

VEGETATION RESPONSE TO SEASONALITY OF PRESCRIBED
FIRE AND POSTFIRE SEEDING FOLLOWING MECHANICAL
FUEL-REDUCTION TREATMENTS IN OAK-CHAPARRAL
COMMUNITIES OF SOUTHWESTERN OREGON

By

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To my partner in life and love, David,
whose constant support and companionship,
especially during those long, hot days on steep,
south-facing slopes covered in poison oak,
made it possible for me to complete this study.

~

Deep in their roots, all flowers keep the light.
– *Theodore Roethke*

In memory of my grandma Ruby
December 23rd, 1907 ~ June 23rd, 2008
whose 101 years of wisdom gently
reminds me how to “keep the light.”

*Thanks gramcracker,
a bushel and a peck with an extra
BIG hug around the neck!*

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ABSTRACT OF THESIS

VEGETATION RESPONSE TO SEASONALITY OF PRESCRIBED FIRE AND POSTFIRE SEEDING FOLLOWING MECHANICAL FUEL-REDUCTION TREATMENTS IN OAK-CHAPARRAL COMMUNITIES OF SOUTHWESTERN OREGON

by Celeste Tina Coulter

Several thousand acres of oak-chaparral within the wildland-urban interface of the Applegate Valley of southwestern Oregon have been mechanically treated by brush mastication to reduce hazardous fuels. Land managers are faced with the challenge of minimizing wildfire hazard while maintaining species richness in degraded oak-chaparral communities. High fuel loads left on the ground following mechanical fuel-reduction treatments have the potential to produce severe-intensity fires that may have a detrimental effect on soils and seedbanks. Over time, as fuel loads decay, the reduction in slash may allow for prescribed fire and postfire seeding. Together, these treatments may minimize invasion by non-native species while retaining local native species diversity derived from the surviving seedbank.

I examined the response of vegetation to seasonality of prescribed fire and postfire seeding in mechanically masticated oak-chaparral communities of the Applegate Valley in southwestern Oregon. Permanent plots were installed at two sites, China Gulch

and Hukill Hollow. At each site, 30 1-m² paired plots (seeded and unseeded) were sampled in each of four treatment blocks: spring burn, spring control, fall burn, and fall control. Fall prescribed fires were conducted in October 2005 and spring prescribed fires were conducted in April 2006. Four native bunchgrass species were used to test postfire seeding in burned and unburned plots: *Achnatherum lemmonii*, *Bromus carinatus*, *Elymus glaucus*, and *Festuca idahoensis* ssp. *roemeri*. Soil samples were collected and analyzed before and after prescribed fire treatments. Pre-treatment vegetation surveys were conducted in summer 2005 and post-treatment vegetation surveys in spring 2006, 2007, and 2008.

The patchy, low-intensity spring burns were dramatically different from the moderate- to severe-intensity fall burns at both sites. Mortality of mature *Quercus garryana* was observed in fall burn treatment blocks at China Gulch and Hukill Hollow. Native species significantly decreased following fall prescribed fire treatments, while invasive annual grasses increased at both sites. Spring prescribed fire treatments did not significantly affect the abundance of invasive species at either China Gulch or Hukill Hollow. Germination of seeded bunchgrass species was successful following fall prescribed fires at both sites. Germination did not occur following spring prescribed fires or in control treatments at China Gulch or Hukill Hollow. Prescribed fire treatments did not noticeably impact soil nutrient levels. Species richness was highest in the first postfire year across all treatment blocks. At both sites, abundance of exotic species peaked in the second postfire year. Exotic annual grasses have remained the dominant life form group in fall burn treatment blocks. Three years following prescribed fire treatments

a significant number of woody seedlings were observed, with the largest increases in control blocks where fire did not occur. Despite the establishment of invasive annual grasses following fall prescribed burns, postfire seeding may be a viable solution for the prevention of exotic annual grass invasion. The experimental design resulted in a matrix effect, with seeded plots dominated by sown bunchgrass species and non-seeded plots dominated by invasive annual grass. Broadcast native seed applications following fall prescribed fire may ultimately meet management goals. While long-term monitoring of study sites will provide a more comprehensive analysis of the effects of seasonality of prescribed fire and postfire seeding, significant oak mortality and increases in invasive annual grasses remind us that fire should be prescribed with care.

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INTRODUCTION

Oak woodlands and shrublands of the Applegate Valley, located in the eastern Siskiyou Mountains, are characterized as the northernmost extent of the Mediterranean climate in North America driven by cool, wet winters and hot, dry summers (Keeley 2002, Detling 1961). The plant assemblages of this valley are grouped primarily within the California Floristic Province encompassing flora from northwest California, the Klamath Range regions, and the Great Basin Province (Hickman 1993). Riegel et al. (1992) describes the oak woodlands of southwestern Oregon as a transitional community from the Scott and Shasta Valleys of northern California and the Willamette Valley of northwestern Oregon. Receiving less than 640 mm of annual rainfall, this region supports sclerophyllous woodlands and shrublands similar to those found in California (WRCC 2008). The Siskiyou Mountains are globally recognized as a center for endemism with a history that is as varied as the species it supports (DellaSala et al. 1999).

Southwestern Oregon is unique to the western Oregon region in supporting a mixed-severity fire regime with highly variable fire frequencies (Agee 1991, Odion et al. 2004, Taylor and Skinner 2003). During 1690-1930, historic fire-return intervals for dry Douglas-fir forests in the western Cascades are speculated as 80-100 years (Agee 1991). By contrast, evidence from southwestern Oregon forests suggests a fire-return interval of

49 years (Agee 1991). At its peak, fire-return intervals for this region reached 12-16 years following Euro-American settlement (Agee 1991).

Despite the frequent and persistent presence of fire in the Applegate Valley, recent research suggests that the plant communities we observe today are primarily a result of the interaction between topographic, edaphic, and climatic variables (Pfaff 2007, Hosten *in prep.*). Fire, both its use and its suppression, has long since been promulgated as the defining variable in explaining present-day species composition in fire-prone ecosystems (Keeley et al. 2006b). Hosten (2006) presents a different outlook for the Applegate Valley by arguing that describing extant plant communities as an outcome of an elongated fire-return interval oversimplifies our understanding of landscape change over time. Patterns of vegetation change in the Applegate Valley can only be elucidated when we overlay topographic, edaphic, and climatic variables—together with fire history—on the consequences of public and private land management disturbances. By exploring this matrix of abiotic factors, coupled with natural and human-induced disturbance regimes, we begin to understand complex vegetation patterns found in the Applegate Valley of southwestern Oregon (Hosten 2006).

While fire has played a role in shaping present-day plant communities in the Applegate Valley, current fire management practices, including the use of prescribed fire for restoration, have the potential to permanently alter the successional trajectories of native and non-native communities (Keeley et al. 2005b). Even though the last century in southwestern Oregon has experienced extreme highs and lows in fire frequency and intensity, research suggests that many of the non-coniferous plant communities have

remained relatively unchanged (Hosten *in prep.*). While tree and shrub encroachment has been documented, chaparral communities continue to support high- severity fires despite changes in stand-density levels and species composition. Similarly, open grasslands and oak woodlands have remained resilient to woody species encroachment from fire suppression due to the constraints of soil factors (Hosten et al. 2007). Frequent low-severity fires continue to shape grassland and oak communities of the Applegate Valley (Hosten and DiPaolo *submitted*).

