RESEARCH ARTICLE

PRESCRIBED FIRE AND POST-FIRE SEEDING IN BRUSH MASTICATED OAK-CHAPARRAL: CONSEQUENCES FOR NATIVE AND NON-NATIVE PLANTS

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ABSTRACT

In fire-suppressed oak-chaparral communities, land managers have treated thousands of hectares by mechanical mastication to reduce hazardous fuels in areas of wildland-urban interface. The chipped debris, which decomposes slowly, can be burned to minimize wildfire hazard. The question is whether controlled burning of masticated debris results in loss of native plant species richness and abundance, allowing for gains in non-native species. We examined the response of vegetation to the seasonality of prescribed fire and to post-fire seeding in mechanically masticated oak-chaparral communities in the Applegate Valley of southwestern Oregon, USA. At the landscape level, treatments did not differ. At the site level, response of native and non-native species varied by site and treatment. Following prescribed fire, native species decreased in cover and increased in species richness; non-native species increased in cover and in species richness. Seven species that were not observed on pre-treatment plots appeared after burn treatments. Non-native annual grasses and forbs increased following both spring and fall burns. Among native species, annuals declined in cover while perennials increased slightly. Both annual and perennial natives increased in species richness following burn treatments. Community patterns at the site scale changed following all treatments. Seeded bunchgrasses, Lemmon's needlegrass (*Achnatherum lemmonii* [Vasey] Barkworth), California brome (*Bromus carinatus* Hook. and Arn.), blue wildrye (*Elymus glaucus* Buckley), and Roemer's fescue (*Festuca idahoensis* Elmer ssp. *roemeri* [Pavlick] S. Aiken), successfully established following fall prescribed fires, but not following spring prescribed fires or in unburned controls. Post-fire seeding and subsequent increased bunchgrass cover correlated with decreased non-native species. Prescribed low severity fire followed by post-fire seeding during the wet, cool season is a viable tool for introducing native bunchgrasses while controlling non-native species in mechanically masticated oak-chaparral in southwestern Oregon.

Keywords: bunchgrasses, fire, fuels reduction, mechanical mastication, native plants, oak-chaparral, post-fire seeding, prescribed brush mastication, weeds

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INTRODUCTION

Oak woodlands and shrublands in southwestern Oregon, USA, are characteristic of the chaparral vegetation formed by the northernmost extent of the Mediterranean climate in North America (Detling 1961, Keeley 2002). These plant communities support a mixed severity fire regime with variable fire frequencies due to synergistic effects of climate and topography (Agee 1991, Taylor and Skinner 2003, Odion *et al.* 2004, Hosten *et al.* 2006). Decades of fire suppression along with residential development in fire-prone ecosystems have increased wildfire occurrences involving life and property, raising fire to a prominent place on the political agenda (Dombeck *et al.* 2004). Furthermore, checkerboard ownership of public land administered by the Bureau of Land Management interspersed with private land has created thousand of hectares of wildland-urban interface.

Mechanical mastication has been used as an efficient and economical fuel management alternative to handcutting, piling, and burning by shifting fuels from the canopy to the ground (Busse *et al.* 2005, Bradley *et al.* 2006, Glitzenstein *et al.* 2006, Kane *et al.* 2009, Reiner *et al.* 2009). In masticated areas, fires burn as more manageable surface fires than as uncontrollable crown fires in untreated shrublands. However, mechanical mastication of brush increases woody fuel loading on the ground, leading to hotter surface fires but with reduced risk of crown fire (Busse *et al.* 2005, Kane *et al.* 2009). Fires implemented within one year of mastication burn as slow, higher severity surface fires that detrimentally affect soils, native seed banks, and remaining tree and shrub longevity (Busse *et al.* 2005, Keeley 2006). Reduction in slash biomass after several years may allow for follow-up burn treatments of less fire intensity and severity.

Because the ecological consequences of prescribed fire on vegetation overlain with the debris layer created by mechanical mastication are unknown in oak-chaparral communities, potential adverse effects are a concern to resource managers. The combination of ground-disturbing activities, reduced canopy cover, and increased fuels at ground level may alter the physical and biological properties of soils, with consequences for plant communities (Neary *et al.* 1999, Boerner *et al.* 2009, Schwilk *et al.* 2009).

The season of prescribed fire may influence the relative responses of native and non-native plants. Burning in spring may prevent seed set of invasive annual grasses and weaken their ability to displace native perennials (LeFer and Parker 2005). On the other hand, spring prescribed fire may delay germination of native species, leaving bare ground that invasive species may colonize (Keeley 2001, 2006; LeFer and Parker 2005). In contrast, fall burning coincides with the historical fire regime of oak-chaparral communities of southwest Oregon and may promote native seedling establishment (LaLande 1995, Pullen 1996, Keeley *et al.* 2005). However, during a fall fire, lethal temperatures produced in mechanically masticated shrublands with high fuel loads and dry conditions may reduce the native seed bank (Busse *et al.* 2005).

Post-fire seeding with native perennial grasses may help restore a native perennial understory following mechanical mastication and prescribed fire. Ground disturbance caused by mechanical treatments creates opportunities for non-native plants to establish in a layer of slash that may inhibit germination of native species (Keeley 2002, Perchemlides *et al.* 2008, Sikes and Muir 2009). Prescribed fire can promote non-native plant establishment by providing an environment rich in nutrients, reducing competition, and increasing light and space to grow. The addition of native seeds can enhance native plant cover and exclude non-native plant cover (Suding and Gross 2006). By removing thatch and slash, seeds sown after fire directly contact bare soil, enhancing germination success and establishment (Maret and Wilson 2005).

In this study we examine the consequences of prescribed fire on native and non-native species in mechanically masticated oak-chaparral communities. The objectives were to determine 1) the relative responses of native and non-native species to spring and fall prescribed fires in mechanically treated oak-chaparral vegetation; 2) the establishment of native perennial grasses by post-fire seeding; and 3) the interaction of seeded grasses with natives and non-natives.

METHODS

Study Sites

We selected two masticated sites 7.2 km apart—China Gulch (42.25 °N, 122.05 °W, elevation 710 m, slope 55%, aspect S to SE) and Hukill Hollow (42.19 °N, 122.98 °W, elevation 720 m, slope 35%, aspect SE to SW)—in the wildland-urban interface on the Medford District Bureau of Land Management in Jackson County, near Ruch, Oregon (Figure 1). Woodland chaparral communities dominated both sites with buckbrush (*Ceanothus cuneatus* [Hook.] Nutt.), whiteleaf manzanita (*Arctostaphylos viscida* Parry), Pacific madrone (*Arbutus menziesii* Pursh), Oregon white oak (*Quercus garryana* Dougl. ex Hook.), and ponderosa pine (*Pinus ponderosa* C. Lawson) (Pfaff 2007, Coulter 2008). The climate is seasonally dry with cool, wet winters and hot, dry summers (mean annual precipitation, 646 mm; mean temperature, 4 °C in January, 21 °C in July) (WRCC 2008). Soils at both sites were a Vannoy-Voorhies complex composed of 16% to 18% clay.

