

AN ABSTRACT OF THE THESIS OF

Keith A. Perchemlides for the degree of Master of Science in Botany and Plant Pathology presented on June 2, 2006.

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Patricia S. Muir

In response to concerns about excessive stand densities and high-severity wildfires, land managers in the western United States are carrying out extensive programs of fuel reduction thinning. How will these sudden reductions in canopy cover and associated changes in habitat affect native and exotic herbaceous vegetation and canopy species regeneration? Where thinning treatments have both fuel-reduction and ecological restoration objectives, to what extent are these goals compatible? I compared vegetation and abiotic site characteristics between thinned and unthinned chaparral and oak communities of southwestern Oregon where landscape-scale fuel-reduction thinning is occurring. I sampled paired thinned and unthinned transects, established four to seven years post-treatment. I also contrasted impacts of manual vs. mechanical thinning methods, and examined differences in herbaceous composition and responses among canopy community types. Thinning treatments had significant impacts on site conditions and herbaceous cover and composition. Herbaceous cover increased on thinned sites, but species richness and diversity did not change. The strongest differences were decreases in perennial species cover and a near doubling of cover by annual species on thinned sites. While overall proportions of natives and exotics did not change, there were changes at the functional group level. Exotic annual grass cover and the proportion of herbaceous community cover composed of these species increased, while cover by native perennial grasses and regeneration of oaks did not. Cover and proportion of native annual forbs increased more than all other functional groups, while exotic annual forbs and native perennial forbs declined. Herbaceous communities at thinned sites had an early-post-disturbance type of composition dominated by native annual forbs and exotic annual grasses four to seven years after treatments. Re-establishment of native shrubs was sparse in thinned areas, likely due to a lack of fire-stimulated germination. Sites that varied in canopy species composition also supported different herbaceous communities in the absence of thinning, and appeared to respond differently to fuel reduction treatments. Manual and mechanical treatment impacts on abiotic site conditions differed, but their overall impacts

on vegetation across canopy community types did not. Treatment type differences in herbaceous responses to thinning were found within some canopy vegetation groups, but sample sizes were small. Results suggest that fuel-reduction thinning may have some unintended negative impacts on oak and chaparral communities of southwestern Oregon. Although treatments have altered fuel conditions, thinning does not appear to have achieved restoration goals and may have substantially changed the composition and regeneration of native perennial communities including canopy species. Continued research and expanded monitoring that account for differences between canopy vegetation communities and treatment types are needed to inform adaptive management in these ecosystems.

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Impacts of Fuel Reduction Thinning Treatments on Oak and Chaparral Communities of
Southwestern Oregon

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Keith A. Perchemlides

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APPROVED:

Redacted for Privacy

Major Professor, Patricia S. Muir, representing Botany and Plant Pathology

Redacted for Privacy

Chair of the Department of Botany and Plant Pathology

Redacted for Privacy

Dean of the Graduate School

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Impacts of Fuel Reduction Thinning Treatments on Oak and Chaparral Communities of Southwestern Oregon

INTRODUCTION

Decades of fire suppression and other human influences have led to changes in vegetation structure and composition in many western ecosystems (Covington and Moore 1994; Heyerdahl et al. 2001; Taylor and Skinner 2003; Odion et al. 2004; Miller et al. 2005). There is widespread concern that resultant increased stand densities and wood biomass accumulations have elevated the risk and occurrence of high-severity, stand-replacing fires (Minnich 1983; Agee 1993, 1998; Covington et al. 1997; Office of the President 2002). Fire exclusion has also been implicated in changes in community composition and loss of habitat diversity, as species that rely on fire-disturbance to persist are replaced by those more competitive in the absence of fire (Kauffman and Martin 1987; Reed and Sugihara 1987; Swetnam et al. 1999; Franklin et al. 2004; Miller et al. 2005). Complicating the situation are the growing numbers and populations of exotic invasive plants, many of which invade or expand following disturbances (including fire), displacing native vegetation and sometimes altering fire regimes (Hobbs and Huenneke 1992; Pyke 1999; Sheley and Petroff 1999; Brooks et al. 2004). Recent trends of residential expansion into wildlands have: exacerbated the risk to human life and property from wildfire (Dombeck et al. 2004); increased fire occurrence due to augmented ignitions (Keeley et al. 1999); and facilitated the spread of invasive plants (Sheley and Petroff 1999; Gelbard and Belnap 2002; Fried et al. 2004).

In response to concerns about high-severity wildfires, land managers in the western United States are carrying out large-scale "hazardous fuel reduction" thinning programs intended to lower canopy densities and remove wood biomass (USDA 2000). In some cases fuel reduction programs are simultaneously intended to facilitate restoration of ecosystems that have been altered by fire suppression (e.g., USDI 1999). How will native species of grasses and forbs respond to these sudden reductions in canopy cover and associated changes in habitat? When invasive species are present, how effective will they be in colonizing the newly opened sites in competition with native species? What will be the longer-term outcome of these vegetation dynamics in terms of community composition and canopy species regeneration? Where thinning treatments have both fuel-reduction and ecological restoration objectives, to what extent are these goals compatible? To seek answers to these questions I sampled vegetation and abiotic site characteristics along paired transects in thinned and unthinned shrub/chaparral and oak communities of southwestern Oregon (OR). Transects were established retrospectively at sites thinned four to seven years prior to

sampling. This study provides new information that will be useful to ecologists interested in thinning impacts and vegetation dynamics in these and similar communities, and provides information for land managers who want to design thinning treatments that achieve fuel-reduction or restoration goals without creating new problems of invasive species expansion and ecosystem type-conversions.

In southwestern OR an extensive program of fuel-reduction thinning is being applied by the Medford District Bureau of Land Management (BLM) to thousands of hectares of public lands every year (USDI 2002). To manage the costs of fuel reduction treatments, new technologies for thinning are being developed and applied. Mechanical mastication makes use of heavy equipment referred to as a Slashbuster™ and offers a potentially cost-effective alternative to manual chainsaw thinning (USDI 1999). In manual treatments, termed "hand-cut and pile-burn," woody vegetation is chainsaw-cut and piled, and piles are burned during late fall or winter. In mechanical mastication treatments, trees and shrubs are fragmented down to the stump and scattered as a layer of coarse debris. In general, substantially more than 50 percent of canopy cover is removed, with shrub species and small diameter trees targeted for thinning.

In addition to removing canopy cover, thinning treatments impact the ground surface, altering the substrate for herbaceous growth and canopy species regeneration. Bare soil patches, and cover from litter influence the suitability of sites for seed germination and seedling survival, and also affect soil moisture and erosion (Keeley 1992, Whisenant 1999). Pile burning following manual thinning creates intense soil heating that can leave persistent burn-scars of sterilized or chemically altered soil, conditions that may favor the establishment of opportunistic or invasive species (Dickinson and Kirkpatrick 1987; Korb et al. 2004). Wood debris from mechanical mastication can cover much of the soil surface, potentially acting as a physical barrier to seed germination and herbaceous growth. Do mechanical mastication and hand-cut and pile-burn treatments differ in their impacts on herbaceous communities and canopy species regeneration? Although the Medford District BLM has conducted fuel-reduction treatments using both thinning methods on over 7000 hectares of shrub and oak woodlands as of 2006 (USDI 2006), no comprehensive monitoring program has previously been completed to assess vegetation responses. Further, no published studies to date have described the ecological impacts of mechanical mastication as an unprecedented type of disturbance in these ecosystems (but see Sikes 2005). My study made use of this opportunity to contrast the impacts of manual and mechanical thinning methods on site conditions and plant communities, and thus addresses both of these vital information gaps.

There are several types of shrub and oak communities in southwestern OR, including chaparral stands dominated by *Ceanothus* or *Arctostaphylos*, and mixed communities of *Quercus* and shrubs. A rich variety of native grasses and forbs grow in the understory and interspaces of these oak and shrub canopies, as do a large number of exotic and invasive plants. The composition and dynamics of these communities have received little research attention, and no published studies to date address the impacts of thinning treatments on them (but see Sikes 2005). Do these diverse canopy vegetation community types differ in herbaceous species composition, and will they differ in their response to thinning treatments? Results from this study address the importance of accounting for vegetation type differences when examining treatment impacts in these ecosystems, and may provide useful information for developing community and site-specific prescriptions.

Research on chaparral systems emphasizes the fire-adaptations of species characterizing these communities, which both allow persistence after fire and create a dependence on fire for regeneration and renewal of habitat (Keeley and Zelder 1978; Keeley et al. 1981; Odion and Davis 2000; Fried et al. 2004). For example, fire-associated cues stimulate germination for many shrub and forb species associated with chaparral systems (Keeley et al. 1985; Keeley 1987; Keeley and Fotheringham 1998; Borchert 1989). We do not know how fire-adapted native species will respond to canopy removal treatments in which fire is absent or dramatically different from wildfire. A prolonged absence of fire and lack of suitable habitat beneath dense canopies may have depleted seed banks for native grasses and forbs (Borchert 1989), limiting their capacity for re-establishment in thinned sites. By contrast, invasive species tend to colonize disturbed sites rapidly, especially when they were present prior to disturbance (Hobbs and Huenneke 1992; Sheley and Petroff 1999; Davis et al. 2000). Will a history of fire suppression plus the disturbances associated with thinning treatments allow invasive species to overtake thinned sites to the detriment of native communities? Results presented here address this question. As such, this study is an important contribution to the understanding of chaparral and mixed shrub/oak communities and their disturbance ecology.

METHODS

Study Area:

Field research for this study was conducted on USDI BLM lands (Medford District, Ashland Resource Area) within the Applegate River watershed of the Rogue River drainage in southwestern OR, USA (42° N, 122° W). Located in the rain-shadow of the Siskiyou Mountains, the valleys and foothills of the Applegate River watershed have a Mediterranean-type climate of cool, wet winters and hot, dry summers. Average annual precipitation for the centrally-located town of Ruch, OR is 64.4 cm, mostly falling as rain from November through April. Average January low temperature is -1.0° C, and the average July maximum is 32.3° C (WRCC 2006). At lower elevations and southerly aspects, conditions are xeric, and vegetation is characterized by complex mosaics of oak woodlands, shrub/chaparral and open grasslands (Franklin and Dyrness 1973).

Ceanothus cuneatus, *Arctostaphylos viscida* and *Quercus garryana* mix with each other and, to a lesser degree, with *Pinus ponderosa*, *Arbutus menziesii*, and *Pseudotsuga menziesii* in these chaparral and woodland communities (Detling 1961; plant nomenclature follows Hickman 1993). For the drier south-facing slopes on which this study focuses, native grasses are primarily perennial bunchgrasses, including *Achnatherum lemmonii*, *Bromus carinatus*, *B. laevipes*, *Elymus glaucus*, *E. elymoides*, *Festuca idahoensis* and *F. californica*. There is also a strong presence of non-native invasive annual grasses, including *Avena fatua*, *Bromus tectorum*, *B. diandrus*, *B. hordeaceus*, *B. madritensis*, *B. japonicus*, *B. sterilis*, *Taeniatherum caput-medusae* and *Cynosurus echinatus* (Pyke 1999; Whitson et al. 2004). Weedy non-native forbs of concern in the watershed include *Centaurea solstitialis*, *Cirsium vulgare*, *Erodium cicutarium*, *Hypericum perforatum*, and *Torilis arvensis* (Pyke 1999, Sheley and Petroff 1999). Both annual and perennial native forbs are abundant and diverse; especially common are *Madia*, *Clarkia* and *Lomatium* species, and drought-tolerant members of the *Boraginaceae*, *Fabaceae*, *Liliaceae* and *Asteraceae* families. (For a full species list, see Appendix 1.)

The area's disturbance history includes widespread grazing, small-scale mining, logging, wildfire, intentional use of fire by settlers and Native Americans to clear land, and ongoing fire-suppression (Agee 1993; USDI 1999; P. Hosten, Medford BLM, pers. comm.). Current land ownership follows a checkerboard pattern of public land interspersed with private properties. Recent expansion of residential development into rural and forested areas led to the designation of > 55,600 hectares of Medford District BLM lands as Rural Interface Areas by 2001 (USDI 2002). These ownership patterns combined with recent wildfires have

heightened concerns about fire risks amongst local residents and land managers, leading to the landscape-scale application of fuel reduction thinning treatments beginning in the mid-1990's (USDI 1999, Applegate Partnership 2002). This study was conducted in some of the over 2000 hectares of chaparral and mixed oak vegetation treated for hazardous fuels between 1996 and 2001 within the Ashland Resource Area (USDI 2002). Within some vegetation communities and at particular sites, BLM objectives include both fuel-reduction and restoration of native vegetation communities. However, at other treatment sites, especially within chaparral vegetation, BLM management is focused almost exclusively on fuel-reduction goals (P. Hosten, Medford BLM, pers. comm.).