How and where fire was used in the past, and how and where we use it today, are fundamental questions facing land managers. Accordingly, before we can examine the effects of current fire management practices, it is essential to understand how fire evolved within the eastern Siskiyou Mountains. The first recorded use of fire by hunter-gatherer populations settled in the Applegate Valley is dated at A.D.1695 (LaLande 1995). Early human settlements used fire as a means for managing plant populations for food, tools, ceremonies, and warfare, and to facilitate hunting and traveling (Pullen 1996). Indigenous populations of the Applegate Valley cultivated fire with respect and sensitivity to its destructive capacity (Pullen 1996). Fire was carefully controlled by using low-intensity burns to manipulate plant communities (Pullen 1996). Two distinct times of year, spring and late summer, were reserved for burning times, and burning was often done at night (Pullen 1996). “Fire setters” held an important position within the community, partly due to the fact that tribal settlements were located within dense thickets and woodlands to

conceal them from nearby enemies (Pullen 1996). Since much of the burning took place near settlements, it was crucial that fires were contained and not at risk of escaping (Pullen 1996).

Late in the 1820's, trappers such as Peter Skene Ogden and Alexander McLeod from the Hudson Bay Company began exploring the Applegate Valley. Between the years 1840 and 1855, Euro-American communities became established within the valley (LaLande 1989). By 1856, following the last "Rogue Indian War," the remaining Native American groups were removed to a reservation on the north-central coast of Oregon. Shortly after the establishment of white settlers, human-set fires increased dramatically in intensity, frequency, and scale (LaLande 1995).

Alterations of anthropogenic fire events were driven by the changes in attitudes toward fire by Euro-Americans. Modifications to the landscape included the use of fire to clear land for gold prospecting; enhance grass communities for grazing; facilitate hunting, farming and logging activities; and maintain trails (LaLande 1995). Early white settlers burned throughout the year rather than restricting burning practices to spring and late summer (LaLande 1995). Furthermore, previously unburned mid-elevation land was burned as a result of mining activities. Accidental ignitions and "burning for enjoyment" also contributed to the growing frequency, size, and intensity of fire in the valley (LaLande 1995). Human-set fire events peaked from 1860 to 1920, with 1902 and 1910 recorded as extreme fire events in the history of the Pacific Northwest (Agee 1991). It is interesting to note that many of the initial white settlers to the Applegate Valley

descended from Appalachian pioneers who brought with them their own traditional burning practices as part of their culture in rural regions of the southeastern United States (LaLande 1995).

By 1906, rangers for the United States Department of Agriculture (USDA) Forest Service arrived in the eastern Siskiyou Mountains to oversee the Crater (Rogue-Siskiyou) National Forest (LaLande 1995). The Applegate and Rogue Valleys became the focus of Forest Service efforts to constrain the use of fire in the region (Agee 1991). Despite the presence of Forest Service rangers, the decades between 1910 to 1930 experienced the most concentrated and unregulated period of human-set fires (LaLande 1995). During 1920, the Forest Service issued a statement that the Applegate Ranger District averaged “32 fires a year with an average of 3 fires growing to a ‘Class C’ (large size)” (LaLande 1995). Dense smoke accumulations in both the Applegate and Rogue valleys plagued residents and drove away tourists (LaLande 1995). As the Forest Service became established in the eastern Siskiyou Mountains, fire-fighting jobs brought badly needed work to both the Applegate and Rogue Valleys. Consequently, the prospect of good paying jobs to fight fires increased incidents of arson (LaLande 1995). In fact, rural valley residents were such fearless advocates of burning that the eastern Siskiyou Mountains were considered the “center of incendiarism” of southern Oregon in the early 20th century (LaLande 1995).

It was not until the late-1920’s that the concept of fire suppression began to take hold in the minds of local residents. It was also during this period, however, that the Crater (Rogue-Siskiyou) Forest Supervisor acknowledged the benefits ranchers were

finding in using fire to restore rangeland habitat for grazing (LaLande 1995). There was some discussion of implementing a prescribed burning policy in the Applegate Valley, but prominent residents in nearby towns like Jacksonville and Ashland voiced their opposition (LaLande 1995). In the end, it was the onset and subsequent distraction of World War I (1916) that brought the policy of fire suppression into practice in the Applegate Valley, and laid to rest the controversy of prescribed burning (LaLande 1995, Agee 1991).

Since then, years of fire suppression and continued development in fire-prone ecosystems have collectively increased wildfire occurrences involving life and property, allowing fire to rise to a prominent place on the political agenda (Daniel et al. 2007, Dombeck et al. 2004). What is different about fire then and fire now is that, historically, fires were typically low-intensity ground fires or high-intensity crown fires, rather than *high-intensity ground* fires that occur today in areas where fuel-reduction treatments are implemented. Another significant change to the valley is the increase in the number of private residences found throughout lowlands and mid-elevation slopes (Tong et al. 2004). The checkerboard ownership of BLM land interspersed with private land creates thousand of hectares of wildland-urban interface. Davis (1989) more accurately redefined this term as “mixed interface,” referring to regions where private land is embedded in a wildland matrix, as we find in the Applegate Valley.

The concerns of residents living in the wildland-urban interface dominated by fire-prone vegetation have prompted efforts to reduce fire hazards in those areas (Bury 2004, USDA 2007, Kauffman 2004). At the end of the 2000 fire season—considered a

landmark year for large fire events, although since then many western states have recorded their largest fire years yet—government agencies instituted the National Fire Plan, to develop response strategies to wildfire and communities affected by large fire events (Daniel et al. 2007, USDA 2008). The Bush administration’s Healthy Forests Initiative, which later became the Healthy Forest Restoration Act, approved an annual sum of 760 million dollars for fuel-reduction activity on 20 million acres of public land, with more than half of funds directed to wildland-urban interface areas (Daniel et al. 2007). Under the act, communities are provided with incentives for property owners to prepare for wildfire by proactively reducing fuels on their lands (Daniel et al. 2007, USDA 2008). The Healthy Forest Restoration Act also streamlines thinning projects by limiting judicial review and by reducing the amount of environmental analysis required under the National Environmental Policy Act of 1969 (Daniel et al. 2007, USDA 2008). Since 2002, thousand of hectares of wildland-urban interface lands (both private and public) have been thinned as a means to reduce fire hazards in Oregon. The 2006 fire season saw the highest number of hectares treated for fuel-reduction, totaling 57,734 hectares for the state (USDA 2007).

Currently, three fuel-reduction treatments are used within the wildland-urban interface of the Applegate Valley: (1) hand cut, pile, and burn; (2) mechanical mastication; and (3) prescribed fire (Brunson and Shindler 2004). In many cases, fuel-reduction treatments are intended to accomplish two goals: reduce the risk of fire and restore plant communities to pre-fire exclusion density levels by acting as a

fire-surrogate. This paper will delve further into the efficacy of mechanical mastication as a fire surrogate in the oak-chaparral communities.