Sites had been mechanically masticated with a large rotating blade (BM-Slashbuster®, Motesana, Washington, USA) attached to a track-mounted excavator; China Gulch in 2001 and Hukill Hollow in 2002. Slash had been left on the ground and approximately half of the slash biomass had decomposed within four years (P.E. Hosten, Bureau of Land Management, personal communication).

Experimental Design

We applied a before-after-control-impact (BACI) design of pre- and post-burn vegetation with spring and fall burns (Smith 2002). Three treatment blocks (spring burn, fall burn, and control), each 40 m × 20 m in size, were established 30 m downslope of roads, totaling 90 1 m^2 plots at each site (Coulter 2008). We established the first plot location by tossing a quadrat frame to the northwest corner of a treatment block, then spaced subsequent plots 1 m to 2 m east of the first plot, moving downslope. When placement of a plot fell on a shrub, we moved the quadrat frame to the other side of the shrub, biasing the surveyed vegetation toward forbs and grasses. Within each treatment block, we paired plots with similar dominant plant species for seeded and unseeded controls.

To estimate vegetation cover, we used the 12 cover classes in the FIREMON protocol (0% to 1%, >1% to 5%, >5% to 15%, >15% to 25%, >25% to 35%, >35% to 45%, >45% to 55%, >55% to 65%, >65% to 75%, >75% to 85%, >85% to 95%, and >95% to 100%) assessing the area defined by the drip line of the plant canopy and assigning the midpoint of the class as the cover area (Lutes *et al.* 2006). In cases in which plants overlapped, the sum cover area for all species in a plot totaled over 100%. Environmental variables assessed as percent cover included rock, thatch, slash, burned ground, bare ground, and charcoal.

In 2005, three and four years after mechanical mastication, we surveyed pre-fire vegetation by identifying species in June and assessing cover in July and August, a particularly wet year with a late blooming period. After

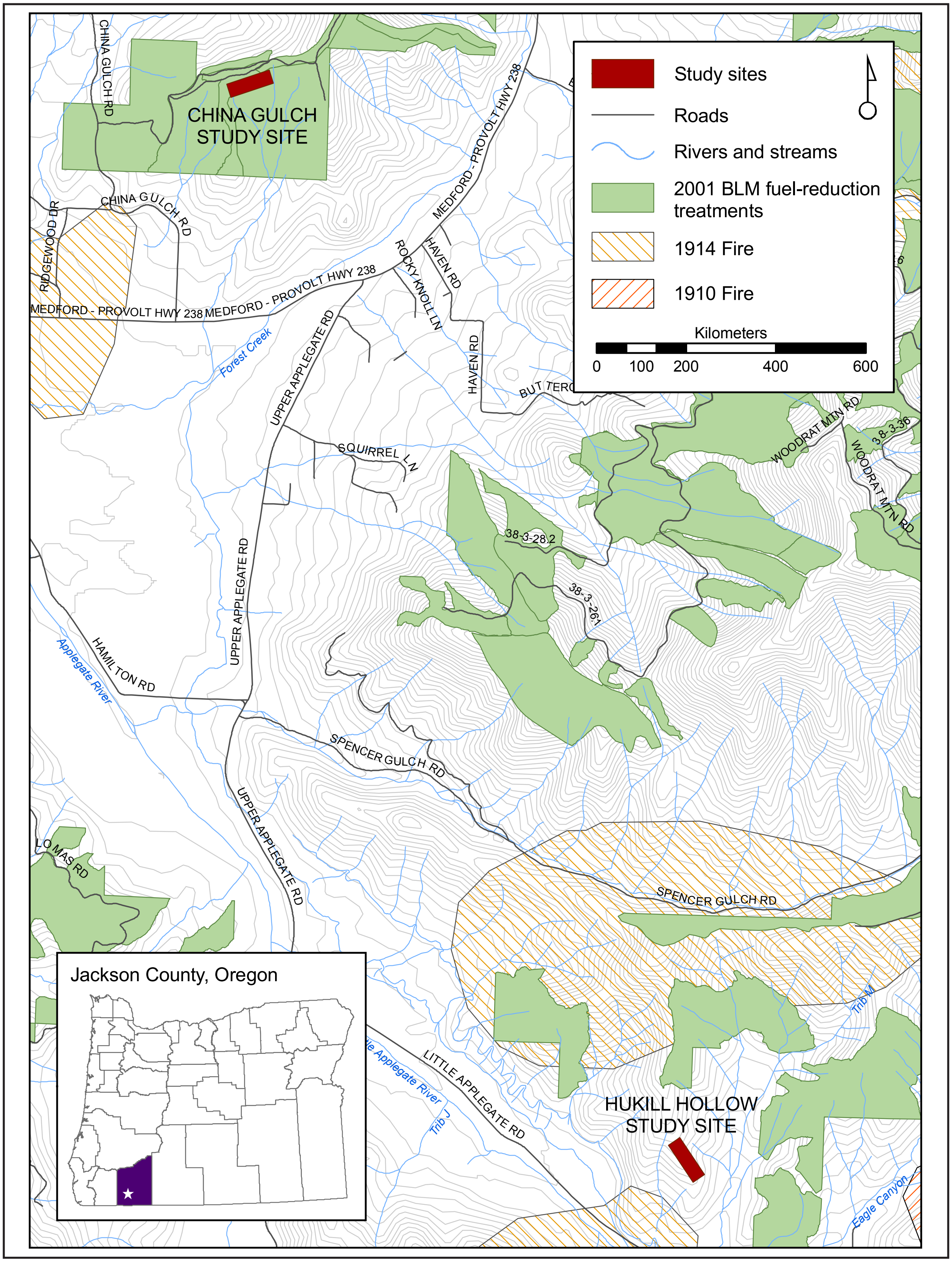

Figure 1. Site map locations for China Gulch and Hukill Hollow, Applegate Valley, Jackson County, Oregon, USA.

fall fires in October 2005 and spring fires in April 2006, we completed a preliminary post-fire vegetation survey in May and June and again in August 2006, but much of the burned ground was bare, and the vegetation had not yet recovered. Thus, we completed our assessment of post-fire vegetation in June 2007. Plant codes, nomenclature, and authorship follow the USDA Plants Database (2008).