Field Methods:

Sampling occurred in sites representative of the area's mixed chaparral and oak communities. Selected sites had canopies dominated by varying mixtures of *Arctostaphylos viscida*, *Ceanothus cuneatus*, and *Quercus garryana*. Sites were field-assigned into one of three canopy vegetation community groups: 1) *A. viscida*-dominated (ARC), 2) *C. cuneatus*-dominated (CEA), or 3) mixed *Q. garryana* and shrub (MIX) with both shrub species well-represented. Sampled sites ranged from approximately 500 to 1000 m elevation, and were on southeast- to southwest-facing slopes with aspects ranging from 105° to 275° and inclines between 10° and 35°.

Both hand-cut and pile-burn (HPB) and mechanical mastication (MM) thinning treatments were sampled. My sampling was limited to sites thinned between May 1998 and June 2001. This allowed for an extended period of post-thinning vegetation response and inclusion of a reasonably large and well-dispersed number of samples, while restricting the range of years to minimize the influence of temporal trends in responses. Thinning prescriptions were dictated by BLM land management priorities focused on fuel and canopy density reductions, and did not include provisions for research or monitoring. Treatments were applied non-randomly and unequally in terms of site, timing and thinning method. Thus, a major challenge for this study was to design a sampling strategy that minimized the likelihood that treatment effects would be confounded by variations due to landscape location and temporal factors.

No pre-thinning vegetation data were available, hence this study was retrospective and based on comparisons between paired thinned and unthinned transects. Field matching of transects was based on within-pair similarity in: canopy species structure and composition (relying on stumps and leave-trees in thinned areas); slope and aspect; landscape features; and proximity to roads or all-terrain-vehicle (ATV) tracks. Potential sites were identified using

BLM fuels-management records in the form of a GIS treatment-unit spatial database (USDI 2005). I combined GIS data with digitized aerial photographs in ArcMap (Version 8.0; ESRI 2002) to allow identification of potentially suitable units, all of which were surveyed on foot during May-June 2005. Paired transects were established based on the presence of acceptably matched thinned and unthinned areas of vegetation of at least 1800 m² each, and an absence of recent disturbances other than thinning. Only one pair of transects was established per vegetation community in a given treatment unit. I established and sampled a total of 32 transect pairs during June and July of 2005 (Appendices 2 and 3).

At each sample site, I located 50-m transects at least 10 m (usually \geq 15 m) from treatment edges or natural boundaries, avoiding atypical vegetation or topographic features. Paired transects followed approximately the same bearing (\pm 20°). Whenever possible transects were continuous, however the spatial configuration of some sites made it necessary to divide transects into two or more parallel segments \geq 10 m apart. Typically, transects within a pair were \leq 50 m apart. Slope and aspect were measured at the mid-point of each transect.

Over the course of the summer, I sampled sites from lower to higher elevations to minimize phenological variation. Five 3-m² canopy measurement plots per transect were located according to constrained randomization, such that one circle fell within each of the five consecutive 10-m transect intervals. Within each canopy plot I recorded percent cover, as cover classes, and stem counts for all tree, shrub and other woody perennial species. Cover classes approximated an arcsine square root transformation (Muir and McCune 1987): 0 (none); 1 (<1%); 2 (1-5%); 3 (5-25%); 4 (25-50%); 5 (50-75%); 6 (75-95%); 7 (95-99%); 8 (>99%). For canopy species, individuals were recorded separately according to height classes (<0.3 m; 0.3-1 m; 1-2 m; 2-3 m; >3 m) and condition classes (dead, stump, or resprouting stump). I recorded cover, but did not count stems for woody groundcover and vining species within canopy plots. Percent cover of burn-pile scars and unburned piles was recorded as encountered. Densimeter readings of percent canopy cover were taken one meter above the ground at the center of each plot.

One 1000-cm² herbaceous cover plot was nested within each of two different randomly assigned quadrants of each canopy plot, for a total of 10 herbaceous cover plots per 50 m transect. Within these plots, I recorded percent cover, as above, by species for all grasses and forbs, and for trees, shrubs and other woody perennial species < 0.3 m tall (hereafter collectively referred to as "herbaceous" vegetation). Total canopy or herbaceous cover sometimes exceeded 100% due to vegetation layering. Within each herbaceous plot, I also recorded substrate cover classes for litter, bare soil, and down wood as fuel-hour based diameter categories (<1 cm; 1-3 cm; 3-10 cm; >10 cm).

Analysis Methods:

Cover classes were converted to percent cover as the arithmetic mid-points of the cover class intervals. Data collected at multiple plots or points within a transect were averaged. Data on species cover and abiotic variables tended to have non-normal distributions or unequal variances that did not meet parametric assumptions and could not be fully corrected with data transformations. As a result, I used non-parametric statistical tools for multivariate analyses (PC-ORD Version 5.35 beta; McCune and Mefford 1999; McCune and Grace 2002) and univariate tests (S-Plus Version 7.0; Insightful 2005). Species occurring in fewer than five percent of transects were dropped from analyses, as were two transect pairs that contained outliers with average compositional dissimilarity > three standard deviations from the mean (McCune and Grace 2002); the resultant herbaceous community matrix contained percent cover data for 95 species in 60 paired transects.

Data analyses were conducted both at the level of individual transects, and with data expressed as the difference between transects within each pair. Within-pair differences indicate responses to treatments (within limits of the retrospective study design), and were calculated by subtracting data for the unthinned transect from that of the corresponding thinned transect. All species are effectively given equal weighting when data are expressed as differences, focusing the analysis on changes in species composition across the entire community regardless of each species' absolute abundance. Percent cover and stem count data were \log_{10} -transformed prior to subtraction so that within-pair differences represented the order-of-magnitude change for each species relative to its total abundance within the pair. These subtractions produced matrices of within-pair differences in species composition and environmental variables; thus sample sizes were reduced by half.

All species were categorized as native or exotic and annual or perennial, and further classified into functional groups based on trait and life-cycle information (Hickman 1993). Functional groups used in this analysis were: native annual forbs, native perennial forbs, exotic annual forbs, exotic annual grasses, and native perennial grasses (see Appendix 1 for a full list of functional groups with species assignments). For each transect, I summed percent cover of all species within each category or functional group to yield summary measures of cover by category and functional group. Percent cover of all herbaceous species was summed per transect to obtain total herbaceous cover. Functional group and category cover values were divided by total herbaceous cover within each transect to calculate proportional measures of functional group or category dominance. Species richness and the Shannon-

Weiner diversity index were calculated for each transect, and for native species within each transect, using PC-ORD.

Percent canopy cover from the canopy plots for all species and all size classes > 0.3 m tall were summed per transect to give an herbaceous-layer perspective of canopy cover appropriate to the low canopy profiles of shrub species. Transect averages for densiometer-based percent canopy cover readings provided a larger-scale and more traditional measure of canopy cover. Average stem counts for size classes > 1 m tall were summed across all canopy species to estimate stand densities per transect. Within each transect, average stem counts for all seedlings, resprouting stumps and plants < 1 m tall were summed separately for oak, conifer and shrub species as measures of canopy species regeneration (a resprouting stump = one stem). Down wood percent cover for all size classes > 1 cm diameter was summed per transect as a measure of coarse wood debris. Slope and aspect were integrated with latitude to calculate a heatload index for each transect (McCune and Keon 2002).

Assignment of transect pairs to canopy vegetation community (ARC, MIX or CEA) and treatment type (HPB or MM) groups allowed for data analysis at three levels: 1) To assess overall treatment impacts, the differences between paired thinned and unthinned transects were compared across all treatment types and canopy vegetation communities using the entire data set of 30 transect pairs. 2) To determine whether there were differences in treatment effects between or among groups, the 30 transect pairs were sorted by treatment or vegetation type. 3) Possible differences in responses between treatment types within a given canopy vegetation type, or among vegetation types to which the same treatment method was applied, were examined by sorting pairs into all possible canopy vegetation group and treatment type combinations (Table 1).

To address the basic question, "Was there a treatment effect?" I used a blocked version of multiple-response permutation procedure (MRBP; Euclidean distance) to compare herbaceous cover data between thinned and unthinned transects blocked as pairs. Multiple response permutation procedure (MRPP) is a non-parametric test for multivariate difference between two or more pre-defined groups (Zimmerman et al. 1985). MRPP and MRBP give both a p-value based on randomized group reassignments and an A-value measure of within-group homogeneity; when $A = 1$ all items are identical within groups and when $A = 0$ homogeneity within groups is the same as expected by chance (McCune and Grace 2002). Three alternative data transformations were used, each emphasizing a different perspective. With raw data, the MRBP test was most influenced by percent cover differences between highly abundant species and total cover differences between transects. With \log_{10} -transformed data, additional weight is given to less abundant species, and MRBP

comparisons are less affected by dominant species and differences in total cover, compared to raw data. MRBP on data relativized by transect totals clarified the extent to which thinned and unthinned sites differed in species composition by removing the influence of absolute percent cover differences between transects.

To characterize treatment effects in terms of vegetation and abiotic variables, I applied a two-step process of multivariate ordinations followed by univariate tests. I chose nonmetric multidimensional scaling (NMS) ordination for its ability to work with data that do not meet parametric or linear assumptions (Kruskal 1964, McCune and Grace 2002). NMS ordinations were run on transect-level herbaceous cover data using Sørensen distance and the "Slow and Thorough" autopilot program setting (PC-ORD Version 5.35 beta; McCune and Mefford 1999). Three-dimensional solutions were accepted for all transect ordinations because they consistently had much lower stress (a measure of solution fit to the original data; McCune and Grace 2002) than one or two dimensional solutions, and had significantly lower stress than solutions based on randomized data (Monte Carlo test $p = 0.004$ for raw, \log_{10} and relativized data). NMS solutions arranged transects as points in a graphical summary such that distances between points represented their degree of similarity in species composition. Solutions were rotated to maximize separation of thinned and unthinned transects along a single axis. I examined non-parametric correlations (Kendall's tau) of vegetation attributes, functional groups and abiotic variables with that axis. Finally, the significance of differences in individual variables between thinned and unthinned paired transects was assessed using Wilcoxon sign-rank tests on within-pair difference data.

Vector lines were added to the ordination of raw herbaceous cover data to connect paired thinned and unthinned transects, using PC-ORD. Vector lines in this context are a measure of herbaceous composition differences between transects with respect to the strongest community gradients as summarized in multidimensional ordination-space. The relative magnitude of treatment effects was compared between vegetation and treatment type groups using lengths of these vectors as indicators and Wilcoxon rank-sum (between two groups) and Kruskal-Wallis (among three groups) univariate tests.

For comparisons of treatment effects between canopy vegetation groups or treatment types, it was important to first establish whether these differed in herbaceous composition in the absence of treatments. To determine this, I used MRPP tests (Sørensen distance) on both raw and \log_{10} -transformed cover data for unthinned transects grouped as HPB or MM, and as ARC, MIX or CEA, with the null hypothesis of no difference between groups. To assess multivariate differences in treatment effects between treatment type and vegetation groups, I performed a series of MRPP tests (using Euclidean distances because some values

were negative) on herbaceous within-pair difference data. To test the significance of differences in treatment effects between treatment types, I used Wilcoxon rank-sum tests on within-pair difference data for individual vegetation and abiotic variables.

Where MRPP tests on all 30 pairs showed significant differences between groups, subsets of pairs sorted by both treatment type and vegetation group were further analyzed. MRPP analysis of these subsets allowed for multivariate comparisons of treatment type differences in herbaceous community response while controlling for canopy vegetation group (Table 1). The statistical power of analyses at the pair-subset level was limited by small sample sizes (Table 1). Because of this, univariate analyses of differences in vegetation or abiotic variable responses between treatment types at the pair-subset level are not presented here (but see Appendix 4).

Table 1. Summary of hierarchical levels of analysis, with corresponding data sets and subsets, group comparisons and sample sizes. Treatment types abbreviated as: MM = mechanical mastication; HPB = hand-cut and pile-burn. Canopy vegetation types abbreviated as: ARC = *A. viscida*-dominated; MIX = mixed *Quercus* and shrub; CEA = *C. cuneatus*-dominated.