In this study, mechanical mastication is accomplished with a large rotating blade (BM-Slashbuster®) attached to a track-mounted excavator. The blade is lowered onto shrubs and trees, shredding them down to < 0.5 m stumps. The slash left behind from brush mastication is either burned or left on the ground to decompose. Mechanically masticated land is left with pockets of unthinned chaparral, called “leave islands,” ranging from 0.04 to 0.4 hectares, scattered mainly in draws of treated units to protect riparian areas (Tong et al. 2004). The rationale behind this fuels-reduction treatment is to bring fuel loads down to the ground. If, later, fire occurs in these treated areas, the intended result will be a low-intensity burn more easily controlled than the characteristic crown fires observed in untreated woodlands and shrublands. Even so, fires implemented within one year of mechanical treatment have been described as slow, higher-intensity surface fires that detrimentally affect soils, native seedbanks, and remaining tree and shrub longevity (Keeley 2006). Local experience suggests that approximately half of the slash biomass is estimated to decompose within five years. Reduction in biomass may allow for follow-up burn treatments and native grass establishment while retaining local native species diversity derived from the surviving seedbank. How prescribed fire impacts the successional trajectory of plant communities in mechanically treated oak-chaparral communities of southwestern Oregon has not been investigated. This study will build on related studies comparing fuel-reduction treatments by evaluating the seasonality of prescribed fire (spring and fall) on mechanically masticated oak-chaparral three to four

years following treatment on native and non-native species. Research has found that non-native species decrease in abundance following late spring prescribed fires, while fall prescribed fires increase native species diversity in chaparral shrublands of southern California (LeFer and Parker 2005). Invasive species have also been known to increase following spring prescribed fire treatments by delaying the germination of native species (LeFer and Parker 2005). Delayed native annual plant responses leave bare ground open to colonization by exotic annual grasses (Keeley 2001).

Furthermore, related studies have found that ground disturbance caused by mechanical treatments create opportunities for non-native plants to become established (Sikes 2005, Perchemlides et al. 2008, Keeley 2002). Land managers attempt to diminish this effect by seeding with native grasses to colonize bare ground and minimize invasion by non-native species. The relative success of these seeding treatments has not been formally researched in southwestern Oregon and will also be addressed in this study. Native bunchgrass species have been described as having a direct influence on the extent of native species diversity (Maslovat 2002). Bunchgrasses increase resource availability and decrease soil-surface temperatures, allowing other native forbs to thrive (Maslovat 2002). Many studies have cited the failure of seeding treatments to prevent erosion of rock and soil (Keeley et al. 2006a, Keeley 1996). Other studies have observed that postfire seeding can sometime be too successful and ultimately prevent the germination and survival of native forbs (Keeley et al. 2006a).

Beginning in 2005, a concerted effort to increase our understanding of how the plant communities of southwestern Oregon respond to fuel-reduction treatments was

undertaken by Dr. Paul Hosten (Medford District Bureau of Land Management) and Dr. Patricia Muir (Oregon State University). Together, with funding from the Joint Fire Science Program, they began a 3-part study entitled *Fuel-reductions in oak woodlands, shrublands, and grasslands of southwestern Oregon: consequences for native plants and invasion by non-native species* (Perchemlides et al. 2008; Pfaff 2007).

I will address the final piece of that study by exploring the effects of the seasonality of prescribed fire and postfire seeding on mechanically masticated oak-chaparral communities in the Applegate Valley. Ultimately, our goal is to develop a land management protocol for high fuel-load oak-chaparral sites that have been mechanically masticated to maintain native plant species richness and to establish a native herbaceous understory community that will protect soils and take the place of dominant woody species. I hypothesized that (1) native species richness would increase following fall prescribed fire; (2) abundance of invasive plant species would decrease following spring prescribed fire; and (3) seeded bunchgrass species would exhibit higher germination rates in prescribed fire treatment blocks.

METHODS

Study Areas

Two sites, China Gulch and Hukill Hollow, were selected for this study to represent the dominant plant communities found in the wildland-urban interface of the Medford District Bureau of Land Management Ashland Resource Area in the Applegate Valley of southwestern Oregon (Table 1) (Figures 1 and 2). Management activities have transformed both sites into a disturbance-mediated woodland/chaparral plant community dominated by *Arbutus menziesii* Pursh, *Quercus garryana* Douglas ex Hook., *Ceanothus cuneatus* (Hook.) Nutt, *Arctostaphylos viscida* Parry, *Bromus tectorum* L., and *Madia* sp. (Pfaff 2007). Each site was mechanically masticated, China Gulch in 2001 and Hukill Hollow in 2002, as part of the Little Applegate Fuel-reduction Project (Tong et al. 2004). The study sites experience a Mediterranean climate with cool, wet winters and hot, dry summers. Mean annual precipitation is 646 mm and mean temperature in January is 4.0°C and 20.8°C in July (WRCC 2008).

Historically, both sites used in this study were similar types of chaparral woodland/shrublands. The xeric, steep slopes of China Gulch supported a buckbrush chaparral shrubland dominated by *Ceanothus cuneatus* and *Bromus hordeaceus* L. (Hosten and Pfaff *in prep.*). Hukill Hollow, more mesic and less steep, was dominated by

a manzanita chaparral shrubland of *Arctostaphylos viscida*, *Pinus ponderosa* C. Lawson, *Quercus garryana*, and *Dichelostemma congestum*. Like China Gulch, fire exclusion enabled the growth of dense thickets of *Arctostaphylos viscida* and *Ceanothus cuneatus* (Hosten and Pfaff *in prep.*).

Table 1. Location and study site characteristics for China Gulch and Hukill Hollow.

	SITE 1: China Gulch	SITE 2: Hukill Hollow
Location	Jackson County, northwest of Ruch: T38S, R3W, Sec. 22, 30 m downslope of undeveloped road on ridge off China Gulch Road 853	Jackson County, south of Jacksonville: T39S, R2W, Sec. 7, 30 m downslope of road 39-7-7.1 off Sterling Creek Road 787.
Latitude/ Longitude	42.2468636; 122.0496019	42.1883492; 122.9783586
Topography	SE facing slope, undulating S to SE	SE facing slope, undulating SE to SW
Elevation	700 m – 714 m	697 m – 723 m
Slope	55%	35%
Soils	Vannoy-Voorhies complex (60% Vannoy, 30% Voorhies) 16-18% clay.	Vannoy-Voorhies complex (60% Vannoy, 30% Voorhies) 16-18% clay.