Prescribed Burns

Fire crews from the Medford District Bureau of Land Management implemented prescribed fires at both sites on 6 October 2005 and on 21 April 2006, with burns covering approximately 0.4 ha. Crews used Brown's transects to measure fuel loads before and after prescribed fires (NWCG 2008). Fire crews ignited prescribed fires using a strip head firing pattern with 3.0 m to 4.6 m between strips of fire and, in areas of higher fuel loads, used a backing fire. Tiles oriented facing upslope next to plot markers included OMEGALABEL® Model TL-10-105 temperature labels and OMEGAPELLET® temperature-indicating pellets PLT Series that burn at 650 °C and 750 °C (OMEGA Engineering, Stamford, Connecticut, USA). Crews documented fire behavior and evaluated surface temperatures during fires with tiles buried 5 cm below the soil surface, temperature labels 3.8 cm below the soil surface, and pellets placed on the soil surface.

Bunchgrass Seeding

Forty-eight hours after the prescribed fires, we hand-sowed one randomly selected plot of each pair in burned and control plots with four native bunchgrass species: Lemmon's needlegrass (*Achnatherum lemmonii* [Vasey] Barkworth), California brome (*Bromus carinatus* Hook. and Arn.), blue wildrye (*Elymus glaucus* Buckley), and Roemer's fescue (*Festuca idahoensis* Elmer ssp. *roemeri* [Pavlick] S. Aiken). The Medford District of the Bureau of Land Management supplied locally grown seed. The Oregon State University Seed Laboratory (Corvallis, Oregon, USA) conducted germination rate tests (http://www.seedlab.oscs.oregonstate.edu).

Standard seed applications for bunchgrasses range from 3.1 kg to 6.8 kg of seed per 0.4 ha (Archibald *et al.* 2000, NRCS 2008). We applied seed at a rate of 4.5 kg per 0.4 ha with 1.12 g of seed per 1 m^2 plot, 0.28 g per species. We evaluated field germination in 2006, eight months post-fire, with follow-up surveys in May 2007 and 2008.

Data Analysis

To evaluate post-treatment responses in the BACI design, we analyzed data at three scales: landscape scale comparing treatments at two sites, site scale comparing treatments within a site, and plot scale comparing relationships of cover and species richness of native and non-native species. We analyzed the responses of native and non-native species, life forms (perennial or annual), the plant community, the most common species, and seeded grasses.

For the landscape scale at two replicate sites, we compared changes in plant cover and species richness with the Kruskal-Wallis one-way analysis of variance using MINITAB version 15 (Minitab Inc., State College, Pennsylvania, USA) with a significance threshold of $P = 0.05$. For each test, we evaluated before and after cover and species richness (2005 and 2007) for spring and fall burns and the unburned control. We examined the results of grass seeding following prescribed fires using 2007 data.

At the site scale, applying the BACI design, we evaluated patterns of change (before and after) in the plant community at each site using Nonmetric Multidimensional Scaling (NMS) for ordination and blocked multi-response permutation procedures (MRBP) with PC-ORD 5.0 (McCune and Mefford 1999, McCune and Grace 2002). For NMS, the number of real runs totaled 50, with a stability criterion of 0.0005, 400 iterations, starting coordinates random, and the Sørensen (Bray-Curtis) distance measure. The joint plot function displayed the environmental variable most strongly correlated with plant community composition. The cut-off r^2 value was set at 0.1. We selected default scaling (% to Max) to display ordination points based on similarity in proportion to the longest axis. The secondary matrix contained six environmental variables (rock, thatch, slash, burned ground, bare ground, and charcoal).

To test the hypothesis of no difference between two groups for plant communities before and after spring and fall burns at both sites, we used MRBP (McCune and Grace 2002, Cai 2006). The distance measure was Euclidean and the grouping variable was time (before and after). The chance-corrected within-group agreement (A) was used to evaluate treatment effect. Values of A approaching 1.0 indicated within-group homogeneity and values approaching zero indicated differences between pre- and post-treatment groups when $P < 0.05$.

We used linear regression with Minitab v. 15 to determine the relationship between cover of native and non-native species on each plot with and without seeded grasses in fall burn plots only.

RESULTS

Spring and Fall Prescribed Fires

Spring fires resulted in a low severity, patchy burn. Fall fires were moderate-to-high severity in all plots with temperatures of 40 °C to 82 °C below the soil surface and 490 °C to 710 °C at the soil surface, and a flame residence time of 4 min around the base of Oregon white oaks. Although Hukill Hollow exhibited higher pre-burn fuel loads than at China Gulch, resulting in higher severity burns, the fall fire yielded a moderate to severe burn at both sites and fuel loads were reduced to similar levels (Table 1).

Native and Non-Native Species Response to Spring and Fall Prescribed Fires

Responses of the 43 native and 20 non-native species varied by site and treatment (Appendix). At the landscape scale, cover and species richness of annual non-native species did not differ among treatments (Table 2). Native species cover declined in the control and following both burn treatments, while native species richness increased in all conditions. Cover and species richness of non-native species increased following burn treatments and in the control (before and after). Among native species, annuals declined in cover while

Table 1. Fuel loads on masticated blocks before and after prescribed burns at China Gulch (CG) and Hukill Hollow (HH) in southwestern Oregon, USA.

		Fuel load (T ha^{-1})								
		Duff + litter		1 to 100 hr		1000 hr		Total		
Burn	Site	Pre	Post	Pre	Post	Pre	Post	Pre	Post	% change
Fall	CG	11.5	4.7	22.6	1.2	0	0	34.1	5.9	-82.7
Fall	HH	10.8	0.9	26.4	4.8	12.4	0.4	49.6	6.2	-87.5
Spring	CG	15.1	4.2	13.0	11.4	0	0.4	28.1	16	-43.1
Spring	HH	31.8	0.7	24.5	9.9	9.4	4.4	65.7	15	-77.2
Mean (SD)		17.3 (9.9)	2.6 (2.1)	21.65 (96)	6.8 (4.7)	5.5 (6.4)	1.3 (2.1)	44.38 (16.86)	10.8 (5.5)	-72.6 (20.1)

Table 2. Plant cover and species richness of native and non-native plants and of natives by life form (annual or perennial) before and after spring and fall burns in southwestern Oregon, USA. Percent cover (median) and species counts (mean) were based on thirty 1 m² plots at two replicate sites.

	Cover				**Species richness**			
	Native		**Non-native**		**Native**		**Non-native**	
Treatment	Δ cover	% change	Δ cover	% change	Δ species	% change	Δ species	% change
Control	-6.2	10.9	28.8	101.9	2	14.3	3.5	24.0
Spring burn	-20.9	-29.3	36.8	151.7	7	36.8	5	26.5
Fall burn	-2.1	1.5	-0.1	-0.4	5	26.9	3	7.1
	Native annual		**Native perennial**		**Native annual**		**Native perennial**	
Treatment	Δ cover	% change	Δ cover	% change	Δ species	% change	Δ species	% change
Control	-8.2	17.2	2.2	2.1	3	36.1	-1	-7.1
Spring burn	-2.1	-3.9	0.4	43.3	5.5	61.1	2	13.0
Fall burn	-31.6	-53.9	10.2	82.3	2.5	35.7	2.5	23.6

perennials increased slightly (Table 2). Both annuals and perennials increased in species richness following burn treatments.