Analysis level	Data set and sample size	Group comparison	Sample sizes per group
<u>1</u>	All transects (60)	thinned vs. unthinned	30, 30
<u>2</u>	All pairs (30)	HPB vs. MM	12, 18
	All pairs (30)	ARC vs. MIX vs. CEA	10, 11, 9
<u>3</u>	HPB pairs (12)	ARC vs. MIX vs. CEA	5, 3, 4
	MM pairs (18)	ARC vs. MIX vs. CEA	5, 8, 5
	ARC pairs (10)	HPB vs. MM	5, 5
	MIX pairs (11)	HPB vs. MM	3, 8
	CEA pairs (9)	HPB vs. MM	4, 5

RESULTS

A total of 126 herbaceous species were recorded for 30 transect pairs; 101 were native and 25 were non-native exotics (Appendix 1). Ninety-five species occurred in at least five percent of transects. Whittaker's beta diversity (McCune and Grace 2002) was 3.4, with mean species richness (S) per transect of 28.6 and mean transect Shannon-Weiner (H') diversity of 2.59. Species richness and average Shannon-Weiner diversity were essentially the same for thinned and unthinned areas (S = 112 and 111, H' = 2.56 and 2.62 respectively). Canopy cover (densiometer-based) ranged between 40% and 93% for unthinned sites, and from 1.5% to 61% at thinned sites. At unthinned sites, herbaceous cover varied from 24% to 126%, compared to a range of 45% to 168% for thinned sites.

Ten of the 30 transect pairs were field-assigned to the *A. viscida*-dominated canopy vegetation group (ARC), 9 to *C. cuneatus*-dominated (CEA) and 11 to mixed *Quercus* and shrub (MIX). Herbaceous species composition differed significantly among these vegetation groups in the absence of treatments ($p < 0.05$ from MRPP, unthinned transects only), except for between ARC and MIX using raw data (Table 2). Twelve pairs were HPB thinned and 18 were MM thinned. Herbaceous composition did not differ significantly between unthinned transects grouped as HPB and MM (minimum $p > 0.07$ from MRPP; Table 2).

Overall Treatment Effect – Thinned vs. Unthinned Sites:

Site conditions on thinned transects differed significantly (Wilcoxon sign-rank, $p < 0.05$) from those on unthinned transects by a number of key measures, with most differences attributable to treatments. Mean canopy cover of all woody species > 0.3 m tall was 71% lower (range of 0.1% to 125%) on thinned sites than on unthinned sites. Mean stem density for trees and shrubs > 1 m tall (excluding stumps) was 3.3 stems per 100 m² in thinned areas, compared to an average of 40 stems per 100 m² in unthinned areas. Percent cover of wood debris > 1 cm in diameter averaged $> 11\%$ more across all thinned sites, and $> 16\%$ more at MM thinned sites (range of 5% to 29%) compared to matched unthinned areas. Burn pile scars covered an average of 17% of the ground surface at HPB thinned sites (range of 0% to $> 40\%$), and mean cover by unburned wood piles at these sites was an additional $\sim 6\%$ (range of 0% to 28%). Mean cover by litter and bare soil was 7% and 9% lower, respectively, on thinned compared to unthinned sites across both treatment types ($p = 0.056$ for bare soil difference).

There was strong evidence of an overall treatment effect on herbaceous species composition when differences between thinned and unthinned transects were examined

across all pairs. Herbaceous composition differed significantly between matched thinned and unthinned transects whether tests were based on raw data or data relativized by transect totals (MRBP, Table 3), indicating that differences were due to changes in species composition, not simply to a general increase in cover. Communities in thinned and unthinned transects also differed when data were \log_{10} -transformed to increase weighting of less abundant species, but the within-group similarity (A-value) declined almost three-fold following this transformation (Table 3).

NMS ordination of raw herbaceous composition data provided a three-dimensional solution that showed clear separation of thinned and unthinned transects and represented 76.9% of the variation in the original data (final stress = 16.2; Fig. 1). Most of the variation in species composition was summarized by Axis 3 (45.5%), the axis that separated thinned (low axis scores) and unthinned (high axis scores) transects. Ordinations based on \log_{10} -transformed or relativized species cover data gave similar solutions, although separation of thinned and unthinned transects was less clear with log-transformed data (ordinations not shown).

Numerous vegetation variables were strongly correlated ($|\text{tau}| > 0.50$) with Axis 3 (Fig. 1, Table 4). Total herbaceous cover, and cover of native species, annual species, exotic annual grasses and native annual forbs were strongly negatively correlated with this axis, indicating higher cover at thinned sites; cover of exotic species was also negatively correlated with Axis 3. In contrast, cover of perennial species tended to be higher at unthinned sites, and cover of native perennial grasses had a similar, but weaker positive correlation. Cover of exotic annual forbs, native perennial forbs, and native species richness and native diversity were not correlated with Axis 3 (Table 4).

When vegetation variables were expressed in terms of community dominance, some of the same patterns persisted, but different patterns also emerged (Fig. 1, Table 4). As a proportion of total herbaceous vegetation, perennial species were strongly positively correlated with Axis 3 (higher at unthinned sites), while the proportion of annual species was strongly negatively correlated with that same axis (higher at thinned sites). Native perennial grasses were more strongly positively correlated with Axis 3 as proportions than as cover. Exotic annual forbs and native perennial forbs, which were not correlated as percent cover, were both positively correlated with Axis 3 as proportions. Positive correlations indicate greater dominance by these functional groups at unthinned sites. As proportions of the herbaceous vegetation, both exotic annual grasses and native annual forbs showed more dominance at thinned sites (Table 4). When expressed as proportions, neither native nor

exotic species, as overall categories, were correlated with the axis separating thinned and unthinned sites.

Single-variable tests for differences in vegetation categories and functional groups between thinned and unthinned transects (Table 5) paralleled these patterns. (Because multiple comparisons were made within the same data set, probabilities from univariate statistical tests must be interpreted with caution.) Total herbaceous cover was significantly greater on thinned than unthinned sites (mean of 34% greater, range from 117% to -42%). At thinned sites, mean percent cover of native and annual species, native annual forbs and exotic annual grasses was significantly higher compared to unthinned sites, with suggestive evidence of greater cover of exotic species at thinned sites as well ($p = 0.074$; Table 5). Mean exotic annual grass cover was > 11% higher at thinned than unthinned sites (range of 62% greater to 14% less). Perennial species, native perennial forbs and exotic annual forbs all had significantly lower cover at thinned sites. There was no significant difference in native perennial grass cover between thinned and unthinned transects.

The proportions of total herbaceous cover comprised of native or exotic species did not differ between thinned and unthinned transects (Table 5). Perennial species, however, represented a proportionally greater component of the herbaceous community in unthinned than in thinned areas, while dominance by annuals, exotic annual grasses and *Vulpia microstachys* (the only native annual grass occurring in my study area) was significantly higher at thinned sites. Proportional cover of both exotic annual forbs and native perennial forbs was lower at thinned sites, while native annual forbs showed over 17% more proportional cover in thinned areas. The proportion of herbaceous cover composed of native perennial grasses did not differ between paired thinned and unthinned transects, even though it was positively correlated with Axis 3 in the ordination (Fig. 1), suggesting higher proportional cover values at unthinned sites.

Shannon-Weiner diversity and species richness overall, as well as native species diversity and richness did not differ between thinned and unthinned sites (all p -values > 0.10; Table 5). Regeneration of *Arctostaphylos* and *Ceanothus* shrubs (seedlings and plants < 1 m tall) was significantly greater in thinned than unthinned sites, despite a lack of correlation with Axis 3, but the absolute difference was small (< 1.4 additional stems per 3 m² canopy plot at thinned sites). Regeneration of oak or conifer species did not differ significantly between thinned and unthinned sites (Table 5). However, there was suggestive evidence ($p = 0.07$) of greater oak regeneration at unthinned sites, and regeneration of oak species was positively correlated with Axis 3 of the transect ordination, suggesting a tendency for higher cover of oak seedlings in unthinned stands compared to thinned (Table 4).

Response Differences Between Treatment Types and Among Vegetation Groups:

There was little evidence that herbaceous responses differed between MM and HPB treatments when analysis did not account for canopy vegetation groups. The MRPP test comparing pair-difference herbaceous species data between treatment types across all pairs resulted in a tiny A-value (0.006) that was not statistically significant (Table 6). Similarly, no differences were found in the magnitude of treatment effect between HPB and MM across all pairs, as measured by length of transect ordination vectors (Table 6). For ease of communication, I will hereafter use the terms “increase” and “decrease” to describe positive and negative thinned minus unthinned within-pair differences, respectively, while acknowledging the lack of certainty about the nature of apparent changes that is inherent to retrospective data.

There was evidence of differences in abiotic variable responses between treatment types across all pairs. HPB sites had significant increases in burn-scar cover that MM sites lacked, and MM sites had larger within-pair increases in wood debris and decreases in litter than did HPB sites (rank-sum tests, all $p < 0.05$; Table 7). However, these between-treatment differences in substrate conditions did not translate into detectable differences in treatment effects on vegetation variables at the all-pairs level of analysis. HPB and MM treatments were indistinguishable in terms of within-pair differences in total herbaceous cover, species diversity and richness, vegetation categories or functional groups (both as cover and proportions), or canopy species regeneration (rank-sum tests, minimum $p > 0.14$; Table 7). (Note: When these tests were run on raw difference data, rather than data \log_{10} -transformed prior to subtraction as presented (see analysis methods), results were consistent, with one important exception: Within-pair increases in raw cover of exotic annual grasses were significantly greater (19%) at HPB compared to MM sites (rank-sum $p = 0.02$), and proportional cover of exotic annual grasses increased almost 12% more at HPB sites (rank-sum $p = 0.01$).)

In some cases, herbaceous responses differed between canopy vegetation groups when compared across treatment types. Herbaceous vegetation responses to treatments differed significantly between ARC and CEA groups, as judged by MRPP tests of within-pair difference data for all pairs (Table 6). MRPP differences between CEA and MIX were suggestive, and there was no evidence of differences in responses between ARC and MIX. Both ARC and CEA groups showed significantly larger treatment-effects than MIX, as measured by the lengths of ordination vectors; but there was no difference in magnitude of response between ARC and CEA groups (Table 6). When herbaceous community responses to treatments were compared between canopy vegetation groups within treatment type

subsets, strong differences were detected among and between all vegetation groups within the HPB subset, but responses to MM treatment were indistinguishable among vegetation groups (MRPP on within-pair difference data, Table 8). However, it is difficult to interpret these comparisons of treatment effects between canopy vegetation groups, because herbaceous composition also differed significantly between these groups in the absence of thinning both at the all-pairs level of analysis and within the HPB treatment subset (MRPP on unthinned transect data; Tables 2 and 8).

Response Differences Between Treatment Types Within Vegetation Groups:

Because vegetation groups differed in herbaceous composition at unthinned sites and seemed to differ in their response to treatments, I compared HPB and MM thinning treatments within vegetation group subsets. There were small sample sizes for treatment type comparisons within canopy vegetation groups, but differences in herbaceous community responses to HPB and MM treatments were clearly indicated within both ARC and CEA subsets (MRPP; Table 8). It is likely that these subset results reflect actual differences in herbaceous responses to HPB and MM treatments, given that herbaceous composition of unthinned transects did not differ significantly between treatment types within canopy vegetation group subsets (MRPP; minimum p-values > 0.10 for raw data, and > 0.06 for \log_{10} data; Table 8). While these apparent differences in herbaceous responses to HPB versus MM treatments within canopy vegetation groups deserve attention, further analysis and interpretation of differences is not presented here because sample sizes are so small within subsets (Table 8) that results cannot be counted on to represent more general patterns of treatment effect differences. (See Appendix 4 for an exploratory presentation of methods and results for univariate analyses of these subsets.)

Table 2. MRPP tests of herbaceous composition differences between unthinned transects grouped by canopy vegetation or treatment type. Results are given for tests based on both raw and \log_{10} -transformed herbaceous cover data (see analysis methods). Bold face text highlights significant differences ($p < 0.05$) and corresponding A-values. See Table 1 for definitions of treatment type and canopy vegetation group abbreviations.

