mastication. Overall shrub abundance did not differ between treatments and controls across treatment types, even though shrubs were targeted for reduction. While *Ceanothus* abundance across all life stages was significantly decreased after treatment, the effect was smaller than anticipated because of the survival of some cut stems and an increase in *Ceanothus* seedlings with treatment. Fuel treatments encouraged regeneration, even in the absence of fire, and it appears that the reduction of standing fuel will be short-lived. *Ceanothus* species can revert to closed crowns within five to seven years after mechanical brush clearing (Green 1977), and even young chaparral can burn readily under some wildfire conditions (e.g., Fried et al. 2004; Keeley and Fotheringham 2001b; Moritz et al. 2004). Overall shrub abundance was also affected by the occurrence of species other than *Ceanothus*. Other shrub species either were not removed because of low density or tended to resprout vigorously. Woody species that appeared to resprout without fail after cutting included *Toxicodendron diversilobum* (Torr. & Gray) Greene (Anacardiaceae), *Quercus garryana*, *Amelanchier alnifolia* (Nutt.) Nutt. ex M. Roemer (Rosaceae), and *Prunus subcordata* Benth. (Rosaceae). Rapid restoration of a shrub canopy following treatment, while perhaps undesirable from a fuel management perspective, can significantly decrease the likelihood that exotic species presence and abundance will increase post-treatment (Keeley et al. 2005a, b).

The HPB treatment caused a general increase in shrub abundance, while mastication decreased them. Abundance of *Ceanothus*, in particular, was apparently more reduced by mastication than by the HPB treatment. The fire rings of the HPB treatment were responsible for this difference between treatment types. Though *Ceanothus* seedlings occurred in most plots, whether control or treatment, their affinity for burned areas was evident, and resulted in greater differences in seedling abundance between treatment and control in HPB than in mastication treatments.

Previous work has emphasized that *C. cuneatus* is an obligate seeder that generally 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 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 and Quick 1961). In addition, some fraction of seed produced by fire-recruiting species, including *Ceanothus* species, often lacks dormancy (Keeley 1991). These alternative situations, which allow germination of *Ceanothus* in the absence of fire, appeared to be in operation throughout our study area, consistent with reports of *C. cuneatus* seedlings and immatures being found 4 to 7 yr after HPB or mastication treatments in a nearby study area (Perchemlides et al. 2008).

Because *Ceanothus* seedlings were generally more abundant in mastication treatments than in controls, it appears that the treatment may have increased germination, perhaps by scarifying seed or improving microhabitats. A congener, *C. greggii*, showed increased germination a year following the clipping of standing chaparral at about 10 cm from the ground (Moreno and Oechel 1991). Though often described as having no substantial germination in the absence of fire, in a previously disturbed site that was quite open and invaded by exotic annual grasses, Keeley

(1992b) found both seedlings and uneven-aged shrubs of *C. cuneatus*.

The other species trait group that was significantly more abundant in HPB treatments than in mastication treatments was native weeds. An average of 26% of native weed species increased in HPB treatments relative to controls, while the corresponding change was a 3% decrease in mastication plots. Native weeds was the only trait group for which significantly more species were present in HPB plots 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*. The proportion of native weeds was not greater in fire rings than in the larger treatment plot, so fire rings were not responsible for the difference between treatments. In fact, only 2 of the 20 indicator species of second year fire rings were native weeds, *Epilobium* L. spp. (Onagraceae) and *Lotus humistratus* Greene (Fabaceae).

Weedy plants would be expected to increase after either type of treatment, especially in the first year, because of the availability of newly exposed sites for establishment. Although treatment types did not differ in overall cover of bare ground or woody debris (Sikes 2005), treatment-specific differences in the distribution of woody debris could help explain the difference in native weed abundance between treatment types. Mastication leaves pieces of debris distributed fairly evenly over the area, perhaps inhibiting seed germination or seedling establishment, while HPB leaves debris massed at the edge of burn piles. Another difference between the two treatments that might affect native weed success is degree of soil disturbance. Both mastication and HPB 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.

The larger increases in species abundance for treated plots compared to controls in the first year versus the second year after treatment may reflect an initial pulse of resource availability. In the first year, more light and resources were newly available, so the ground layer responded accordingly. Disproportionately large increases might be expected for weedy plants, since they specialize in colonizing new and disturbed habitats. However, the weedy trait groups were not significantly more abundant in the first year than the second year, and their percentage changes were not large compared to those for other trait groups. These findings contrast with other studies, perhaps due to the already abundant weed presence in our sites prior to treatment. For example, weedy and exotic plants were more abundant on fuel breaks than on adjacent untreated areas throughout California (Merriam et al. 2006). Indeed, four to seven years after treatment, cover by exotic annual grasses was nearly twice as high on masticated or HPB treated sites compared to controls in a chaparral study area within the same BLM district (Perchemlides 2006; Perchemlides et al. 2008).

The fire rings that result from HPB treatment deserve individual consideration. The footprints of burn piles provide sites for invasive or weedy plants to establish (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 can burn at higher temperatures or over a longer period than most naturally occurring fuel loads, resulting in increased soil heating and greater damage to biotic and abiotic soil properties. Long duration soil heating causes more damage than shorter duration heating (DeBano et al. 1979). Prolonged heating is also associated with the burning of deep accumulations of masticated wood residues (Busse et al. 2005), either in prescribed burns or with unintended wildfire. Increased damage may take the form of greater water repellency (e.g., MacDonald and Huffman 2004), altered soil chemistry and structure (Shea 1993), seed mortality, and mycorrhizal sterilization (Korb et al. 2004).