Figure 1. Site map of China Gulch

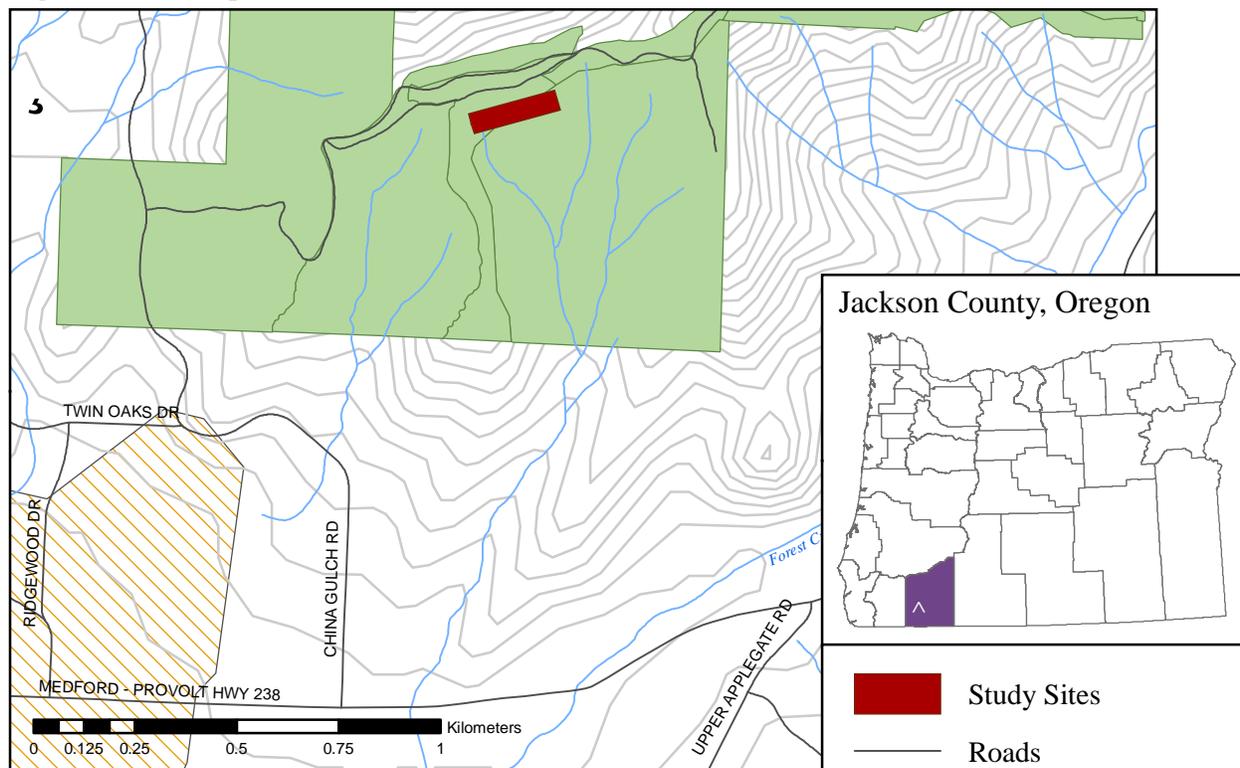
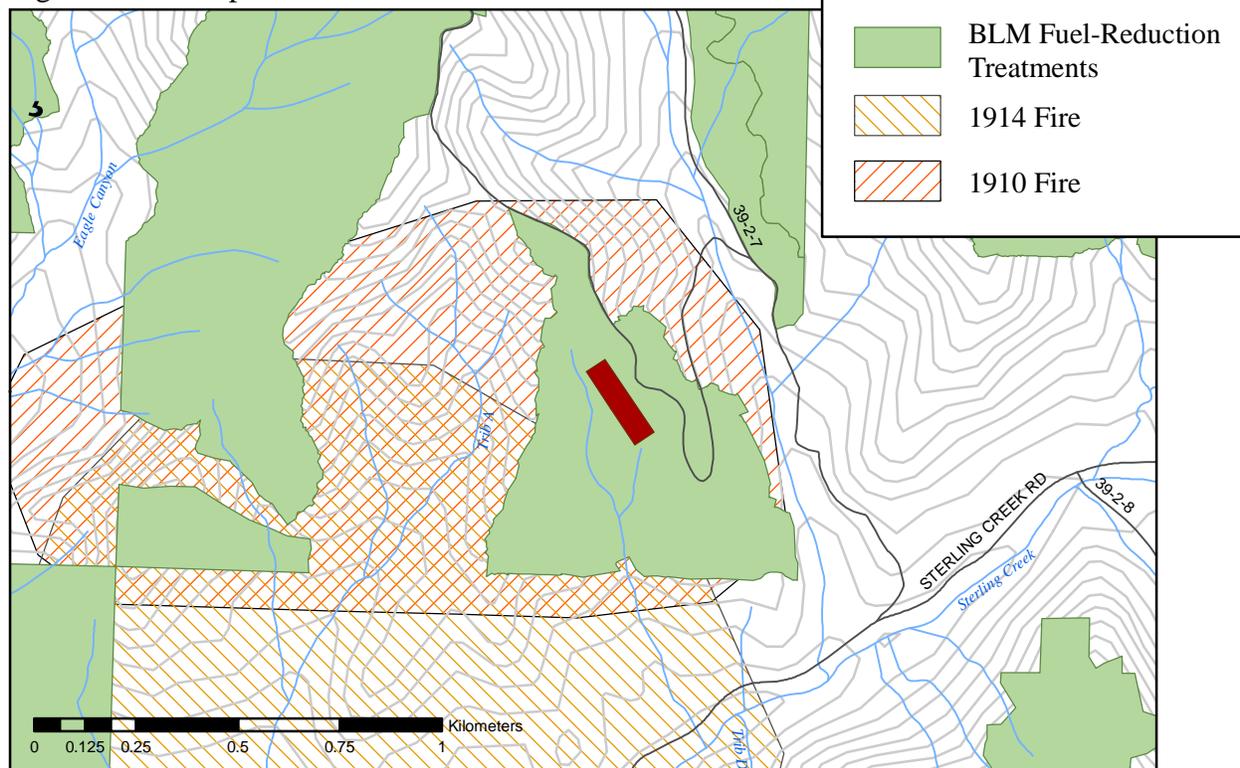


Figure 2. Site map of Hukill Hollow



Plot Establishment

At each site, 120 1-m² paired plots were installed, with metal stakes in each of four treatment blocks: spring burn, spring control, fall burn, and fall control. In each treatment block, 15 1-m² paired plots (30 total) were permanently installed. Two metal markers were placed diagonally in the NE and SW corners of each plot to ensure the same plot was sampled over consecutive years. In some cases, the soil was too shallow to allow for the metal stakes to be pounded into the ground. For these plots, instead of placing the metal stakes diagonally they were installed in corners on the same side (NE and SE corners). Each plot was identified with a numbered metal tag attached to the NE metal stake.

Treatment blocks measured roughly 40 m x 20 m and were located approximately 30 m downslope of the roads accessing the study areas. Blocks receiving prescribed fire treatment were flagged with a 30 m buffer on each of the four sides to reserve space for fire crews to create a hand fireline and mop-up zone after implementing the burns (Figure 3). Plots were established within each of the treatment blocks by tossing the quadrat frame to the NW corner of the treatment block. Subsequent plots were selected by spacing each plot one to two meters due east of the first plot, moving downslope when space required, until 30 plots were installed. All plots were oriented to run parallel with contour of slope. Paired plots were selected by evaluating adjacent plots similar in

dominant plant species. When placement of plot fell on a shrub, the quadrat was moved to the other side of the shrub, biasing the surveyed plant community towards forbs and grasses (Figure 4).

Figure 3. Sketch of study design for treatment blocks.