Group comparison	Transects per group	MRPP on raw data		MRPP on \log_{10} data	
		p	A	p	A
HPB : MM	12, 18	0.282	0.003	0.073	0.011
ARC : MIX : CEA	10, 11, 9	0.000	0.048	0.000	0.068
ARC : CEA	10, 9	0.000	0.054	0.000	0.085
ARC : MIX	10, 11	0.080	0.015	0.011	0.030
CEA : MIX	9, 11	0.002	0.043	0.001	0.047

Table 3. MRBP tests comparing herbaceous community composition between thinned and unthinned sites. Tests based on all 60 transects blocked as pairs; results presented based on raw, \log_{10} -transformed, and relativized herbaceous species percent cover data.

Herbaceous data transformation	MRBP for thinned vs. unthinned	
	p	A
Raw data	0.000	0.159
\log_{10} -transformed	0.000	0.056
Relativized by transect totals	0.000	0.152

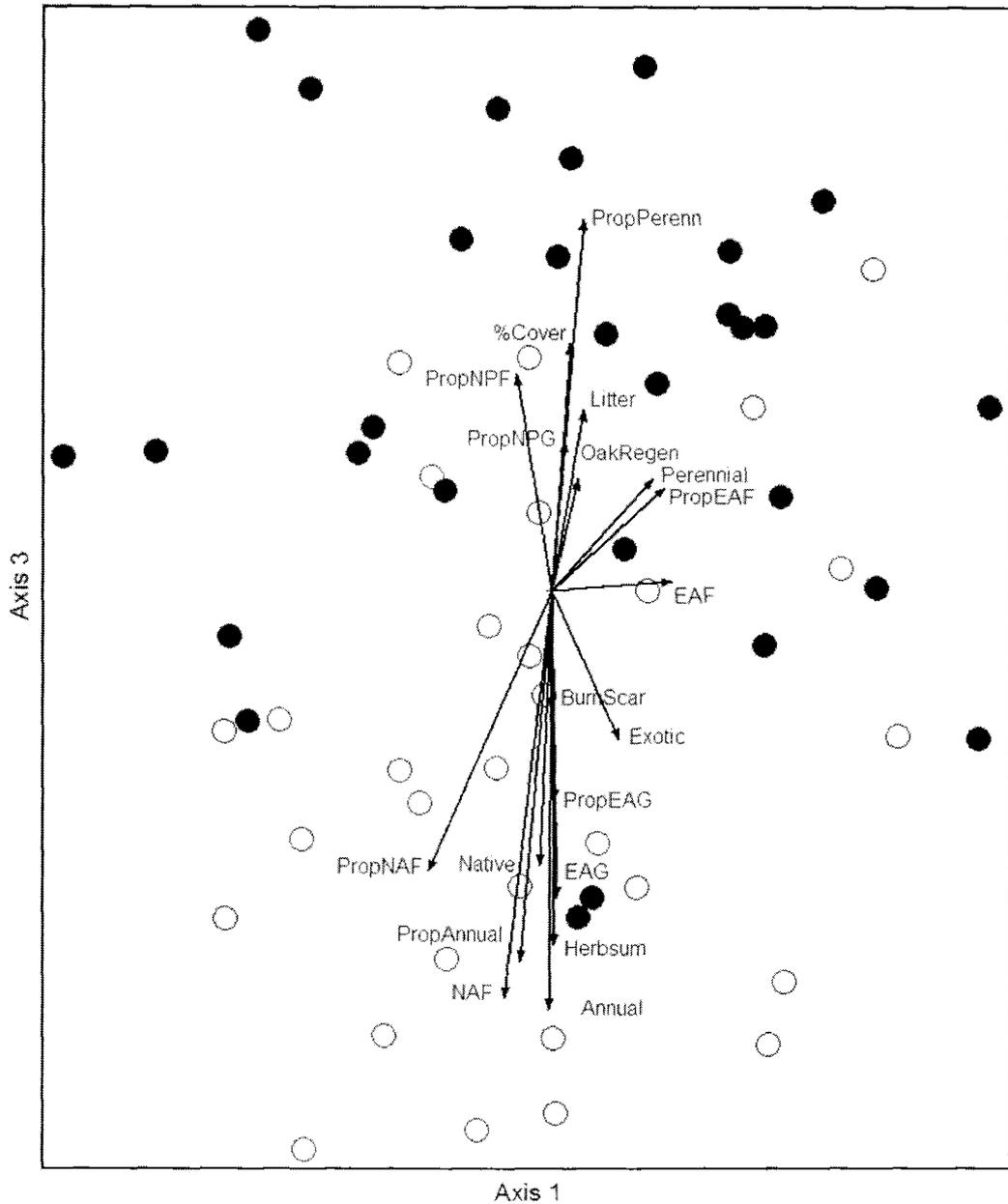


Figure 1. NMS ordination of transects based on raw herbaceous species percent cover data. Unthinned transects (solid points) separate from thinned transects (hollow points) along Axis 3. Overlay joint lines show correlations of variables with ordination axes; arrows indicate the region of transects with the highest values for each variable, and joint line lengths are scaled proportional to the strength of correlation. See Table 4 for full names of abbreviated variables and Kendall's tau values for correlations with Axis 3.

Table 4. Variable correlations (Kendall's tau) with Axis 3 of the raw herbaceous cover ordination (Fig. 1). Axis 3 tended to separate thinned and unthinned transects such that when tau is positive, higher values for the variable are associated with unthinned sites; when tau is negative, the variable has higher values associated with thinned sites. Variables with $|\text{tau}| \geq 0.30$ are reported as "correlated" and are in bold face text; variables with $|\text{tau}| \geq 0.50$ were considered "strongly correlated" and are bolded and underlined in the table. A key to abbreviations for variables used in the following tables and in Figure 1 is included.

Variable name (as given in text)	Abbreviated as:	tau
Canopy cover (woody species > 0.3 m)	% Cover	0.489
Wood debris cover	WoodDebris	-0.147
Burn scar cover	BurnScar	-0.334
Litter cover	Litter	0.404
Bare soil cover	Soil	-0.036
Total herbaceous cover	Herbsum	<u>-0.604</u>
Species richness	Richness	-0.245
Native species richness	NatRichness	-0.101
Shannon-Weiner diversity	Diversity	-0.057
Native Shannon-Weiner diversity	NatDiversity	0.120
Native species cover	Native	<u>-0.527</u>
Exotic species cover	Exotic	-0.320
Annual species cover	Annual	<u>-0.682</u>
Perennial species cover	Perennial	0.328
Exotic annual grass cover	EAG	<u>-0.570</u>
Native perennial grass cover	NPG	0.259
Exotic annual forb cover	EAF	0.092
Native annual forb cover	NAF	<u>-0.666</u>
Native perennial forb cover	NPF	0.192
Proportion of native species	PropNative	0.107
Proportion of exotic species	PropExotic	-0.107
Proportion of annual species	PropAnnual	<u>-0.664</u>
Proportion of perennial species	PropPerenn	<u>0.664</u>
Proportion of exotic annual grasses	PropEAG	-0.480
Proportion of native perennial grasses	PropNPG	0.371
Proportion of exotic annual forbs	PropEAF	0.315
Proportion of native annual forbs	PropNAF	<u>-0.522</u>
Proportion of native perennial forbs	PropNPF	0.435
Oak regeneration	OakRegen	0.314
Conifer regeneration	ConRegen	0.328
<i>A. viscida</i> and <i>C. cuneatus</i> regeneration	ShrubRegen	-0.090

Table 5. Comparison of differences in means between thinned and unthinned paired transects for key variables. See Table 4 for definitions of the variable abbreviations. P-values are from Wilcoxon sign-rank tests of within-pair difference data (thinned minus unthinned), with percent cover and stem count data \log_{10} -transformed prior to subtraction (see analysis methods). Means and standard deviations (SD) presented are from raw data, and are percentages, except for regeneration variables (stem counts) and measures of species richness and diversity. Significant p-values (< 0.05) and corresponding means and standard deviations are in bold; italics indicate suggestive p-values.

Variable	p	Mean difference	Thinned mean	Thinned SD	Unthinned mean	Unthinned SD
% Cover	0.000	-71.25	25.38	18.07	96.63	18.79
WoodDebris	0.000	11.56	15.76	9.35	4.20	2.27
BurnScar	0.003	6.83	6.83	12.56	0.00	0.00
Litter	0.028	-7.41	51.84	10.93	59.25	16.20
Soil	<i>0.056</i>	<i>-9.18</i>	<i>10.84</i>	<i>5.82</i>	<i>20.03</i>	<i>14.43</i>
Herbsum	0.000	33.61	102.90	29.15	69.27	25.51
Richness	0.380	0.70	28.93	5.33	28.23	7.12
NatRichness	0.951	0.00	21.97	4.47	21.97	4.99
Diversity	0.622	-0.06	2.56	0.29	2.62	0.34
NatDiversity	0.139	-0.13	2.30	0.30	2.43	0.33
Native	0.000	27.58	74.59	22.37	47.01	15.81
Exotic	<i>0.074</i>	<i>6.03</i>	<i>28.29</i>	<i>21.05</i>	<i>22.26</i>	<i>15.59</i>
Annual	0.000	38.76	82.42	32.74	43.66	25.09
Perennial	0.016	-5.14	20.47	9.37	25.61	11.26
EAG	0.000	11.31	19.50	18.93	8.19	8.76
NPG	0.165	2.36	6.51	7.15	4.15	5.79
EAF	0.032	-5.14	8.65	6.58	13.79	12.28
NAF	0.000	27.48	46.72	22.43	19.24	12.49
NPF	0.028	-2.26	6.18	4.07	8.43	4.76
PropNative	0.181	3.53	73.66	14.35	70.13	15.43
PropExotic	0.435	-3.53	26.34	14.35	29.87	15.43
PropAnnual	0.000	18.14	77.47	14.17	59.33	19.59
PropPerenn	0.000	-18.14	22.53	14.17	40.67	19.59
PropEAG	0.001	6.18	17.12	13.15	10.94	11.28
PropNPG	0.959	1.02	7.60	9.12	6.58	8.80
PropEAF	0.000	-9.31	9.10	7.02	18.41	12.28
PropNAF	0.000	17.60	44.08	16.36	26.47	13.51
PropNPF	0.028	-8.16	6.30	4.31	14.46	9.11
OakRegen	<i>0.073</i>	<i>-0.39</i>	<i>1.73</i>	<i>1.89</i>	<i>2.13</i>	<i>2.04</i>
ConRegen	0.415	-0.03	0.03	0.09	0.05	0.14
ShrubRegen	0.000	1.37	1.67	2.32	0.30	0.42

Table 6. Tests for differences in treatment effects between treatment types and among canopy vegetation groups. Numbers of pairs per group are indicated for each test. MRPP tests compare community responses as the within-pair differences in herbaceous species percent cover between and among groups. Wilcoxon rank-sum and Kruskal-Wallis tests compare lengths of vectors connecting paired thinned and unthinned transects in ordination space as a unitless measure of the magnitude of treatment effect (see analysis methods). Significant p-values (< 0.05) and corresponding A-values are in bold. See Table 1 for definitions of treatment type and canopy vegetation group abbreviations.

Group comparison	Pairs/group	Response measure	Test	p	A
HPB : MM	12, 18	Within-pair difference	MRPP	0.084	0.006
HPB : MM	12, 18	Vector length	Rank-sum	0.228	
ARC : MIX : CEA	10, 11, 9	Within-pair difference	MRPP	0.023	0.013
ARC : MIX : CEA	10, 11, 9	Vector length	Kruskal-Wallis	0.027	
ARC : CEA	10, 9	Within-pair difference	MRPP	0.025	0.013
ARC : CEA	10, 9	Vector length	Rank-sum	0.775	
ARC : MIX	10, 11	Within-pair difference	MRPP	0.150	0.006
ARC : MIX	10, 11	Vector length	Rank-sum	0.038	
CEA : MIX	9, 11	Within-pair difference	MRPP	0.072	0.012
CEA : MIX	9, 11	Vector length	Rank-sum	0.015	

Table 7. Comparison of mean within-pair differences for key variables between MM and HPB treatments across canopy vegetation groups. P-values are from Wilcoxon rank-sum tests of within-pair difference data (thinned minus unthinned), with percent cover and stem count data \log_{10} -transformed prior to subtraction (see analysis methods). Means and standard deviations (SD) presented are from raw data, and are percentages, except for regeneration variables (stem counts) and measures of species richness and diversity. Significant p-values (< 0.05) and corresponding mean differences and standard deviations are in bold. See Table 4 for definitions of the variable abbreviations.