Individual species differed in response to burns. Peregrine thistle (*Cirsium cymosum* [Greene] J.T. Howell), a native species, vigorously responded to the fall burn at one site, while varileaf phacelia (*Phacelia heterophylla* Pursh) was abundant at the other site. Several common species increased in cover following spring burns: common fiddleneck (*Amsinckia menziesii* [Lehm.] A. Nelson and J.F. Macbr. var. *intermedia* [Fisch. and C.A. Mey.] Ganders), weakstem cryptantha (*Cryptantha flaccida* [Douglas ex Lehm.] Greene), American wild carrot (*Daucus pusillus* Michx.), and tarweed (*Madia* sp.). Desert deervetch (*Lotus micranthus* Benth.) decreased after spring burns. Four species dominated the decline in native annuals after fall fires: winecup clarkia (*Clarkia purpurea* [W. Curtis] A. Nelson and J.F. Macbr. ssp. *quadrivulnera* [Douglas ex Lindl.] F.H. Lewis and M.E. Lewis), American wild carrot, tarweed, and desert deervetch (Table 2). Tarweed declined from an average of 30 % cover in pre-treatment plots to 13 % post-fire. Both common fiddleneck and bluehead gilia (*Gilia capitata* Sims.) increased following fall burns. Native annual grasses varied in response to spring fires. Small fescue (*Vulpia microstachys* [Nutt.] Munro) increased following the fall burn at one site while no significant response was observed at the other site. Native perennial shrubs increased after burns due to the prolific resprouting of poison oak (*Toxicodendron diversilobum* [Torr. and A. Gray] Greene).

Seven species that were not observed on pre-treatment plots appeared after burns. The native species Pacific hound's tongue (*Cynoglossum grande* Douglas ex Lehm.), stickywilly (*Galium aparine* L.), northern sanicle (*Sanicula graveolens* Poepp. ex DC.), rod wirelettuce (*Stephanomeria virgata* Benth.), and Klamath plum (*Prunus subcordata* Benth.) appeared, as did two non-natives, common mullein (*Verbascum thapsus* L.) and prickly lettuce (*Lactuca serriola* L.)

Non-native species increased following spring and fall burns with redstem stork's bill (*Erodium cicutarium* [L.] L'Hér. ex Aiton) and knotted hedge-parsley (*Torilis nodosa* [L.] Gaertn.) dominating the increases. Only one non-native perennial, common St. Johnswort

(*Hypericum perforatum* L.), occurred on the plots. The cover of non-native annual grasses did not change following spring prescribed fire, while the cover of five invasive annual grasses, silver hairgrass (*Aira caryophyllea* L.), cheatgrass (*Bromus tectorum* L.), soft brome (*B. hordeaceus* L.), red brome (*B. rubens* L.), and rat-tail fescue (*Vulpia myuros* [L.] C.C. Gmel.) increased following fall prescribed fire. Non-native perennial grasses increased following the spring burn due to an increase in abundance of bulbous bluegrass (*Poa bulbosa* L.)

Overall, the cover of non-native species had little effect on the cover of native species (Figure 2). Following the spring burn, there was a small increase in native plant cover as non-native cover increased. Increases in non-native species cover correlated with small declines in native species cover in fall burn plots.

Establishment of Seeded Perennial Grasses Following Spring and Fall Prescribed Fires

Native perennial grasses became established when sown following fall burns but not following spring burns (Table 3). Seeds of Lemmon's needlegrass, California brome, blue wildrye, and Idaho fescue (*Festuca idahoensis* Elmer) germinated in all plots following fall burns but not after spring burns or on control plots. Although all seeded species remained for two years following the fall burn, blue wildrye showed the greatest increase in cover and Lemmon's needlegrass the least. Seed germination rates were much lower for Lemmon's needlegrass.

The overall effect of seeded grasses was to reduce the cover of non-native species (Figure 3*a*). At Hukill Hollow, with the higher-severity fall burn, total native species cover correlated with a decline in non-native species in seeded plots (Figure 3*b*).

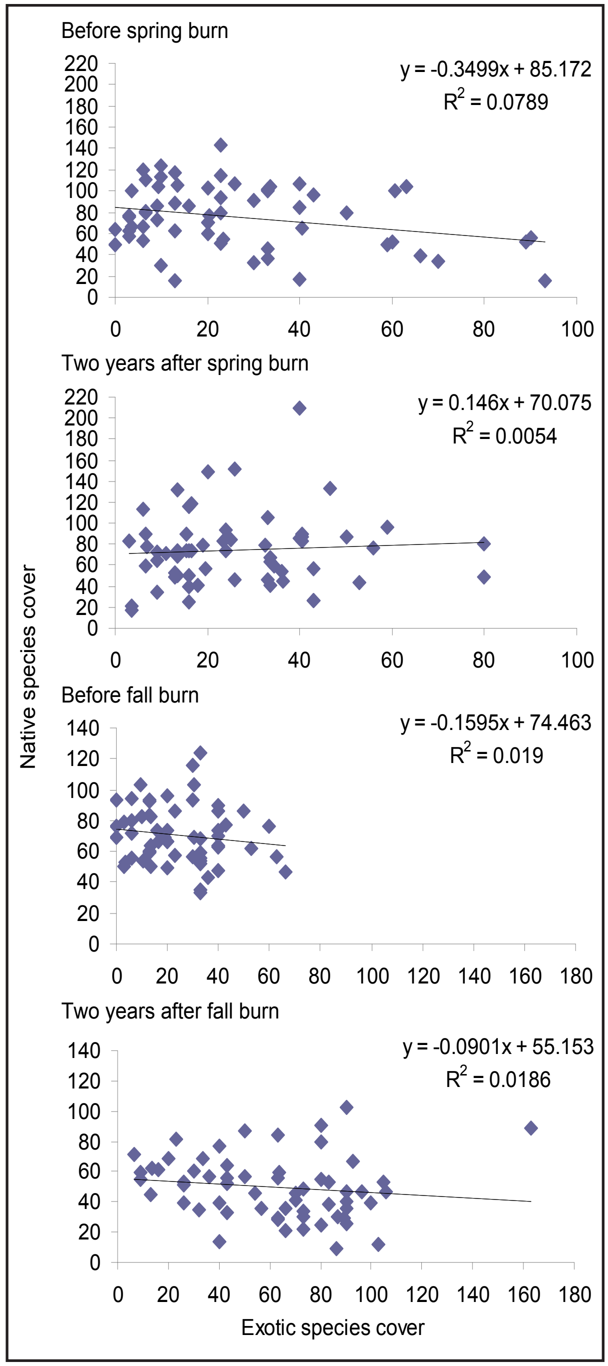

Figure 2. Correlations of native species cover with non-native species before and after spring and fall burns (for each treatment, n = 60 1 m^2 plots) and on plots seeded and unseeded with native bunch grasses (n = 15) at Hukill Hollow in southwestern Oregon, USA.