Natural fires usually occur under dry conditions rather than in the moist conditions that are purposely chosen for burning piles to reduce fire danger. Burn season has complex influences on soils and biota (e.g., Knapp et al. 2007). For example, while moist soil can reduce many impacts of heating during fire, damage to soil micro-organisms can be increased in moist soils (DeBano et al. 1979; Busse et al. 2005; but see Smith et al. 2004) and seed germination of some chaparral species is decreased under moist as compared to dry heat (LeFer and Parker 2005). These factors, coupled with seasonal differences in availability of native and exotic species seed, indicate that differences in timing of pile burning could affect post-treatment responses of the plant communities.

Some factors relevant to fire effects are unique to the chaparral ecosystem. Waxy-leaved shrublands are especially susceptible to post-fire water repellency of soil, because of hydrophobic substances produced by chaparral plants (Beschta et al. 2004). Chaparral has a thinner litter layer than forests and therefore soils are less insulated from heat (DeBano et al. 1979). In addition, shrubland wildfires often 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, although, even in chaparral, it is not likely that natural conditions would produce such concentrated fuel loads and resultant heat, or heating of as long duration, as occur in burn piles. In standing chaparral, seed recruitment after an autumn fire tends to be concentrated in areas that were gaps in the pre-burn vegetation, reflecting the high temperatures that occur in relatively dense areas of vegetation (Odion and Davis 2000).

Though *Ceanothus* germination was promoted in the fire rings, soil temperatures may have been higher than those occurring during wildfires. Occasional *Ceanothus* (*C. cuneatus* var. *fascicularis*) and *Arctostaphylos* seedlings occurred in the highest fire intensity areas in a chaparral system, whereas no other taxa germinated or resprouted at those temperatures (Odion and Davis 2000). *Ceanothus greggii* showed increased germination when fuel loads were moderately increased in a chaparral stand, and its germination was similar to that occurring under normal fuel load when fuels were greatly enhanced (Moreno and Oechel 1991). Reactions to changes in fuel load varied by species, but the majority of chaparral species present showed decreased seedling production with increased fire intensity (Moreno and Oechel 1991). Without the diversity of fire intensity inherent in a fire through standing chaparral, in which soil temperatures range from slightly heated to severely heated over the landscape, species diversity is likely to be reduced.

The indicator species of the second year fire rings were mostly weedy and half of the species were non-native. The fact that most of the indicators for the associated HPB treatment plots were native perennials 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 under the burning piles. The plants that occur in the fire rings tend to be species whose underground tissues can withstand high temperatures and prolonged heating or colonizers that are efficient dispersers. It is apparent that many non-native species succeed 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), *Toxicodendron diversilobum*, *C. cuneatus*, *Gilia capitata* Sims (Polemoniaceae), *Stephanomeria virgata*, *Clarkia purpurea* (Curt.) A. Nelson & J.F. Macbr. (Onagraceae; Keeley 1991), *Trifolium microcephalum* Pursh (Fabaceae), and *Juncus bufonius* L. (Juncaceae; 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 include species that have been noted as post-fire recruiters, including *Calystegia*, *Lotus*, *Cryptantha*, *Gnaphalium*, *Galium*, *Collinsia* (Keeley 1991), and *Navarretia* (Odion 2000).

The changes in proportion of species by trait across years in fire rings give an interesting snapshot of fire ring colonization, showing increased domination by annuals and introduced weeds over the two years. The proportions of species attributes in the larger associated treatment plot lie between the two extremes of the first year and second year fire rings' distribution of species attributes. Longer term studies of succession in fire rings are needed to determine whether they provide a locus for enhanced invasion by such species into the surrounding area or they eventually return to a similar species composition as their environs.

Conclusion

This study did not show a large effect of fuel treatments on herbaceous plant communities in the short term, though vegetation structure was altered by removing or redistributing woody biomass. Fuel treatments may have caused relatively little community alteration because of a history of disturbance and the already extensive occurrence of introduced species in our study site. Alternatively, our relatively coarse abundance data may have been insufficiently detailed to detect changes that occurred.

Both fuel treatments increased species richness initially, probably because both caused disturbance and increased resource availability. 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 primary factor responsible for the difference between the two treatments appeared to be the fire rings that remain after piles are burned in the HPB treatment. Though soil heating is likely greater or longer lasting 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, and it may allow fire-adapted species in the community to persist.

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

With the evidence at hand, it appears that neither fuels reduction treatment is a definite detriment to the plant community over the one to two year post-treatment period we studied. Short term data suggest that the HPB treatment may lead to an increase in weedy and non-native species. At the same time, however, it may increase native 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, and rapid recovery of a shrub canopy is likely to reduce abundance of weedy and non-native species over time (Keeley et al. 2005a). With patchy application, both treatments will increase the heterogeneity of the overall northern chaparral community in the absence of wildfire.

Our results do not lead to a clear recommendation concerning future management of chaparral fuels in southwestern Oregon. Further monitoring will increase our understanding of the longer term effects of these fuel treatments on plant communities. In the meantime, managers should be aware of the potential negative effects that are linked to either treatment.

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