Variable	p	Mean MM difference	Mean MM dif. SD	Mean HPB difference	Mean HPB dif. SD
% Cover	0.916	-69.81	23.46	-73.41	25.90
WoodDebris	0.001	16.79	7.73	3.71	4.21
BurnScar	0.000	0.00	0.00	17.07	15.00
Litter	0.029	-12.15	12.58	-0.31	19.92
Soil	0.882	-9.38	14.55	-8.90	16.99
Herbsum	0.386	27.39	35.87	42.95	44.56
Richness	0.382	-0.22	6.59	2.08	8.14
NatRichness	0.610	-0.50	4.90	0.75	5.88
Diversity	0.363	-0.08	0.37	-0.02	0.49
NatDiversity	0.597	-0.14	0.37	-0.12	0.48
Native	0.434	27.95	26.51	27.02	23.27
Exotic	0.434	-0.57	15.02	15.93	27.38
Annual	0.719	34.53	36.21	45.10	44.48
Perennial	0.849	-7.14	10.30	-2.15	10.50
EAG	0.197	3.71	9.02	22.71	22.36
NPG	0.341	1.34	3.92	3.88	6.01
EAF	0.626	-4.34	11.03	-6.34	11.47
NAF	0.539	27.57	22.14	27.35	22.52
NPF	0.849	-2.45	6.41	-1.97	2.96
PropNative	0.280	5.24	11.76	0.97	10.45
PropExotic	0.983	-5.24	11.76	-0.97	10.45
PropAnnual	0.916	19.80	21.41	15.66	17.65
PropPerenn	0.341	-19.80	21.41	-15.66	17.65
PropEAG	0.280	1.45	9.82	13.27	11.05
PropNPG	0.446	-0.69	5.81	3.59	8.97
PropEAF	0.144	-6.64	8.78	-13.31	12.82
PropNAF	0.983	18.89	13.44	15.68	13.02
PropNPF	0.300	-7.52	11.48	-9.13	8.20
OakRegen	0.865	-0.49	2.12	-0.25	1.66
ConRegen	0.332	0.00	0.15	-0.07	0.18
ShrubRegen	0.458	1.70	2.79	0.88	1.36

Table 8. MRPP tests for differences in treatment effects between groups within canopy vegetation and treatment type pair-subsets, and matched tests for differences in unthinned transects between the same subset groups. Tests between pair groups are based on within-pair differences (thinned minus unthinned) in herbaceous species percent cover data. Tests between unthinned transect groups are based on \log_{10} -transformed herbaceous species percent cover data. Significant differences ($p < 0.05$) and corresponding A-values are in bold. See Table 1 for definitions of treatment type and canopy vegetation group abbreviations.

Pairs subset	Within-subset group comparison	Members per group	Data source and type	MRPP	
				p	A
ARC	HPB : MM	5, 5	Within-pair difference	0.008	0.032
ARC	HPB : MM	5, 5	Unthinned transects cover	0.065	0.042
MIX	HPB : MM	3, 8	Within-pair difference	0.282	0.007
MIX	HPB : MM	3, 8	Unthinned transects cover	0.515	-0.007
CEA	HPB : MM	4, 5	Within-pair difference	0.034	0.026
CEA	HPB : MM	4, 5	Unthinned transects cover	0.493	-0.002
HPB	ARC : MIX : CEA	5, 3, 4	Within-pair difference	0.002	0.067
HPB	ARC : MIX : CEA	5, 3, 4	Unthinned transects cover	0.001	0.121
HPB	ARC : CEA	5, 4	Within-pair difference	0.031	0.040
HPB	ARC : CEA	5, 4	Unthinned transects cover	0.005	0.135
HPB	ARC : MIX	5, 3	Within-pair difference	0.007	0.055
HPB	ARC : MIX	5, 3	Unthinned transects cover	0.023	0.079
HPB	CEA : MIX	4, 3	Within-pair difference	0.030	0.072
HPB	CEA : MIX	4, 3	Unthinned transects cover	0.031	0.049
MM	ARC : MIX : CEA	5, 8, 5	Within-pair difference	0.417	0.001
MM	ARC : MIX : CEA	5, 8, 5	Unthinned transects cover	0.053	0.029

DISCUSSION

Fuel reduction thinning treatments in chaparral and mixed shrub/oak vegetation of southwestern OR appear to have significant impacts on abiotic site conditions, the cover and composition of herbaceous vegetation and the regeneration of canopy species. These effects were clearly apparent four to seven years post-treatment. Interpretation of results is, however, constrained by a lack of pre-thinning and early post-thinning data, limited sample sizes within vegetation and treatment type groups, a scarcity of unthinned reference sites, and evidence of differences between vegetation communities unrelated to thinning treatments. Nevertheless, some generalizations about treatment impacts can be made. The main focus of this discussion is on overall treatment effects, inclusive of both treatment types and all three vegetation communities as representative of the range of variation in the landscape and in management practices. While I use the term "treatment effects" to refer to differences between paired thinned and unthinned transects, the retrospective and observational nature of this study makes it impossible to determine cause and effect with certainty.

Thinning dramatically reduced canopy cover and stand density, and substantially increased coarse wood debris and burn pile scars, while litter cover decreased relative to that in unthinned areas. Although not directly measured, these abiotic differences indicate the occurrence of greater fluctuations in soil moisture and temperature and higher sunlight intensities at thinned compared to unthinned sites, and altered conditions for seed availability, dispersal and germination, and plant survival for both native and exotic species (Keeley and Zedler 1978; Keeley 1992b; Whisenant 1999; Davis et al. 2000).

These site condition differences between thinned and unthinned areas were associated with significant changes in herbaceous communities. Thinning increased herbaceous cover, but this increase was not accompanied by detectable changes in species diversity. Although both exotic and native species (as general categories) had expanded cover in thinned areas, there was no difference in their relative dominance between thinned and unthinned sites. Thus, at the most general level, thinning treatments in my study area cannot be said to favor either native or exotic species, and cannot be said to enhance species diversity within a four to seven year post-treatment time span.

More complex dynamics were, however, apparent when communities were examined at the level of functional groups. Amongst exotic species, composition shifted from forbs to grasses; exotic annual forbs decreased while exotic annual grasses increased strongly in response to thinning. Among native species, annuals were favored over perennials by thinning treatments. Native annual forbs showed the greatest expansion in thinned sites of

any functional group in this study, while native perennial forbs declined after thinning. The only species of native annual grass that occurred in sampled plots increased at thinned sites, but there was no change in cover of native perennial grasses. These compositional dynamics at the functional group level reveal differential treatment responses amongst native and exotic species, despite the apparent lack of change in relative proportions of native and exotic vegetation overall.

The most pronounced overall pattern of vegetation response to thinning was the loss of perennial species cover and expansion of annual cover. Cover of annual species nearly doubled following thinning, reaching an average cover of > 82% on thinned sites. This pattern did not involve a reversal of dominance; annual species had greater absolute and proportional cover relative to perennials at both thinned and unthinned sites. However, thinning treatments substantially widened the gap between them – the average proportional cover of annuals was 19% greater than perennials at unthinned sites, but was 55% greater at thinned sites.

This dramatic increase in annuals and decline in perennials fits an expected early-post-disturbance pattern of vegetation response, as does the marked increase in invasive annual grasses (Keeley et al. 1981; Hobbs and Huenneke 1992; Davis et al. 2000). Herbaceous vegetation at thinned sites seems to have persisted in an early post-disturbance community composition, dominated by native annual forbs (> 44% proportional cover) and exotic annual grasses (> 17% proportional cover), even four to seven years after treatment. Abundance of annual and exotic species increased at treated sites during the first two years after HPB and MM thinning in *C. cuneatus* chaparral of southwestern OR (Sikes 2005), which is consistent with my longer-term findings and suggests a stagnation of the herbaceous communities on thinned sites in my study area.

There are no published studies on the oak and chaparral communities of southwestern OR, or their response to disturbance, with which my results can be directly compared. Post-fire studies in California (CA) chaparral provide the best available context for reference. Because wildfire is the primary natural disturbance to which chaparral communities have adapted, in evaluating the impacts of fuel-reduction thinning I compared my findings to patterns of vegetation response and succession following wildfire. In contrast to my results, CA chaparral showed trends of herbaceous succession from annual to perennial species over the first five years after wildfire (Keeley et al. 1981). Proportional cover of perennial species in those systems tended to increase over time while annual species peaked during the first three years, such that by the fourth or fifth post-fire year, perennial species comprised most of the herbaceous cover. A similar successional trend does not appear to have occurred during the first four to seven years following thinning treatments in my study area. (However, because

no earlier post-treatment data are available for my study sites, I cannot rule out the possibility that a more gradual expansion of perennial grasses and forbs is occurring at thinned sites.) In some cases, extreme or repeated disturbances in chaparral ecosystems can lead to semi-permanent degraded states dominated by annuals or exotic grasses (Vogl 1982; Fried et al. 2004). The herbaceous composition at thinned sites in my study area four to seven years after treatment has features in common with this condition.

Keeley et al. (1981) also reported trends of rapid regeneration of shrub cover over the first three years post-fire, with native chaparral shrub cover approaching pre-fire levels after five years. I did not find comparable levels of shrub regeneration. Although there was significantly more shrub regeneration at thinned sites compared to unthinned, the difference was relatively small. Within the four to seven year post-thinning period encompassed by my study, shrub regeneration accounted for an average of < 3% cover at thinned sites, compared to ~ 50% average cover from shrub regeneration at most sites by the third post-fire year in CA chaparral (Keeley et al. 1981). Although high levels of shrub regeneration may be undesirable from a long-term fuel-reduction management perspective, the very low shrub regeneration found at my study sites may not be adequate to allow chaparral ecosystems to re-establish, potentially leading to ecosystem type-conversion at thinned sites.

Arctostaphylos viscida and *C. cuneatus* are obligate seeders, reproducing only from seed, with germination stimulated by fire- and heat-related cues (Keeley 1987; Fried et al. 2004); neither species re-sprouts following fire, nor were they observed to re-sprout after thinning. At thinned sites in my study area, shrub seedlings were concentrated at the periphery of burn scars and in areas of intense solar soil heating, which is consistent with observations for *C. cuneatus* reported by Sikes (2005). Just east of my study area, thinned *A. viscida* sites showed dense shrub regeneration from seed, with canopy cover of young shrubs in excess of 50% across extensive areas < 8 years post-treatment (author's unpublished data). These sites were HPB-treated in 1996, but the pile-burning phase closely resembled a moderate-intensity broadcast burn (P. Hosten, Medford BLM, pers. comm.), which apparently stimulated shrub regeneration. In the absence of fire, reproduction of obligate-seeding shrub species is greatly reduced or negligible (Keeley 1992a, 1992b), while high levels of fire-related seedling regeneration by *Arctostaphylos* and *Ceanothus* shrub species have been reported following wildfires in CA (Keeley and Zedler 1978, Odion and Davis 2000). A general lack of fire-related germination cues at thinned sites in my study area is probably responsible for the absence of substantial post-thinning shrub reestablishment after four to seven years.

Fire-related cues are also important for seed germination in numerous species of native forbs associated with shrub and chaparral communities (Keeley et al. 1985; Keeley and

Fotheringham 1998; Borchert 1989). A lack of fire cues probably influenced post-thinning herbaceous composition both directly, due to altered germination relative to post-fire conditions, and indirectly because the shrub canopy has not reestablished. Regenerating shrub canopies reduced herbaceous cover and altered herbaceous species composition within a few years following wildfires in chaparral (Keeley et al. 1981; Fried et al. 2004). Thus, the combination of a lack of fire-induced germination, potential loss of soil seed banks due to fire suppression (Borchert 1989), competition from aggressive non-native species (Hobbs and Huenneke 1992), and a near-absence of canopy regeneration, could explain the apparent persistence of an early-succession-like herbaceous community dominated by native annual forbs and exotic annual grasses at thinned sites in my study.