Table 3. Bunchgrass germination test rate and cover (mean) on seeded (n = 30) and unseeded (n = 30) 1 m^2 plots before and after fall burns of masticated chaparral in southwestern Oregon, USA.

		Species cover				
			1 year postburn		2 years postburn	
Species	**Germination test (%)**	**Preburn**	**Unseeded**	**Seeded**	**Unseeded**	**Seeded**
Lemmon's needlegrass	9	0	0	0.1	0	0.3
California brome	97	0.8	0.2	3.5	0.2	2.7
Blue wildrye	91	0	0	0.7	0	5.9
Roemer's fescue	60	0	0	0.9	0	0.8
All bunchgrasses		0.8	0.2	5.0	0.2	9.7

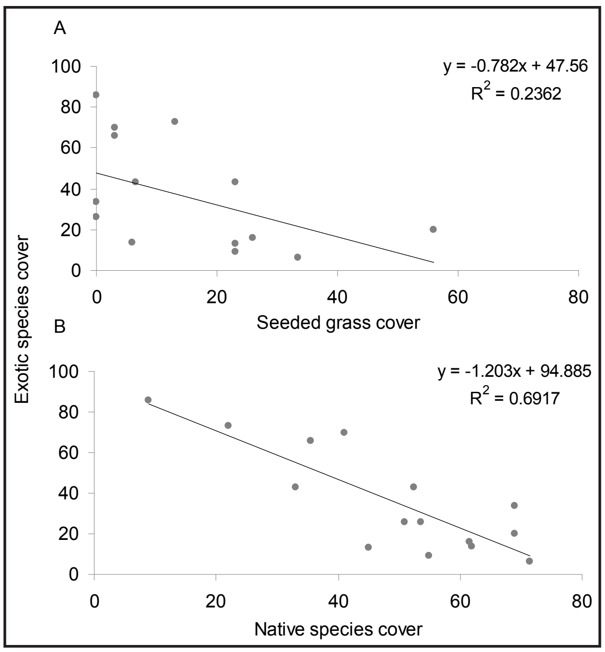

Figure 3. Correlations between seeded grasses, native species and non-native species following fall burns and grass seeding at Hukill Hollow in southwestern Oregon, USA. Correlation of seeded grass cover with decline in non-native species (A). Correlation of total native species including seeded grasses on decline in non-native species (B).

Plant Community Response to Prescribed Fires

Pre- and post-fire plant communities differed at both sites (MRBP, P < 0.001) for spring and fall burns and for the unburned control with the largest pre- and post-treatment differences in the fall burn plots (Table 4). These differences were driven by changes in a few species. Following the spring burn at China Gulch, increases in common fiddleneck and decreases in winecup clarkia dominated changes in the plant community. At Hukill Hollow, desert deervetch decreased following the spring burn. Following fall burns, decreases in tarweed at both sites resulted in significant changes in plant community composition.

Changes in plant community composition in control plots at China Gulch were driven by increases in common fiddleneck, knotted hedge-parsley, and rat-tail fescue. Similar to spring burn plots, plant community changes in Hukill Hollow control plots were due to a decrease in desert deervetch.

The NMS ordinations of plant species and environmental variables between pre- and post-fire surveys displayed distinct grouping patterns between pre- and post-treatment plots at China Gulch, but not at Hukill Hollow, which had more outlier species (Figure 4).

DISCUSSION

Comparison of Spring and Fall Prescribed Fires

Patchy, light severity spring burns differed dramatically from the moderate-to-high severity fall burns at both sites. In spring burn treatment blocks, fires did not consume all fuels or burn all plots, leaving perimeter plants only

Table 4. Within group homogeneity (A) of plant communities in burn treatments and controls, compared before and after, by blocked multi-response permutation procedures (MRBP) on 30 paired 1 m^2 plots per treatment at China Gulch (CG) and Hukill Hollow (HH) in southern Oregon, USA.

	Spring burn		Fall burn		Control	
Site	**A**	***P***	**A**	***P***	**A**	***P***
CG	0.13	1E-07	0.31	2E-08	0.25	2E-07
HH	0.05	0.004	0.10	9E-06	0.07	0.0003

scorched. Spring soil moisture reduced temperature and heat duration (Busse *et al.* 2005). By contrast, the moderate-to-high severity fall burns consumed or charred fuels and burned all plots.

Response of Native Species to Prescribed Fires

As a group, native species decreased in response to fall prescribed fire. Among native species, annuals declined in cover while perennials increased slightly. Both annuals and perennials increased in species richness following burn treatments.

Tarweed, which dominated pre-treatment plots, was still reduced in cover two years post-fire. Our finding of declines in native species differs from results in which fall prescribed fire increased native species diversity (LeFer and Parker 2005). High severity fires can produce lethal temperatures for native seed banks (Busse *et al.* 2005). Keeley (1977) observed seed mortality when southern California chaparral communities were exposed to temperatures greater than 120 °C. Temperatures below the soil surface in our study reached a maximum of 82 °C during fall prescribed fires, so it is not likely that the weak native species response to fall burns resulted from damage to the seed bank. One native shrub, poison oak, resprouted vigorously following both spring and fall fires.

It is unclear why native forb species regenerated poorly following fall prescribed burns. The role of a seed bank in preserving species diversity in fire-prone ecosystems is well known (Parker and Kelly 1989, LeFer and Parker 2005). Chaparral shrubland communities in southern California, excluded from fire for more than 100 years, recovered following fire, as did chaparral shrublands experiencing fire within the historic fire-return interval of 50 to 60 years (Keeley *et al.* 2005). Repeated management activities (e.g., mechanical mastication, ripping, and scarification) at both study sites may have resulted in a native seed bank more degraded than would be expected in an intact oak-chaparral community (Keeley 2002, Maret and Wilson 2005).

Response of Non-Native Species to Prescribed Fires

At the landscape scale, non-native species did not change in response to spring prescribed fires, while they increased following fall fires. Certain non-native species, particularly redstem stork's bill and knotted hedge-parsley, increased following spring and fall burns. Bare ground remained at both sites eight months after both burn treatments. The high severity fall burn may have postponed native plant response, leaving bare ground open for invasion by non-native annual grasses. A greater abundance of non-native species occurred in the second post-fire year compared to the first post-fire year. The invasive annual grasses red brome, cheatgrass, and rat-tail fescue increased following fall burns. All three invasive annual grasses were observed along roads near study sites and may have established from off-site seed sources (FEIS 2008).