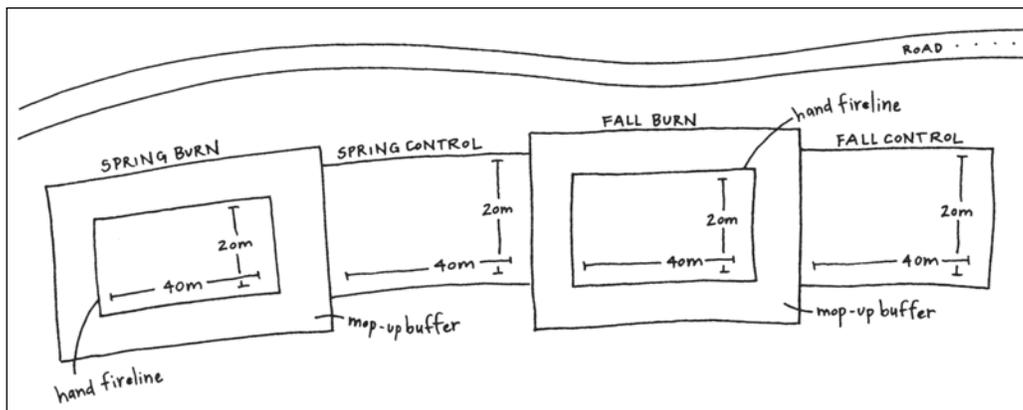
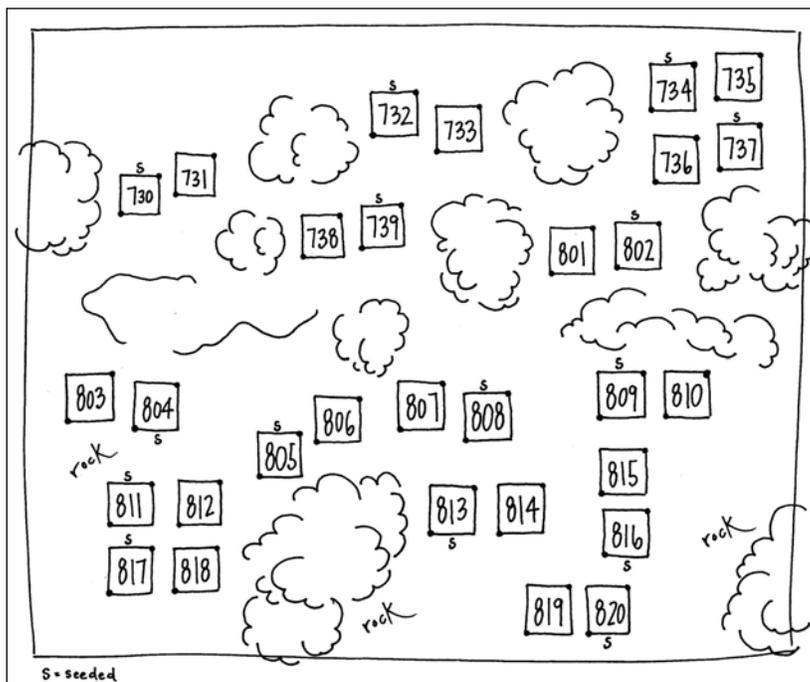


Figure 4. Sketch of plot diagram of spring control treatment block at China Gulch.



Sampling Methods

Pre-treatment vegetation surveys were conducted in August 2005. Prior to pre-treatment surveys, plant species were collected and identification verified. Post-treatment vegetation surveys took place in May 2006, August 2006, and May 2007. Plant codes, nomenclature, and authorship are listed according to the USDA Plants Database (<http://plants.usda.gov>). Surveys of density of surviving seeded bunchgrass germinants and mature *Quercus garryana* trees in fall burn treatment blocks at China Gulch and Hukill Hollow were conducted in May 2008. Observations of woody seedlings throughout all treatment blocks at China Gulch and Hukill Hollow were also documented in May 2008. Changes in plant communities in all treatment blocks at China Gulch and Hukill Hollow were also documented using digital photography on an annual basis in late spring from 2005-2007. Also in 2005 and 2006 additional photos were taken following spring and fall prescribed fires (Appendices H and I).

A 1-m² quadrat frame was constructed to delineate the outline of the plot. Recorded plant species were determined by placing the quadrat frame over the permanently installed metal stakes at each plot. Standard FIREMON protocol was used to estimate cover by assessing the area defined by the outside drip line of the plant crown. In some cases, the sum cover for all species in a plot totaled over 100% (Lutes et al. 2006). All plant species within each plot were recorded and assigned a cover class (Lutes et al. 2006). Cover estimates were entered for each species using the cover class code represented by the mean value of the cover class (Table 2). Plant species that had reached

senescence and were no longer identifiable were considered thatch. Environmental variables recorded as cover included rock, thatch, slash, burned ground, bare ground, and charcoal. Cover estimates for environmental variables were reached by assessing the total area defined by the variable within the plot. Those variables recorded by presence/absence were gopher mounds and browsing. The number of gopher mounds in each plot and any plant species that showed visible signs of browsing were also recorded. Ecological notes included deer trails through plots, lichen litter fall, rock outcrops near plots, and significant alterations to plots following prescribed fires. Following spring prescribed fire at Hukill Hollow, mature manzanita shrubs (*Arctostaphylos viscida*) split at the base of the plant and fell on plots hh895 and hh896 covering 80% and 40% of each plot respectively.

Table 2. Cover classes used to estimate plant species abundance and cover area for measured environmental variables (Lutes et al. 2006).

CODE	COVER CLASS
0	0 %
0.5	0 - 1 %
3	1 - 5 %
10	5 - 15 %
20	15 - 25 %
30	25 - 35 %
40	35 - 45 %
50	45 - 55 %
60	55 - 65 %
70	65 - 75 %
80	75 - 85 %
90	85 - 95 %
98	95 - 100 %

Soil Sampling Methods

Before treatments were applied, four soil samples, 10 cm in depth, were collected within each treatment block at both sites. Samples throughout both sites were characteristic of the Vannoy-Voorhies complex composed of approximately 16-18% clay.

Approximately 48 hours following prescribed fires, four soil samples, 10 cm in depth, were taken from each treatment block at both sites. Quart-sized bags were half-filled with large organic matter removed. The samples were then screened through a 1.981 mm soil screen (Tyler Standard Screen Scale), re-bagged, and labeled. Samples

were analyzed for organic matter (%C; N ENR lbs/A), P, K, Mg, Ca, Na, pH, SO₄^{-S} by A & L Western Agricultural Laboratories in Modesto, California.

Prescribed Fire Methods

Measurements of fuel loading and prescribed fires were implemented by fire crews from the Medford District Bureau of Land Management. Fall prescribed fires at China Gulch and Hukill Hollow were conducted on 6 October 2005. Spring prescribed fires at China Gulch and Hukill Hollow were conducted on 21 April 2006. All four burns measured approximately 0.4 hectare in size.

Fuel load data prior to spring and fall prescribed fires were collected using Browns transects across research plots at China Gulch (2 transects) and Hukill Hollow (3 transects) (Interagency standards for fire and fire aviation operations 2008). Following spring and fall prescribed fires, fuel load data were re-collected. Standard data collection methods were conducted according to the Interagency Standards for Fire and Fire Aviation Operations protocol (2008). A prescribed fire plan and complexity rating worksheet was completed for each of the four prescribed fires (Interagency standards for fire and fire aviation operations 2008).