Of particular concern amongst these vegetation patterns is the widespread presence of exotic annual grasses at unthinned sites, and their marked increase in response to thinning treatments. The mean cover of exotic annual grasses more than doubled in response to treatments, reaching almost 20% at thinned sites. Eleven species of exotic annual grasses occurred in sampled transects, at least nine of which are considered to be invasive, noxious or species of concern (Pyke 1999; Sheley and Petroff 1999; Whitson et al. 2004). At many thinned sites, invasive annual grasses grew in dense continuous stands across large areas, especially *Avena fatua*, *Bromus diandrus*, *B. madritensis*, *B. tectorum* and *Cynosurus echinatus*. For example, *Bromus tectorum* had up to 55% cover on thinned transects, with an average cover of 8.6% across all thinned sites (compared to a maximum cover of 8.4% and mean of 1.2% at unthinned sites). Invasive annual grasses were a pre-existing problem within my study area, but thinning treatments appear to have facilitated their expansion.

These invasive species threaten native ecosystems in at least two interrelated ways: competition with and displacement of native vegetation, and changes in fuel profiles that shift the fire regime outside of the range of tolerance for native species (Whisenant 1990; Hobbs and Huenneke 1992; Keeley 2001; Brooks et al. 2004). Invasive grasses can dominate sites to the exclusion of native species with profound impacts on ecosystem processes, habitat and forage for wildlife (Pyke 1999; Sheley and Petroff 1999; Lambrinos 2000; Seabloom et al. 2003). Thinning treatment activities, crews and equipment may have directly introduced and assisted the spread of exotic species at thinned sites (Backer et al. 2004). The thinning-related expansion of cover and dominance of known invasive annual grasses reported here should be cause for concern and reaction by land managers engaged in fuel-reduction treatments in these and similar ecosystems, especially given the landscape-scale at which treatments are carried out. It is, however, important to note in this context that thinning

treatments were also associated with declines in the cover and dominance of exotic annual forbs, with potential benefits to native communities.

When assessing impacts of fuel-reduction treatments on plant communities, it is also important to consider those functional groups and species that did not change in response to thinning. These included native species that land managers have hoped would benefit from canopy reduction treatments. It has been suggested (e.g., USDI 1999) that fire-suppression has allowed for the growth of overly-dense canopies that have reduced herbaceous habitat through shading and caused declines in native species abundance and diversity. In this context, thinning treatments have been presented by the BLM as ecosystem restoration for certain locations and vegetation communities (e.g., USDI 1999, 2004). However, while thinning increased the cover of native annual species and total herbaceous vegetation, there was no evidence that thinning increased the proportion of native species within the community or improved native species richness or diversity overall. This differs from findings for post-wildfire community dynamics of herbaceous chaparral species, in which species richness and herbaceous cover were positively correlated (Keeley et al. 1981). If a management goal for thinning treatments in some areas is enhancement of native species diversity, my data suggest that this goal is generally not being met within a post-treatment span of four to seven years.

Similarly, land managers in the study area have assumed that, in the absence of fire, shrubs have encroached into previously open oak and perennial bunchgrass systems, causing the loss of native perennial grasses. Thinning has been advocated as a way to restore these presumed historic communities (e.g., USDI 1999). However, neither native perennial grasses nor native oak (*Quercus garryana* and *Q. kelloggii*) regeneration responded positively to thinning treatments within four to seven years. Regeneration of oaks and cover or dominance of native perennial grasses did not differ between thinned and unthinned sites – resprouting stumps, cut during thinning, accounted for much of the oak regeneration in thinned areas. In fact, my results indicate that oak regeneration and proportional cover of native perennial grasses may have declined overall in response to thinning. Fuel-reduction treatments, as applied in my study area, have clearly met one fuel management goal of reducing canopy fuels, and have also reduced or redistributed total fuels at treated sites. However, fuel-reduction thinning has not accomplished ecosystem restoration in terms of increasing cover or regeneration of native oak and perennial bunchgrass species within the time span of my study. Furthermore, treatments have apparently facilitated the expansion of invasive annual grass species that out-compete and exclude perennial bunchgrasses in other ecosystems (Whisenant 1990, Pyke 1999, Sheley and Petroff 1999).

Restoration goals for thinning treatments in my study area are primarily focused on sites with a substantial presence of oak trees, while at chaparral-dominated sites, fuel-reduction goals take priority and restoration goals are generally undefined or absent (P. Hosten, Medford BLM, pers. comm.). Although thinning treatments have reduced canopy fuels, the substantial increase in ground fuels at thinned sites, both as fine herbaceous cover and coarse wood debris, has the potential to increase both the frequency of ignitions and the severity of soil impacts from future fires at thinned sites. Such alterations to the fire regime and burn severity could have profound impacts on ecosystem processes, vegetation structure and composition, and habitat (Agee 1993; Keeley 2001; Brooks et al. 2004; Fried et al. 2004; Korb et al. 2004).

My study revealed differences in herbaceous community composition between various canopy vegetation groups, and highlighted the importance of accounting for canopy vegetation differences when examining treatment impacts in these ecosystems. Differences in treatment effects between HPB and MM thinning methods were detected only for abiotic site conditions when treatments were compared across all vegetation types. This apparent lack of difference in herbaceous responses to HPB vs. MM treatments may be largely attributable to high levels of variation from community composition differences among ARC, MIX and CEA when analyses spanned these vegetation groups. There *did* appear to be differences in herbaceous responses between treatment types when HPB and MM impacts were compared within ARC and within CEA vegetation groups. The absence of a detectable difference between treatment types within the MIX vegetation group may be due to the overall smaller magnitude of herbaceous treatment effect in MIX relative to ARC and CEA. The MIX vegetation group supported more oak trees, and thus had more canopy retention and canopy cover following thinning than ARC or CEA sites, which may explain the smaller magnitude of treatment impacts on MIX sites. These results suggest that HPB and MM treatments interact with different canopy vegetation types in distinct ways, although small sample sizes at the within-vegetation-group level of analysis make inferences and interpretations uncertain. Additional research with larger within-vegetation-type sample sizes will be necessary to confidently characterize differences in HPB and MM treatment impacts. Treatment prescriptions and thinning methods may need to be selected and adjusted for specific oak and chaparral community types to minimize negative impacts and improve restoration success.

CONCLUSION

Fuel-reduction thinning treatments in the oak and chaparral communities of my study area have had significant and persistent impacts on herbaceous vegetation composition and canopy regeneration, while appearing to fall short of restoration goals within the time-span of this study. Success in terms of fuel-reduction goals appears to be mixed, with reduced canopy fuels but increased ground fuels, especially at Slashbuster™ (MM) treated sites. Post-treatment expansion of invasive annual grasses, and the decrease (or lack of change) in native perennials and oak tree regeneration should be of particular concern for those interested in the restoration and conservation of native vegetation and habitats. Taken as a whole, my results suggest a degree of incompatibility between fuel reduction and restoration goals for thinning treatments as applied within my study area. A lack of clearly stated restoration goals and objectives, both in general and for specific vegetation communities, makes it difficult to evaluate the efficacy of thinning treatments as restoration tools. For chaparral communities in which treatment goals are entirely focused on fuel-reduction, the ecological impacts of thinning, at least some of which are undesirable, are potentially being ignored by management.

Results from this study emphasize the need for continued research on the oak and chaparral ecosystems of southwestern OR, and expanded monitoring of the impacts of fuel-reduction thinning treatments on these vegetation communities. Research focused on describing the historic vegetation structure and fire regimes for these ecosystems, and the impacts of fire suppression and wildfires on them would provide a basis for evaluating the ecological appropriateness of current thinning prescriptions (see Keeley et al. 1999; Keeley and Fotheringham 2001; Moritz et al. 2004). At the management level, exploration of alternative prescriptions, such as higher levels of canopy retention, reduced size and connectivity of thinning units, or the seeding of native perennials is recommended, as these might ameliorate negative thinning outcomes and enhance restoration successes. The introduction of prescribed fire in thinned areas as a follow-up treatment (USDI 2002; P. Hosten, Medford BLM, pers. comm.) offers both hope and concern for the recovery of native vegetation communities, and careful monitoring of vegetation impacts will be essential for assessing its effectiveness.

The Ashland Resource Area of the Medford District BLM is an ideal setting for continued research and experimentation on consequences of canopy thinning treatments, given the extensive and ongoing application of such treatments in this landscape. Further, because it is part of the Applegate River Adaptive Management Area, public lands

management in this area carries a responsibility for active monitoring of project outcomes and ongoing re-assessment of goals and methods (USDA-USDI 1994; USDI 1999). The application of landscape-scale thinning treatments and extensive use of the relatively new Slashbuster™ equipment warrant continued monitoring and research-based adaptive management. While I found apparent differences in thinning impacts between MM and HPB treatments, inferences were limited because of small sample sizes within canopy vegetation types. Treatment planning and execution in my study area did not include provisions for monitoring and research, resulting in a limited availability of sampling sites and a lack of pre-treatment data. In order for research to effectively inform future adaptive management, treatment prescriptions should incorporate collection of pre-thinning data and inclusion of control areas that are large enough to represent the associated unthinned plant communities. Finally, both fuel-reduction and ecological restoration goals for thinning treatments should be clearly defined for all vegetation communities and treatment areas to allow for appropriate evaluation of management outcomes.

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APPENDICES

Appendix 1. Species list sorted by functional groups for all transects sampled in this study. Within functional groups, species are sorted alphabetically. Scientific names and authorities follow Hickman (1993).

Species sorted by functional groups	Authority	Family
Native trees:		
<i>Arbutus menziesii</i>	Pursh	Ericaceae
<i>Pinus ponderosa</i>	Laws.	Pinaceae
<i>Pseudotsuga menziesii</i>	(Mirbel) Franco	Pinaceae
<i>Quercus garryana</i>	Hook.	Fagaceae
<i>Quercus kelloggii</i>	Newb.	Fagaceae
Native shrubs:		
<i>Arctostaphylos viscida</i>	C. Parry	Ericaceae
<i>Ceanothus cuneatus</i>	(Hook.) Nutt.	Rhamnaceae
<i>Ceanothus integerrimus</i>	Hook. & Arn.	Rhamnaceae
<i>Cercocarpus betuloides</i>	Torrey & A. Gray	Rosaceae
<i>Prunus subcordata</i>	Benth.	Rosaceae
Native woody perennials:		
<i>Galium porrigens</i>	Dempster	Rubiaceae
<i>Lonicera hispidula</i>	Douglas	Caprifoliaceae
<i>Lonicera interrupta</i>	Benth.	Caprifoliaceae
<i>Toxicodendron diversilobum</i>	(Torrey & A. Gray) E. Greene	Anacardiaceae
Native perennial grasses:		
<i>Achnatherum lemmonii</i>	(Vasey) Barkworth	Poaceae
<i>Bromus carinatus</i>	Hook. & Arn.	Poaceae
<i>Bromus laevipes</i>	Shear	Poaceae
<i>Elymus elymoides</i>	(Raf.) Swezey	Poaceae
<i>Elymus glaucus</i>	Buckley	Poaceae
<i>Festuca californica</i>	Vasey	Poaceae
<i>Festuca idahoensis</i>	Elmer	Poaceae
<i>Koeleria macrantha</i>	(Ledeb.) J.A. Schultes	Poaceae
<i>Melica californica</i>	Scribner	Poaceae
<i>Poa secunda</i>	J.S. Presl	Poaceae
Exotic perennial grass:		
<i>Poa bulbosa</i>	L.	Poaceae
Native annual grass:		
<i>Vulpia microstachys</i>	(Nutt.) Munro	Poaceae
Exotic annual grasses:		
<i>Aira caryophylla</i>	L.	Poaceae
<i>Avena fatua</i>	L.	Poaceae
<i>Bromus diandrus</i>	Roth	Poaceae
<i>Bromus hordeaceus</i>	L.	Poaceae
<i>Bromus japonicus</i>	Murr.	Poaceae
<i>Bromus madritensis</i>	L.	Poaceae

Appendix 1 (Continued)