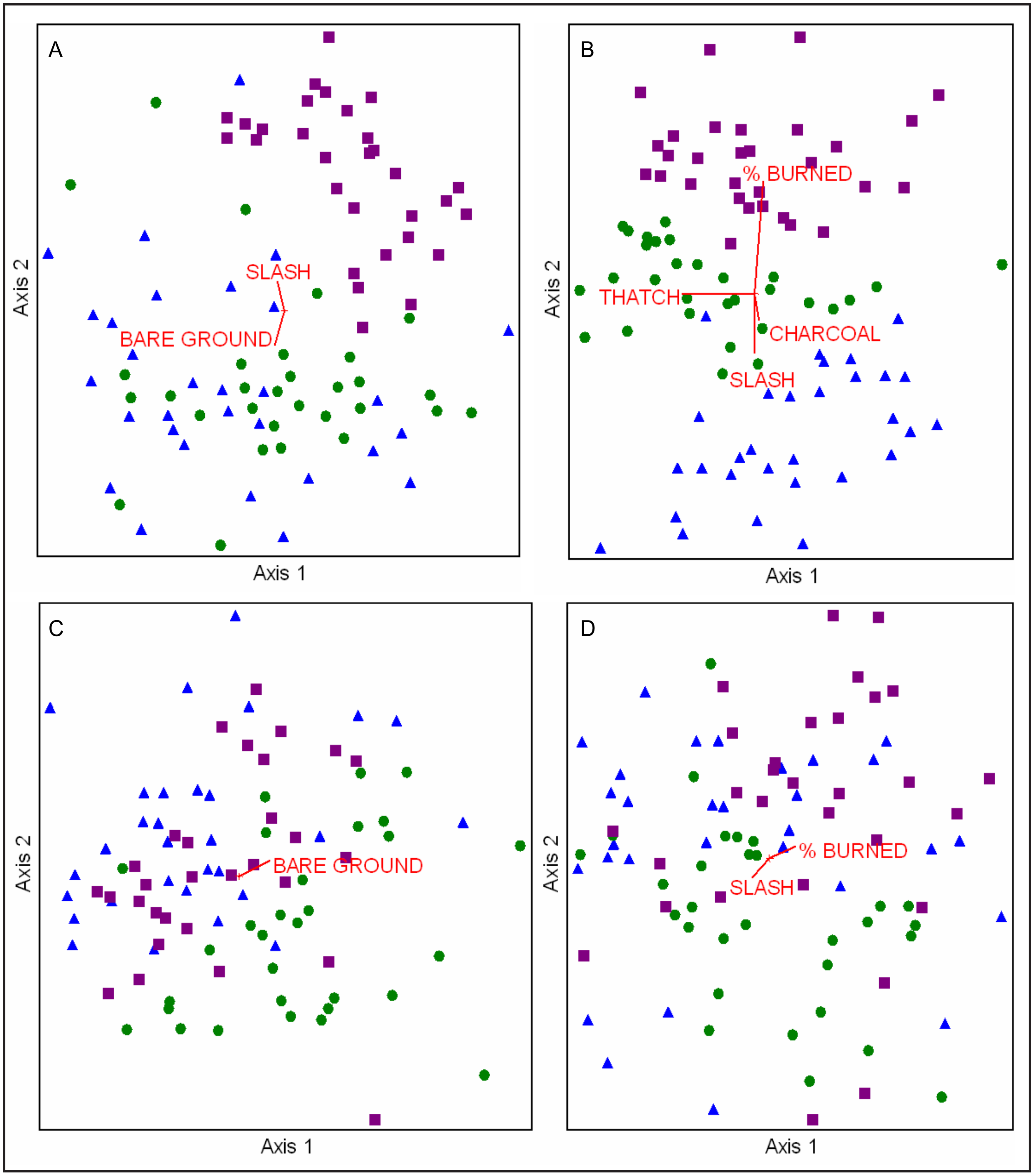

Figure 4. NMS ordinations of plant species and environmental variables comparing pre- and post-treatment (second post-fire year) plant communities: China Gulch pre-treatment (A), China Gulch post-treatment (B), Hukill Hollow pre-treatment (C), and Hukill Hollow post-treatment (D) in southwestern Oregon. Treatments include control plots (green ●), spring burn plots (blue ▲), and fall burn plots (purple ■).

Factors known to influence invasive species abundance in chaparral include woody canopy closure, the non-native seed bank, and fire severity (Keeley *et al.* 2005). Results from this study indicate that these three factors played a role in the abundance of invasive species at both study sites following fall burns. A shrub canopy cover, now absent at both sites, will exclude shade-intolerant invasive annuals over time (Keeley *et al.* 2005). In addition, the study sites were located within 30 m of roads that are known conduits for non-native plant invasion (Alston and Richardson 2006, Merriam *et al.* 2006). And lastly, deep-burning fires due to high levels of fuel loading in fall burns may have increased the competitive ability of invasive plant species by delaying the establishment of native seedlings (LeFer and Parker 2005). The loss of a major native shrub component from our sites is the primary reason for concern. Native species have limited ability to displace non-native species once an invasive plant community is established (Kulmatiski 2006). Managing fuel loads in oak-chaparral communities by mechanically masticating woody species may reduce wildland fire hazards while increasing non-native plant invasion, forcing land managers to weigh the ecological trade-offs.

Post-Fire Seeding Following Spring and Fall Prescribed Fires

Seeded bunchgrasses established well following fall burns but did not establish following spring burns at either site. In this seasonally dry climate, there may not have been sufficient precipitation before summer drought to allow seeds to become established. Because the cold, wet period followed the fall burn time, seeds would have had a longer time for stratification and germination. Success of seeded grasses following fall burns may be due to the closeness of the burn to the colder rainy period and not to the particular burn time. Similar germination success might have followed spring burns if seeding had been delayed until October. The lack of germination on control plots indicates that bare ground or thatch removal is necessary for establishment of seeded grass.

Individual species of bunchgrass varied in establishment success. Blue wildrye, the most abundant germinant after fall burns, does not require stratification or scarification to germinate, only the onset of fall precipitation (FEIS 2008). California brome germinates once cool, rainy weather begins, and it germinates slowly, which may place it at a disadvantage compared to faster growing non-native species such as cheatgrass. Roemer's fescue germinates once winter precipitation has sufficiently infiltrated soils. Unlike other bunchgrasses, Roemer's fescue requires an after-ripening period of six months, a delay that may account for the increase in germinants from first to third post-fire year. Lemmon's needlegrass requires 60 to 80 days of cold, moist chilling, a treatment that would occur naturally in fall and winter. In the field, Lemmon's needlegrass has high seed dormancy and delayed germination, which reduces its competitive ability following disturbance (NRCS 2008).