All four prescribed fires were contained by using a strip-head firing pattern with a 3-4.6 m width between strips of fire. In areas of higher fuel loads a backing fire was utilized (Table 3). Tiles with OMERGALABEL© Model TL-10-105 temperature labels and OMEGAPELLETS© temperature indicating pellets PLT Series were oriented facing

upslope next to plots markers and numbered according to plot number. Tiles were buried 4.8 cm below the soil surface with temperature labels at a 3.8 cm depth. Pellets which burn at 650°C and 750°C were placed on the surface of the soil.

Table 3. General description of spring and fall prescribed fire containment methods.

Treatment	Prescribed Fire Containment Methods
China Gulch and Hukill Hollow Spring Burn	Used a strip-head firing pattern with a 3-4.6 m width between strips of fire. In areas of higher fuel loading, a backing fire was utilized.
China Gulch and Hukill Hollow Fall Burn	Used a strip-head firing pattern with a 1.5-3 m width between strips of fire. In areas of higher fuel loading, a backing fire was utilized.

Bunchgrass Seeding Methods

Four native bunchgrass species were used to test seeding in burned and unburned plots: *Achnatherum* (Vasey) Barkworth *lemmonii*, *Bromus carinatus* Hook. and Arn., *Elymus glaucus* Buckley, and *Festuca idahoensis* Elmer ssp. *roemeri* (Pavlick) S. Aiken. Of the four bunchgrass species, professionally cleaned seed was supplied by Medford District BLM for *Festuca idahoensis* ssp. *roemeri* and *Elymus glaucus*. Seeds of *Achnatherum lemmonii* and *Bromus carinatus* were cleaned at the Oregon State University Extension Services in Central Point, Oregon. All seeds used in this study were

collected within the Rogue Valley of southern Oregon. Germination rate, viability, and tetrazolium tests were conducted by the Oregon State University Seed Laboratory in Corvallis, Oregon (Table 4). Seeds were stratified at 10°C for seven days with the exception of *F. idahoensis* ssp. *roemeri* which was pre-chilled for 14 days. Germination and viability tests were performed by placing 100 seeds 1 cm apart in a container on top of filter paper soaked in a 0.2% solution of KNO_3^- (ISTA 2008). Four containers were prepared for each species (totaling 400 seeds) then placed in an alternating germinator with daytime temperature set at 25°C and nighttime temperature set at 15°C (ISTA 2008). Seeds were checked over a four-week period, counting and removing each germinating seed (ISTA 2008). Tetrazolium tests were performed in three steps: (1) preconditioning, (2) preparation, and (3) evaluation (AOSA 2000). Preconditioning involved soaking 200 seeds of each species in water overnight at a temperature of 25°C (AOSA 2000). To prepare the seeds, each seed was cut laterally just above the embryo, soaked in a 1% tetrazolium solution overnight at a temperature of 30°C, and then cleared with a 85% lactic acid solution for 30-45 minutes at a temperature of 30°C (AOSA 2000). Evaluation was conducted by recording which seeds stained, with viable seeds staining evenly and non-viable seeds remaining unstained (AOSA 2000).

Table 4. Oregon State University Seed Laboratory: report of seed analysis.

Variables Tested	<i>Achnatherum lemmonii</i>	<i>Bromus carinatus</i>	<i>Elymus glaucus</i>	<i>Festuca idahoensis ssp. roemerii</i>
Germination %	9%	97%	91%	60%
Total Viable %	9%	97%	91%	60%
Number of Seeds Tested	400 seeds	400 seeds	400 seeds	400 seeds
Days Tested	28 days	7 days	14 days	21 days
TZ %	88%	97%	88%	69%
Days Pre-Chilled	14 days	7 days	7 days	7 days

Germination = percentage of seed that produce normal seedlings in a test sample

TZ % = percentage of viable seeds in a sample in 24-48 hrs even if seeds are dormant

Days Pre-Chilled = scarification required for germination tests

Standard seed application methods suggest seeding 4.5 kg 0.4 ha⁻¹ of seed.

To replicate this quantity per acre in test plots, 1.12 g of seed per 1-m² plot was used for seeding treatment. A mixture of the four native bunchgrass species used in the seeding treatment was created by measuring 0.28 g of each individual grass species. Following prescribed fire treatment in fall of 2005 and spring of 2006, one randomly selected plot (chosen by coin toss) of each paired plot was sowed by hand with a measured bunchgrass mixture approximately 48 hrs after burning.

Germination success was first evaluated in 2006 with follow-up surveys in 2007 and 2008. Cover estimates were taken in 2006 and 2007, and density counts were taken in 2007 and 2008. To evaluate survival of seeded bunchgrass species that

germinated following the fall prescribed fire treatments, comparisons of mean cover for each germinant from 2006 to 2007 and density of live germinants from 2007 to 2008 were conducted.

Statistical Analysis

Vegetation data were collected in 2005, 2006, 2007, and 2008. Pre- and post-treatment group comparisons were conducted by evaluating 2005 data against 2007 data. Success and survival of seeded bunchgrass germinants following prescribed fires were evaluated by comparing 2006 and 2007 data. Response of vegetation to seeding and prescribed fires was evaluated using 2007 data as the best representation of treatment effect. Treatment effect on plant communities was analyzed with non-parametric, parametric, and descriptive statistical methods. Patterns of change gathered from the analyses were considered only when the results were supported by more than one statistical method. A P -threshold of 0.10 was used for each test. To control for the number of statistical tests performed, which increases the chance of a Type I error, I applied a Bonferroni adjustment to the threshold P value. Another reason for applying the Bonferroni adjustment involved the number of pairwise comparisons performed. For example, data were analyzed using t-tests by comparing 2005 burn plots versus 2007 burn plots and 2007 burn plots versus 2007 unburned plots, resulting in two pairwise comparisons. Therefore, the adjustment was applied by dividing the threshold P value by the number of comparisons (two) conducted (Elzinga et al. 1998). Subsequently, test

results were only considered significant if they fell below $P = 0.05$ (Elzinga et al. 1998). The assumption that data were randomly collected from the study areas was not met for either parametric test (paired t-test and ANOVA) due to unavoidable constraints of implementing the prescribed fires. All plots in burned treatment blocks were burned at the same time with the same prescribed fire treatment. Their grouping apart from other treatments constitutes pseudoreplication (Hurlbert 1984, Carpenter 1990).

Chi-square Test

Chi-square tests were used to examine change in frequency (presence/absence) of the most abundant plant species at China Gulch and Hukill Hollow. Tests were done using a two-by-three contingency table comparing data from 2005, 2006, and 2007 spring and fall burn treatments. A two-by-two contingency table was used to compare 2007 burn plots to 2007 control plots in both spring and fall treatments. Yates correction for continuity was not applied to presence/absence data since none of the expected frequencies was less than five (Elzinga et al. 1998).