Species sorted by functional groups	Authority	Family
Exotic annual grasses (Continued):		
<i>Bromus sterilis</i>	L.	Poaceae
<i>Bromus tectorum</i>	L.	Poaceae
<i>Cynosurus echinatus</i>	L.	Poaceae
<i>Taeniatherum caput-medusae</i>	(L.) Nevski	Poaceae
<i>Vulpia myuros</i>	(L.) C. Gmelin	Poaceae
Native perennial forbs:		
<i>Achillea millefolium</i>	L.	Asteraceae
<i>Agoseris grandiflora</i>	(Nutt.) E. Greene	Asteraceae
<i>Antennaria argentea</i>	Benth.	Asteraceae
<i>Astragalus accidens</i>	S. Watson	Fabaceae
<i>Calochortus tolmiei</i>	Hook. & Arn.	Liliaceae
<i>Cirsium cymosum</i>	(E. Greene) Jepson	Asteraceae
<i>Cynoglossum grande</i>	Lehm.	Boraginaceae
<i>Dichelostemma congestum</i>	(Sm.) Kunth	Liliaceae
<i>Dodecatheon hendersonii</i>	A. Gray	Primulaceae
<i>Erigeron inornatus</i>	A. Gray	Asteraceae
<i>Eriophyllum lanatum</i>	(Pursh) James Forbes	Asteraceae
<i>Fragaria vesca</i>	L.	Rosaceae
<i>Hieracium scouleri</i>	Hook.	Asteraceae
<i>Lomatium californicum</i>	(Torrey & A. Gray) Mathias & Constance	Apiaceae
<i>Lomatium dissectum</i>	(Torrey & A. Gray) Mathias & Constance	Apiaceae
<i>Lomatium nudicaule</i>	(Pursh) J. Coulter & Rose	Apiaceae
<i>Lomatium triternatum</i>	(Pursh) J. Coulter & Rose	Apiaceae
<i>Marah oreganus</i>	(Torrey & A. Gray) Howell	Cucurbitaceae
<i>Monardella sheltonii</i>	Torrey	Lamiaceae
<i>Osmorhiza chilensis</i>	Hook. & Arn.	Apiaceae
<i>Penstemon deustus</i>	Lindley	Scrophulariaceae
<i>Perideridia oregana</i>	(S. Watson) Mathias	Apiaceae
<i>Phacelia heterophylla</i>	Pursh	Hydrophyllaceae
<i>Potentilla glandulosa</i>	Lindley	Rosaceae
<i>Ranunculus occidentalis</i>	Nutt.	Ranunculaceae
<i>Sanicula crassicaulis</i>	DC.	Apiaceae
<i>Scutellaria antirrhinoides</i>	Benth.	Lamiaceae
<i>Scutellaria siphocampyloides</i>	Vatke	Lamiaceae
<i>Sidalcea malviflora</i>	(DC.) Benth.	Malvaceae
<i>Silene hookeri</i>	Nutt.	Caryophyllaceae
Exotic perennial forb:		
<i>Hypericum perforatum</i>	L.	Hypericaceae

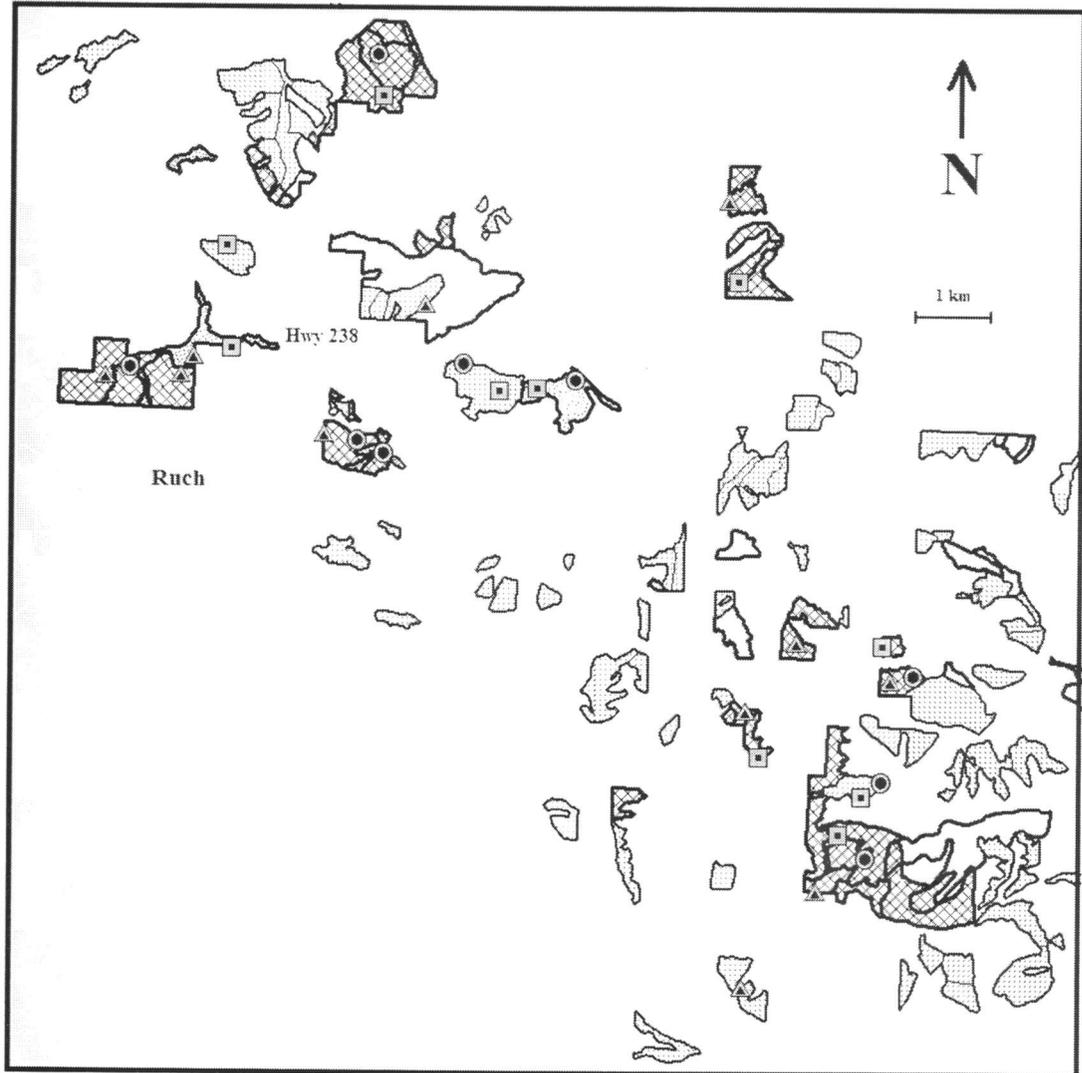
Appendix 1 (Continued)

Species sorted by functional groups	Authority	Family
Native annual forbs:		
<i>Agoseris heterophylla</i>	(Nutt.) E. Greene	Asteraceae
<i>Amsinckia menziesii</i>	(Lehm.) Nelson & J.F. Macbr.	Boraginaceae
<i>Athysanus pusillus</i>	(Hook.) E. Greene	Brassicaceae
<i>Castilleja attenuata</i>	(A. Gray) Chuang & Heckard	Scrophulariaceae
<i>Clarkia gracilis</i>	(Piper) Nelson & J.F. Macbr.	Onagraceae
<i>Clarkia purpurea</i>	(Curtis) Nelson & J.F. Macbr.	Onagraceae
<i>Clarkia rhomboidea</i>	Douglas	Onagraceae
<i>Claytonia parviflora</i>	Hook.	Portulacaceae
<i>Collinsia linearis</i>	A. Gray	Scrophulariaceae
<i>Collomia grandiflora</i>	Lindley	Polemoniaceae
<i>Cryptantha flaccida</i>	(Lehm.) E. Greene	Boraginaceae
<i>Cryptantha intermedia</i>	(A. Gray) E. Greene	Boraginaceae
<i>Cryptantha torreyana</i>	(A. Gray) E. Greene	Boraginaceae
<i>Daucus pusillus</i>	Michaux	Apiaceae
<i>Epilobium brachycarpum</i>	C. Presl	Onagraceae
<i>Epilobium minutum</i>	Lehm.	Onagraceae
<i>Eremocarpus setigerus</i>	(Hook.) Benth.	Euphorbiaceae
<i>Eriogonum vimineum</i>	Benth.	Polygonaceae
<i>Euphorbia spathulata</i>	Lam.	Euphorbiaceae
<i>Galium aparine</i>	L.	Rubiaceae
<i>Gilia capitata</i>	Sims	Polemoniaceae
<i>Githopsis specularioides</i>	Nutt.	Campanulaceae
<i>Hesperolinon micranthum</i>	(A. Gray) Small	Linaceae
<i>Linanthus bicolor</i>	(Nutt.) E. Greene	Polemoniaceae
<i>Linanthus bolanderi</i>	(A. Gray) E. Greene	Polemoniaceae
<i>Lotus humistratus</i>	E. Greene	Fabaceae
<i>Lotus micranthus</i>	Benth.	Fabaceae
<i>Madia exigua</i>	(Smith) A. Gray	Asteraceae
<i>Madia gracilis</i>	(Smith) Keck	Asteraceae
<i>Micropus californicus</i>	Fischer & C. Meyer	Asteraceae
<i>Minuartia douglasii</i>	(Torrey & A. Gray) Mattf.	Caryophyllaceae
<i>Nemophila parviflora</i>	Benth.	Hydrophyllaceae
<i>Pectocarya pusilla</i>	(A. DC.) A. Gray	Boraginaceae
<i>Phlox gracilis</i>	(Hook.) E. Greene	Polemoniaceae
<i>Plagiobothrys nothofulvus</i>	(A. Gray) A. Gray	Boraginaceae
<i>Plectritis congesta</i>	(Lindley) A. DC.	Valerianaceae
<i>Rafinesquia californica</i>	Nutt.	Asteraceae
<i>Ranunculus hebecarpus</i>	Hook. & Arn.	Ranunculaceae
<i>Rigiopappus leptocladus</i>	A. Gray	Asteraceae
<i>Stellaria nitens</i>	Nutt.	Caryophyllaceae
<i>Stephanomeria elata</i>	Nutt.	Asteraceae

Appendix 1 (Continued)

Species sorted by functional groups	Authority	Family
Native annual forbs		
(Continued):		
<i>Tonella tenella</i>	(Benth.) A.A. Heller	Scrophulariaceae
<i>Trifolium albopurpureum</i>	Torrey & A. Gray	Fabaceae
<i>Trifolium ciliolatum</i>	Benth.	Fabaceae
<i>Uropappus lindleyi</i>	(DC.) Nutt.	Asteraceae
<i>Yabea microcarpa</i>	(Hook. & Arn.) Koso-Polj.	Apiaceae
Exotic annual forbs:		
<i>Anthriscus caucalis</i>	M. Bieb.	Apiaceae
<i>Arenaria serpyllifolia</i>	L.	Caryophyllaceae
<i>Cerastium glomeratum</i>	Thuill.	Caryophyllaceae
<i>Crepis pulchra</i>	L.	Asteraceae
<i>Erodium cicutarium</i>	(L.) L'Her.	Geraniaceae
<i>Galium parisiense</i>	L.	Rubiaceae
<i>Lactuca serriola</i>	L.	Asteraceae
<i>Torilis arvensis</i>	(Hudson) Link	Apiaceae
<i>Trifolium dubium</i>	Sibth.	Fabaceae
<i>Valerianella locusta</i>	(L.) Betcke	Valerianaceae
<i>Verbascum thapsus</i>	L.	Scrophulariaceae
<i>Veronica arvensis</i>	L.	Scrophulariaceae

Appendix 2. Map of study area showing locations of the 30 transect pairs used in analyses. Areas of fuel-reduction thinning treatments completed from 1998 through 2001 are outlined in black. MM treatments are cross-hatch filled, and HPB treatments are dot filled. Private lands are shaded grey, with major roadways shown in white. Locations of each pair of matched transects are indicated by a single symbol coded by canopy vegetation group: ARC = square; MIX = triangle; CEA = circle. See Table 1 for definitions of treatment type and canopy vegetation group abbreviations.



Appendix 3. Paired transect names, location coordinates, bearings, treatment type and dates, and related information. Coordinates for transects are given in UTM zone 10, NAD 27, and specify the location of a permanent metal post marking the origin of each transect. Bearings in degrees are given with an 18° East declination adjustment. For treatments, HPB = hand-cut and pile-burn, and MM = mechanical mastication.