Unseeded plots were dominated by non-native annual grasses. The experimental design for this study using 1 m^2 plots created a matrix effect, with islands of seeded bunchgrass amid a spread of invasive annual grasses. In a larger scale study with a smaller proportion of edge effects, broadcast application of native seed after prescribed burns might minimize non-native plant invasion.

Although prescribed fire may benefit native plant communities, it also has negative impacts (Keeley and Bond 2001, Korb *et al.* 2003). Our results suggest that when post-fire seeding does not follow mechanical mastication and prescribed fire, the cover of non-native species increases. Cover by non-native species was lower in plots subjected to mechanical mastication alone. Perchemlides *et al.* (2008) found fewer non-native species in untreated sites compared to mechanically treated oak-chaparral sites in southwestern Oregon. Prescribed fire is a useful management tool, particularly when it stimulates a strong response from native perennials, allowing the native community to maintain dominance over the constant encroachment of invasive annual species (Maret and Wilson 2005).

Plant Community Response to Prescribed Fires and Post-Fire Seeding

Community analyses on pre-treatment and second-post-fire year data indicated a significant treatment effect for spring and fall burns at both sites. Post-fire plant species abundance differed from pre-fire abundance. Communities differed between pre- and post-fire years for spring and fall burns from China Gulch. Ordinations of pre- and post-fire year cover from Hukill Hollow yielded a weak grouping of plant communities for spring and fall burns, suggesting minor changes in plant community composition. Burned plots grouped along vectors with higher percentage of plot burned and higher cover of rock, charcoal, and bare ground. Unburned or scorched plots grouped along vectors with higher cover slash and thatch. Distinct grouping patterns found in the China Gulch ordinations may be due to large changes in plant abundance of several species, compared to only one or two species at Hukill Hollow. Seven species that were not observed on pre-treatment plots appeared after burns.

Conclusions

1. At the landscape level, prescribed burning of masticated vegetation did not decrease native plant species richness or abundance nor did it result in large increases of weedy non-native species. Although there was some shift in the community and changes in some species, the overall effect was not harmful to native vegetation.
2. At the site level, native annuals declined in cover while perennials increased slightly. Both annual and perennial natives increased in species richness following burn treatments. Spring burns facilitated increases in several native forbs and shrubs. While burning in spring did not suppress non-native annual grasses, it also did not increase their abundance.
3. Spring burns were patchy and light in contrast to moderate-to-high severity fall burns.
4. Either spring or fall is an appropriate season for burning mechanically masticated slash that has decomposed for at least three years following mechanical treatments.
5. Fall is the more appropriate season for seeding native grasses. Seeding of native bunchgrasses immediately following fall burns promoted their establishment. Seeding following spring burns, by contrast, did not provide adequate conditions for establishment. Native bunchgrasses did not become established in unburned plots.
6. Post-fire seeding and subsequent increased bunchgrass cover correlated with decreased non-native species. When post-fire seeding did not follow mechanical mastication and prescribed fire, non-native annual grasses invaded disturbed ground.

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LITERATURE CITED

Agee, J.K. 1991. Fire history along an elevational gradient in the Siskiyou Mountains, Oregon. Northwest Science 65(4): 188-199.

Alston, K.P., and D.M. Richardson. 2006. The roles of habitat features, disturbance, and distance from putative source populations in structuring alien plant invasions at the urban/wildland interface on the Cape Peninsula, South Africa. Biological Conservation 132: 183-198. doi: 10.1016/j.biocon.2006.03.023

Archibald, C., S. Feigner, and J. Visser. 2000. Seed and seedling production of blue wild-rye. Native Plants Journal (Spring): 32-34.

Boerner R.E., J.J. Huang, and S.C. Hart. 2009. Impacts of fire and fire surrogate treatments on forest soil properties: a meta-analytical approach. Ecological Applications 19: 338-358 doi: 10.1890/07-1767.1

Bradley, T., J. Gibson, and W. Bunn. 2006. Fuels management and non-native plant species: an evaluation for fire and fire surrogate treatments in a chaparral plant community. <http://jfsp.nifc.gov/projects/01B-3-3-27/01B-3-3-27_final_report.pdf>. Accessed 10 March 2010.

Busse, M.D., K.R. Hubbert, G.O. Fiddler, C.J. Shestak, and R.F. Powers. 2005. Lethal soil temperatures during burning of masticated forest residues. International Journal of Wildland Fire 14: 267-276. doi: 10.1071/WF04062

Cai, L. 2006. Multi-response permutation procedure as an alternative to the analysis of variance: an SPSS implementation. Behavior Research Methods 38: 51-59.

Coulter, C.T. 2008. Vegetation response to seasonality of prescribed fire and post-fire seeding following mechanical fuel-reduction treatments in oak-chaparral. Thesis, Southern Oregon University, Ashland, USA. <http://soda.sou.edu>. Accessed 12 March 2010.

Detling, L.E. 1961. The chaparral formation of southwestern Oregon, with considerations of its post-glacial history. Ecology 42: 348-357. doi: 10.2307/1932085

Dombeck, M.P., J.E. Williams, and C.A. Wood. 2004. Wildfire policy and public lands: integrating scientific understanding with social concerns across landscapes. Conservation Biology 18: 883-889. doi: 10.1111/j.1523-1739.2004.00491.x

Fire Effects Information System [FEIS]. 2008. Fire Effects Information System: plant species life form. <http://www.fs.fed.us/database/feis/plants/index.html>. Accessed 11 March 2010.

Glitzenstein, J.S., D.R. Streng, G.L. Achtemeier, L.P. Naeher, and D.D. Wade. 2006. Fuels and fire behavior in chipped and unchipped blocks: implications for land management near the wildland-urban interface. Forest Ecology and Management 15: 18-29. doi: 10.1016/j.foreco.2006.06.002

Hosten, P.E., O.E. Hickman, and F. Lang. 2006. Oak woodland and savanna restoration. Pages 87-123 in: D. Apostol and M. Sinclair, editors. Restoring the Pacific northwest: the art and science of ecological restoration in Cascadia. Island Press, Washington, D.C., USA.

Kane J.M., J.M. Varner, and E.E. Knapp. 2009. Novel fuelbed characteristics associated with mechanical mastication treatments in northern California and southwestern Oregon. International Journal of Wildland Fire 18: 686-697. doi: 10.1071/WF08072

Keeley, J.E. 1977. Seed production, seed populations in soil, and seedling production after fire for two congeneric pairs of sprouting and nonsprouting chaparral shrubs. Ecology 58: 820-829. doi: 10.2307/1936217

Keeley, J.E. 2001. Fire and invasive species in Mediterranean-climate ecosystems of California. Pages 81-94 in: K.E.M. Galley and T.P. Wilson, editors. Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species. Fire Conference 2000: the First National Congress on Fire Ecology, Prevention, and Management. Miscellaneous publication 11, Tall Timbers Research Station, Tallahassee, Florida, USA.