Criteria used to determine which species were most abundant and frequent in each treatment block were determined by constraining the species list to those observed in greater than 24 plots (20% of the plots) and with a cover greater than one percent. The most common native/non-native plant grouping for China Gulch included 13 species when looking at the entire data set (2006-2007), and 15 species when considering data

from only the most recent collection period (2007). At Hukill Hollow, the most common native/non-native plant grouping list included 11 species when looking at the entire data set (2006-2007) or just the most recent (2007).

Paired t-Test

Paired t-tests were used to detect changes in plant community groups following spring and fall prescribed fires. Groups evaluated included (1) life form (native/exotic perennial forbs, native/exotic annual forbs, native/exotic perennial grasses, native/exotic annual grasses, native trees and shrubs); (2) native versus exotic species; and (3) species with highest abundance and frequency. Paired t-tests were also performed on variables measured in soil samples before and after burns were implemented. Data were formatted by using the sum of cover for all species in each plot. Since assumptions of normality and equality of variances do not apply to paired t-tests, as they do to the two-sample t-test, data were not transformed (Zar 1999). Instead, paired t-tests assume that only the differences between the two pairwise populations are normally distributed (Zar 1999). Paired t-tests are considered to be more powerful than two-sample t-tests when samples from each population are correlated as is the case in this study where plots were paired (Zar 1999). Furthermore, the proportion data for each species were recorded as a cover class rather than estimated to the nearest percentage. According to McCune and Grace (2002), using cover classes that are narrow at the extremes and broad in the middle can

approximate the function of an arcsine-squareroot transformation typically used in proportion data to meet the assumption of normality.

For each paired t-test, 5 different pair-wise comparisons were evaluated: (1) 2005 spring burn vs. 2007 spring burn; (2) 2005 fall burn vs. 2007 fall burn; (3) 2007 spring burn vs. 2007 spring control; (4) 2007 fall burn vs. 2007 fall control; (5) 2007 spring burn vs. 2007 fall burn.

Analysis of Variance

ANOVA tests were performed using proportion data recorded for plant abundance and environmental variables. Data were transformed with an arcsine-squareroot transformation using PC-ORD 4.0 to meet the assumption of normality, even though Zar (1999) and Elzinga et al. (1998) agree that both the ANOVA test and the t-test are robust enough to compensate for a slight deviation from the assumptions of normality and equal variance among populations. Given that multiple comparisons were not performed, applying the Tukey test was not necessary (Elzinga et al. 1998).

Single-Factor ANOVA

Single-factor ANOVA tests were used to compare the percentage of ground burned following spring and fall prescribed fires at China Gulch and Hukill Hollow. Independent variable (x) was seasonality of burn (spring versus fall) and dependent variable (y) was percentage of plot burned. (Zar 1999)

Two-Factor ANOVA with Replication

Two-factor ANOVA with replication was used to evaluate the difference in the sum cover of germinants between spring and fall prescribed fires, burned plots and control plots, and China Gulch and Hukill Hollow (Zar 1999).

Several different combinations of variables were examined. Independent variables (x , 1st factor) included site (China Gulch versus Hukill Hollow), and year of sampling (2005 versus 2007, 2006 versus 2007). Independent variables (x , 2nd factor) were seasonality of burn (spring versus fall). Dependent variables (y) included cover of seeded bunchgrass germinants, cover of individual seeded bunchgrasses, and percentage of plot burned (Zar 1999).

Multivariate Data Analysis

Multivariate data analyses were conducted using PC-ORD 4.0 statistical software to explore patterns of change in the plant communities at China Gulch and Hukill Hollow.

A valuable tool for analyzing community data, PC-ORD helps ecologists elucidate patterns or structure in plant or animal communities that otherwise might go unnoticed (McCune et al. 2002). Multivariate analysis serves two basic roles in the study of community ecology: (1) it helps ecologists discover structure or patterns in the data; and (2) it provides relatively objective, easy summarizations of the data, which facilitate the comprehension of the data and provide a means for effective communication of the results (McCune et al. 2002). The multivariate analysis techniques used in PC-ORD can contribute to hypothesis generation (McCune et al. 2002). Using these techniques allows for the exploration of preliminary findings. Then, once determined, correlations between variables can be tested for significance using other statistical methods (McCune et al. 2002).

Analyzing ecological community datasets becomes challenging due to the lack of normality and large number of zeros (species observed infrequently during sampling) often present in community data. PC-ORD allows an investigator to get around these problems by examining the data using different methods. If different analytical methods in PC-ORD all tell the same story (i.e., give similar results), one can reasonably assume

that the community pattern or structure (similarities or dissimilarities among species or groups) is real (McCune et al. 2002).

Two of the statistical methods used to analyze plant community and environmental data in PC-ORD are presented in this paper: Nonmetric Multidimensional Scaling (NMS) ordinations and Mantel test group comparisons.

Nonmetric Multidimensional Scaling Ordination

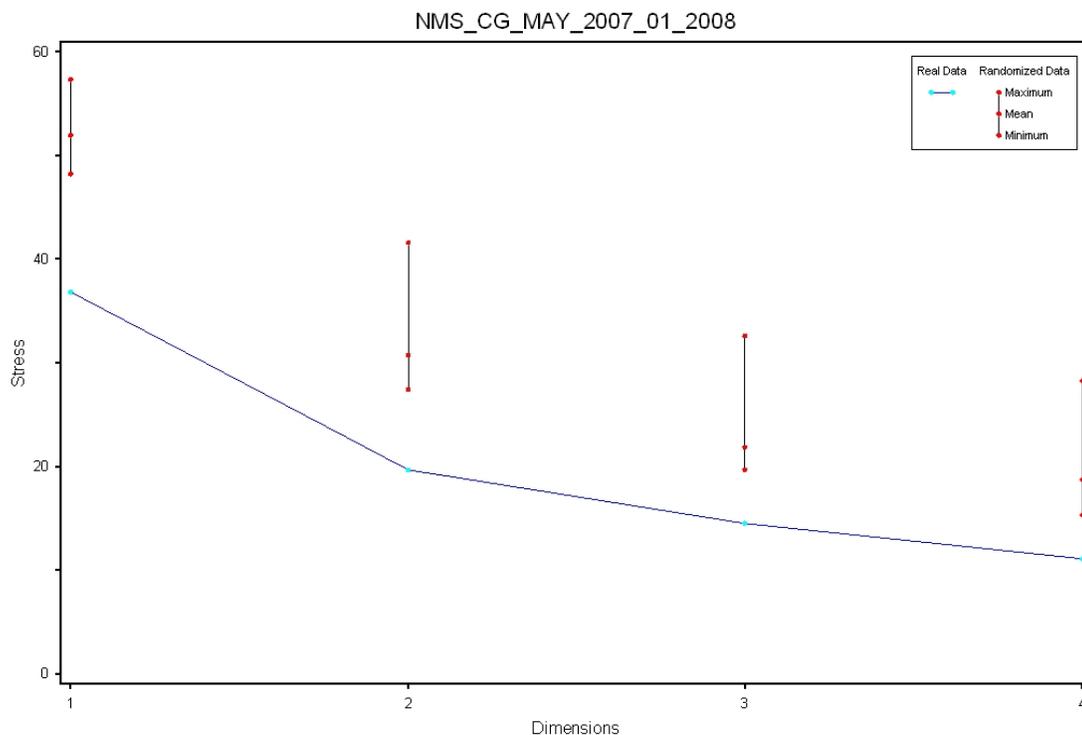
Ordination methods are used to determine the order of individuals based on their correlation with underlying environmental variables (Kent and Coker 1994). Ordination techniques allow scientists to perform data reduction and exploration that may lead to hypothesis generation. Areas of plant research that are addressed with ordination methods include (1) summarizing plant communities and the variation existing within habitat being studied; (2) defining individual species distributions within a larger community; and (3) summarizing variation between different communities and identifying environmental variables that define those different communities (Kent and Coker 1994). In this study, ordinations were used to illustrate patterns of species composition.