Transect	Treatment	Thin month	Thin year	UTM East	UTM North	Bearing	Notes
AP5Ct	MM	May	2001	495691	4677065	25	
AP5Cu	none			495738	4677101	205	
AP6Xt	MM	May	2001	495558	4676967	260	1
AP6Xu	none			495548	4676992	260	2
B10bCt	MM	Feb.	2001	505207	4671515	60	
B10bCu	none			505192	4671534	60/50	3
B10bMt	MM	Feb.	2001	504805	4671634	65	
B10bMu	none			504850	4671624	65/75	4
B10bXt	MM	Feb.	2001	504358	4671002	105	
B10bXu	none			504385	4670965	110	
B32-3bXt	MM	May	1998	504066	4674011	40	
B32-3bXu	none			504063	4674070	230	5
B33-1aMt	MM	Jun.	1998	505142	4674006	0	
B33-1aMu	none			505112	4673998	0	
B33-2Ct	MM	Jun.	1998	505448	4673657	280	
B33-2Cu	none			505457	4673704	260	
B33-2Xt	MM	Jun.	1998	505222	4673597	200	
B33-2Xu	none			505194	4673609	210	
B5-2Xt	MM	May	1998	503373	4673204	330	
B5-2Xt	none			503374	4673180	140/120	6
B5-5Mt	MM	May	1998	503751	4672586	275	
B5-5Mu	none			503707	4672618	95	
BB4Ct	HPB	May	1998	505191	4672308	240	
BB4Cu	none			505220	4672290	65	
BB4Mt	HPB	May	1998	505121	4672193	255	
BB4Mu	none			505270	4672233	65	
FC13Mt	HPB	Mar.	2000	496943	4677436	100	
FC13Mu	none			497005	4677390	280	
FC14Ct	MM	May	2000	498566	4676261	200	
FC14Cu	none			498598	4676253	210	
FC14Xt	MM	May	2000	498152	4676468	175	
FC14Xu	none			498120	4676454	170/180	7
FC1bMt	MM	unknown	1999	498772	4680529	150	
FC1bMu	none			498750	4680505	150	
FC22Xt	MM	Jun.	2001	503085	4679453	180	
FC22Xu	none			503050	4679436	200	
FC25Ct	HPB	Feb.	2000	501281	4677092	195	
FC25Cu	none			501254	4677100	200	

Appendix 3 (Continued)

Transect	Treatment	Thin month	Thin year	UTM east	UTM north	Bearing	Notes
FC25Mt	HPB	Feb.	2000	500690	4676948	15	
FC25Mu	none			500670	4676930	190	8
FC27Ct	HPB	Feb.	1999	498737	4676102	70	
FC27Cu	none			498640	4676030	245	
FC29Mt	MM	Jun.	2001	503086	4678393	40	
FC29Mu	none			503160	4678415	215	
FC33Xt	HPB	Oct.	1999	499163	4677901	260	
FC33Xu	none			499154	4677928	260	
FC38XtHP	HPB	Jun.	2000	496433	4677221	90	9
FC38Xu	none			496425	4677189	275	
FC38XtSB	MM	Jun.	2000	496453	4677175	270	
FC38Xu	none			496425	4677189	275	
FC44Ct	HPB	Apr.	1999	499980	4677291	255	
FC44Cu	none			500003	4677301	80	
FC44Mt	HPB	Apr.	1999	500451	4676997	255	
FC44Mu	none			500450	4677033	255	
FC8Ct	MM	Jun.	2000	498750	4680917	170	
FC8Cu	none			498795	4680897	350	
FCFB12Mt	HPB	Jul.	2000	496855	4678625	260	
FCFB12Mu	none			496848	4678656	260	
SW42Xt	HPB	Nov.	1999	503574	4669875	185	
SW42Xu	none			503608	4669877	5	

Notes on transects:

1. Split transect, 30 m at listed UTMs, 20 m at E 495548, N 4676954.
2. Split transect, 30 m at listed UTMs, 20 m at E 495541, N 4677002.
3. Bearing of 60° to 30 m, then 50° to 50 m.
4. Bearing of 65° to 30 m, then 75° to 50 m.
5. Split transect, 30 m at listed UTMs, 20 m at E 504062, N 4674055.
6. Bearing of 140° to 25 m, then 120° to 50 m.
7. Bearing of 170° to 30 m, then 180° to 50 m.
8. Split transect, 35 m at listed UTMs, 15 m section approximately 10 m to W.
9. Split transect in 3 sections, 20m at listed UTMs, 15m at E 496437, N 4677200, and a third section 15 m long originating midway between these two.

Appendix 4. Exploratory analysis and results describing differences in treatment effects between HPB and MM thinning methods within the ARC and CEA vegetation group subsets.

Analysis methods:

To characterize differences in treatment effects between HPB and MM treatment types within vegetation group subsets, I used NMS ordinations of herbaceous within-pair difference data and examined variable correlations with ordination axes. This approach allowed for comparisons of treatment type differences while controlling for canopy vegetation group. NMS solutions were obtained for ARC and CEA pair-subsets independently using the "Slow and Thorough" PC-ORD autopilot setting and Euclidean distance. Two-dimensional solutions were accepted for these ordinations because they had significantly lower stress than solutions based on randomized data (Monte Carlo test). In these ordinations, pairs are represented as single points arranged according to their similarity in within-pair differences in herbaceous cover, and ordination axes summarize gradients of difference in vegetation response to thinning treatments. Ordinations of within-pair differences were rotated to maximize the separation of treatment types along a single axis, and correlations of difference-values for vegetation and abiotic variables with that axis were calculated (Kendall's tau). These correlations were used to identify apparent differences in treatment effects for key variables between HPB and SB treated pairs within ARC and CEA canopy vegetation groups. The statistical power of analyses at the pair-subset level was limited by small sample sizes. Because of this, univariate tests for the significance of differences in vegetation or abiotic response variables between treatment types were not pursued at the pair-subset level.

Results:

While apparent differences in herbaceous responses to HPB vs. MM treatments within canopy vegetation groups deserve attention, caution must be used in interpreting results because sample sizes are so small within subsets (Table 9). Nevertheless, some tentative results are presented here comparing HPB and MM treatments based on ordinations of within-pair difference data for ARC and CEA vegetation group subsets (Table 9). NMS ordination of pair-differences in herbaceous composition for the ARC subset of 10 pairs yielded a two-dimensional solution that represented 84.3% of the variance in the original data (final stress = 9.9). MM and HPB groups clearly separated along Axis 1 (low and high scores respectively), which summarized 70.1% of the variation in species cover differences between pairs (ordination not shown). Substrate changes in response to treatments were as expected, with increases in burn scar cover strongly correlated with HPB sites, while increases in wood debris were correlated with MM sites, although only weakly (Table 9).

Greater increases in herbaceous cover were correlated with HPB sites than with MM sites in the ordination of ARC pairs. Both overall and native Shannon-Weiner diversity were positively correlated with Axis 1. Overall diversity apparently increased and native diversity remained essentially unchanged on average at HPB sites, but both overall and native diversity seem to have decreased at MM sites (based on within-group mean differences, Table 9). Changes in cover of native perennial grasses, native annual forbs, and native species overall were also positively correlated with Axis 1, increasing overall following treatments, but more so at HPB sites. Changes in the proportion of herbaceous vegetation comprised of native perennial forbs and exotic annual forbs were negatively correlated with Axis 1; in this case, mean differences suggest that proportions of both functional groups decreased overall, but less so at MM sites (Table 9). Change in cover of native perennial forbs was also negatively correlated with Axis 1, but comparison of mean within-group differences did not allow for a clear interpretation of this correlation. Variables for degree of change in annual or perennial species overall were not correlated with the axis separating HPB and MM pairs in this subset ordination.

For the CEA subset of 9 transect pairs, NMS ordination gave a two-dimensional solution explaining 85.5% of the variation in the original within-pair difference data (final stress 7.5), and of this 59.8% was summarized by Axis 1 which separated MM pairs (low scores) from HPB pairs (high scores) (ordination not shown). Differences in treatment impacts for burn scar and wood debris cover were correlated with Axis 1, and thus with HPB and MM treatments, following the same pattern as for the ARC subset (Table 9). In contrast to the ARC subset, litter cover was positively correlated with Axis 1 in the CEA subset ordination, with means of within-group differences indicating increases at HPB sites and decreased litter cover at MM sites. Greater increases in species richness were correlated with HPB sites for the CEA subset, but native richness was only weakly correlated at best, and measures of diversity were not correlated with Axis 1 (Table 9). Differences in cover and in proportion for exotic annual grasses were positively correlated to Axis 1. Cover of exotic annual grasses increased overall, but apparently more so at HPB than at MM sites; however, as a proportion of total herbaceous cover, exotic annual grasses apparently decreased somewhat at MM sites while increasing at HPB sites (Table 9). Changes in proportional cover of perennial species were negatively correlated with Axis 1, suggesting greater increases in perennials as a proportion of total herbaceous cover at MM sites relative to HPB. No other changes in species traits or functional groups were correlated with Axis 1 in this ordination of CEA pairs.

Table 9. Correlations (Kendall's tau) of difference variables with Axis 1 for both the ARC and CEA subset ordinations of within-pair differences in herbaceous composition (thinned minus unthinned); and mean differences for HPB and MM treatment types within subsets. For both ordinations, Axis 1 tended to separate HPB and MM pairs such that when tau is positive, higher values for difference variables are associated with HPB sites; when tau is negative, higher values for difference variables are associated with MM sites. In this context, higher values for a difference variable can mean either an increase, or a smaller decrease, relative to values for pairs at the opposite end of the axis gradient. Mean within-pair differences for HPB and MM groups, based on raw data, are provided for both the ARC and CEA subsets to assist in interpretation of tau values. Variables with $|\text{tau}| \geq 0.30$ were reported as "correlated" and are in bold; variables with $|\text{tau}| \geq 0.50$ were considered "strongly correlated" and are underlined and in bold. See Table 4 for definitions of the variable abbreviations.

Variable	ARC subset, 5 MM, 5 HPB pairs			CEA subset, 5 MM, 4 HPB pairs		
	tau	MM mean difference	HPB mean difference	tau	MM mean difference	HPB mean difference
% Cover	-0.289	-66.18	-80.30	-0.111	-76.16	-81.43
WoodDebris	-0.244	19.73	3.99	-0.222	9.81	1.69
BurnScar	<u>0.653</u>	0.00	18.84	<u>0.719</u>	0.00	22.65
Litter	-0.067	-20.79	-14.43	<u>0.500</u>	-2.97	13.15
Soil	0.022	1.16	0.88	0.056	-20.33	-19.56
Herbsum	0.422	18.94	59.21	0.167	42.03	45.41
Richness	0.276	-0.40	4.60	0.310	0.00	2.00
NatRichness	0.023	-1.00	1.00	0.254	0.80	2.75
Diversity	<u>0.600</u>	-0.26	0.24	0.111	-0.12	-0.25
NatDiversity	0.422	-0.42	0.01	0.222	-0.03	-0.18
Native	0.333	19.83	27.89	-0.111	41.59	29.56
Exotic	0.156	-0.89	31.32	0.278	0.44	15.85
Annual	0.200	26.98	62.45	0.111	52.59	53.20
Perennial	0.156	-8.04	-3.24	-0.167	-10.56	-7.79
EAG	0.200	1.29	33.59	0.389	3.62	20.53
NPG	0.422	0.94	5.23	0.029	-0.54	-0.45
EAF	-0.111	-2.24	-2.27	0.167	-3.06	-3.41
NAF	0.467	21.57	28.92	-0.056	45.25	36.91
NPF	-0.333	-4.28	-3.59	0.000	-4.43	-0.26
PropNative	-0.156	-0.31	-6.89	-0.111	8.28	1.83
PropExotic	-0.200	0.31	6.89	0.111	-8.28	-1.83
PropAnnual	-0.111	-26.10	-27.44	0.000	-23.88	-18.22
PropPerenn	-0.200	26.10	27.44	-0.389	23.88	18.22
PropEAG	0.156	2.32	20.24	0.389	-2.60	9.07
PropNPG	0.289	-5.31	3.74	0.085	-0.94	-0.81
PropEAF	-0.378	-2.07	-13.35	0.111	-5.47	-8.27
PropNAF	0.156	20.61	20.16	0.000	27.96	19.12
PropNPF	-0.422	-11.90	-14.62	0.111	-9.21	-6.27
OakRegen	-0.289	-1.72	-0.88	-0.087	1.08	-0.10
ConRegen	-0.028	-0.12	-0.16	0.000	0.00	0.00
ShrubRegen	0.225	4.88	1.92	0.145	0.16	0.30

Appendix 5. CD containing Excel spreadsheets of study data. A single Excel file with four worksheets containing species percent cover by transect, a key to species abbreviation codes, vegetation-related variables by transect, and abiotic variables by transect. See analysis methods for sources of calculated vegetation and abiotic variables. See Table 1 for definitions of treatment type and canopy vegetation group abbreviations.