Keeley, J.E. 2002. Fire management of California shrubland landscapes. Environmental Management 29: 395-408. doi: 10.1007/s00267-001-0034-Y

Keeley, J.E. 2006. Fire management impacts on invasive plants in the western United States. Conservation Biology 20: 375-384. doi: 10.1111/j.1523-1739.2006.00339.x

Keeley, J.E., M. Baer-Keeley, and C.J. Fotheringham. 2005. Alien plant dynamics following fire in Mediterranean-climate California shrublands. Ecological Applications 15: 2109-2125. doi: 10.1890/04-1222

Keeley, J.E., and W.J. Bond. 2001. On incorporating fire into our thinking about natural ecosystems: a response to Saha and Howe. American Naturalist 158: 664-670. doi: 10.1086/323594

Korb, J.E., N.C. Johnson, and W.W. Covington. 2003. Arbuscular mycorrhizal propagule densities respond rapidly to ponderosa pine restoration treatments. Journal of Applied Ecology 48: 101-110. doi: 10.1046/j.1365-2664.2003.00781.x

Kulmatiski, A. 2006. Non-native plants establish persistent communities. Plant Ecology 187: 261-275. doi: 10.1007/s11258-006-9140-5

LeFer, D., and V.T. Parker. 2005. The effect of seasonality of burn on seed germination in chaparral: the role of soil moisture. Madroño 52: 166-174. doi: 10.3120/0024-9637(2005)52[166:TEOSOB]2.0.CO;2

Lutes, D.C., R.E. Keane, J.F. Caratti, C.H. Key, N.C. Benson, S. Sutherland, and L.H. Gangi. 2006. FIREMON: Fire effects monitoring and inventory system. General Technical Report RMRS-GTR-164-CD, USDA Forest Service Rocky Mountain Research Station, Fort Collins, Colorado, USA. <http://frames.nbii.gov/projects/firemon/FIREMON_SamplingMethods.pdf> Accessed 24 February 2010.

Maret, M.P. and M.V. Wilson. 2005. Fire and litter effects on seedling establishment in western Oregon upland prairies. Restoration Ecology 13: 562-568. doi: 10.1111/j.1526-100X.2005.00071.x

McCune, B., and J.B. Grace. 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach, Oregon, USA.

McCune, B., and M.J. Mefford. 1999. PC-ORD, multivariate analysis of ecological data, version 4. MjM Software Design, Gleneden Beach, Oregon, USA.

Merriam, K.E., J.E. Keeley, and J.L. Beyers. 2006. Fuel breaks affect nonnative species abundance in California plant communities. Ecological Applications 16: 515-527. doi: 10.1890/1051-0761(2006)016[0515:FBANSA]2.0.CO;2

National Wildfire Coordinating Group [NWCG]. 2008. Interagency standards for fire and fire aviation operations 2008. National Interagency Fire Center NFES 2724 (January). <http://www.nifc.gov/policies/red_book.htm>. Accessed 11 December 2008.

Natural Resources Conservation Service [NRCS]. 2008. Plant fact sheet: Lemmon's needlegrass *Achnatherum lemmonii* (Vasey) Barkworth. <http://plants.usda.gov/factsheet/pdf/fs_acle8.pdf>. Accessed 12 December 2008.

Neary, D.G., C.C. Klopatek, L.F. DeBano, and P.F. Ffolliott. 1999. Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management 122: 51-71. doi: 10.1016/S0378-1127(99)00032-8

Odion, D.C., E.J. Frost, J.R. Strittholt, H. Jiang, D.A. DellaSala, and M.A. Moritz. 2004. Patterns of fire severity and forest conditions in the western Klamath Mountains, California. Conservation Biology 18: 927-936. doi: 10.1111/j.1523-1739.2004.00493.x

Parker, V.T., and V.R. Kelly. 1989. Seed banks in California chaparral and other Mediterranean climate shrublands. Pages 231-255 in: M.A. Leck, V.T. Parker, and R.L. Simpson, editors. Ecology of soil seed banks. Academic Press, San Diego, California, USA.

Perchemlides, K., P.S. Muir, and P.E. Hosten. 2008. Responses of chaparral and oak woodland plant communities to fuel-reduction thinning in southwestern Oregon. Rangeland Ecology and Management 61: 98-109. doi: 10.2111/07-026R1.1

Pfaff, E. 2007. Patterns of grassland, shrubland, and woodland vegetation abundance in relation to landscape-scale environmental and disturbance variables, Applegate watershed southwest Oregon. Thesis, Southern Oregon University, Ashland, USA. <http://soda.sou.edu>. Accessed 12 March 2010.

Reiner, A.L., N.M. Vaillant, J. Fites-Kaufman, and S.M. Dailey. 2009. Mastication and prescribed fire impacts on fuels in a 25-year old ponderosa pine plantation, southern Sierra Nevada. Forest Ecology and Management 258: 2365-2372. doi: 10.1016/j.foreco.2009.07.050

Schwilk, D.W., J.E. Keeley, E.E. Knapp, J. McIver, J.D. Bailey, C.J. Fettig, C.E. Fiedler, R.Y. Harrod, J.J. Moghaddas, K.W. Outcalt, C.N. Skinner, S.L. Stephens, T.A. Waldrop, D.A. Yaussy, and A. Youngblood. 2009. The national fire and fire surrogate study: effects of fuel reduction methods on forest vegetation structure and fuels. Ecological Applications 19: 285-304. doi: 10.1890/07-1747.1

Sikes, K.G., and P.S. Muir. 2009. A comparison of the short-term effects of two fuel treatments on chaparral communities in southwest Oregon. Madroño 56: 8-22. doi: 10.3120/0024-9637-56.1.8

Smith, E.P. 2002. BACI design. Pages 141-148 in: A.H. El-Shaarawi and W.W. Piegorsch, editors. Encyclopedia of environmetrics. Volume 1. John Wiley & Sons, Chichester, United Kingdom.

Suding, K.N., and K.L. Gross. 2006. Modifying native and non-native species richness correlations: the influence of fire and seed addition. Ecological Applications 16: 1319-1326. doi: 10.1890/1051-0761(2006)016[1319:MNAESR]2.0.CO;2

Taylor, A.H., and C.N. Skinner. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications 13: 704-719. doi: 10.1890/1051-0761(2003)013[0704:SPACOH]2.0.CO;2

USDA Plants Database. Name search. Natural Resources Conservation Service. <http://plants.usda.gov/>. Accessed 24 February 2010.

Western Regional Climate Center [WRCC]. 2008. Western Regional Climate Center individual weather station historical summary for Ruch, Oregon. <http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?or7391>. Accessed 9 March 2010.