The NMS ordination was chosen because it is the most suited for community data (McCune et al. 2002). NMS works by iteratively ranking and placing variables into a reduced dataset that retains as much of the structure of the original dataset as possible (McCune et al. 2002). The ranking process based on distances tends to linearize the relationship between distances measured in species space and the distances in

environmental space. This function relieves the “zero truncation” problem, which is common in community data sets (McCune et al. 2002).

After performing the NMS ordination, a scree plot (defined by a downward-trending slope) was generated to determine the amount of stress in the ordination structure which tests for the assumption of monotonicity (Figure 5). Stress is a measure of the distance of departure from monotonicity (defined by an upward-trending slope). When the real data fall above or within the randomized data, then stress is high and the data do not meet the assumption of monotonicity. Figure 5 shows the real data falling outside the range of the randomized data, illustrating that there is sufficient structure in the ordination. The real data line shows that it begins to level off below the value of 20 on the y-axis. The leveling of the line shows the point where stress is reduced. (McCune and Grace (2002) warn that stress reduction of more than 20 indicates that the ordination contains too much noise. Reduction in stress was evaluated by observing the position of the trend line within each dimension (axis). McCune and Grace (2002) state that it is preferable for the largest reduction in stress to occur after dimension one with a leveling off in dimensions two or three.

Figure 5. Scree plot for China gulch NMS ordination on May 2007 PCOMP data.



Initial NMS ordinations were performed on autopilot to obtain the lowest number of dimensions (axes) for the dataset. The ordination is then run on manual, plugging in the dimensions determined from autopilot mode. After accepting the number of dimensions as three, the ordinations were performed in manual mode. Number of real runs were entered as 50, the stability criterion as 0.0005, the number of iterations between 400, starting coordinates as random, and Sorensen Bray-Curtis as the distance measure (McCune et al. 2002).

Graphed ordinations contained four different symbol shapes, with each symbol representing 30 plots in each of the four treatment blocks (spring burn, spring control, fall burn, fall control). Distance of the symbols from one another in ordination space represents how similar or dissimilar they are to one another. Thus, two symbols (plots) very close together are very similar; two symbols (plots) far apart are dissimilar (McCune et al. 2002).

In an NMS ordination, the axis numbers have no order of importance and are arbitrary. Different combinations of axes were compared to find the best grouping pattern. The ordination graphs were rotated to enhance alignment with axes based on a variable from the first (species abundance) or second (environmental variables) matrix. The joint plot function was added to create a vector overlay displaying the environmental variables most strongly correlated with plant species abundance. The longer the vector lines, the more correlation between species abundance and the associated environmental variable. Vectors perpendicular to and opposite each other are correlated. Lowering the cut-off r^2 value increased the length and number of vectors. Cut-off r^2 values ranged from 0.1 to 0.2 (Figures 25-29). Default scaling (% to Max) was selected to display ordination points based on similarity in proportion to the longest axis (McCune et al. 2002).

Mantel Test Group Comparison

Group comparisons were performed using the Mantel test. McCune and Grace (2002) recommend this test for groups that have the same number of sample units (rows). The Mantel test works by randomly shuffling the rows and columns in one matrix (post-treatment data) and then comparing it to the other non-randomized matrix (pre-treatment data) using a distance measure to explore similarity between the two matrices. If the randomizations result in frequent correlations between matrices that are as strong as comparisons between the original non-randomized matrices, then little or no confidence is observed in that relationship. This test allowed me to ask whether the plant community was fundamentally altered after treatment, testing the null hypothesis of no correlation between plant abundance and diversity between pre-treatment and post-treatment groups (McCune et al. 2002).

Grouped comparisons using the randomization (Monte Carlo) method were conducted on pre- (2005) and post- (2007) spring and fall prescribed fire treatments. The size of matrix varied according to the groups evaluated. Sorensen (Bray-Curtis) was used as a distance measure and time of day was used as random number seed supplier for 1000 runs.

The result for a Mantel Test is a text file rather than a graph. At the end of the result file appears the Z statistic. A positive association between matrices is indicated by an observed Z that is greater than the average Z from the randomized runs (McCune et al. 2002). Therefore, a significant value of P indicates there was not an effect of treatment

based on dissimilarity between groups. In other words, the matrices were similar even after randomizations were performed. The Pearson correlation coefficient (r) is a measure of -1.0 to 1.0. A strong correlation between matrices is indicated as r approaches a value of 1.0 (McCune et al. 2002).

Microclimate Study Methods

During spring 2005 a short-term study was conducted at Hukill Hollow to evaluate the effect of microclimate differences between thinned and unthinned oak-chaparral stands. A direct comparison of the microclimate in two stands, thinned and unthinned, required that both stands have a similar dominant overstory (correlating to canopy cover), aspect, elevation, and slope. The Hukill Hollow site was selected for this microclimate study because it contains a substantial number of unthinned chaparral thickets or “leave islands” in the draws of the unit. Thinned areas were brush masticated in 2001.

Site selection for the microclimate study involved recording data for aspect, elevation, and slope to evaluate the best suited counterparts. Comparison in elevation was done by sight, setting up the plots on an east-west longitudinal line perpendicular to the slope. Data on aspect and slope were recorded with a clinometer (Suunto PM-5\SPC) and compass (Silva “The Ranger” Type 15T). In addition to these measurements, the dominant understory and overstory plant communities were recorded (Table 5).

A woody plant (*Quercus garryana*, Oregon white oak seedling) common to each site was selected to record environmental variables.

Microclimates of thinned and unthinned oak-chaparral stands were measured with micrometeorological instruments attached to a datalogger (Campbell Scientific CR10 Datalogger). Environmental variables recorded included light intensity (GASP phosphied photocell calibrated by LICOR 190S Quantum sensor), air temperature and relative humidity (Campbell Scientific (CS500) temperature and humidity probe), leaf temperature (fine wire copper Constantan thermocouple), soil temperature (coarse wire copper Constantan thermocouple), and relative water content (CS616 water content Reflectometer) on an hourly and daily basis with hourly averages of 10-second readings for 360 measurements per data point.

Equipment setup involved attaching a thermocouple to a healthy leaf on an oak seedling with breathable surgical tape at 0.5 m in height to measure leaf temperature. Light, air temperature, and relative humidity sensors were placed 0.5 m off the ground. A relative water content probe was inserted to a depth of 25 cm, and a soil temperature probe was inserted 5 cm into the soil. Once the sensors were in place, the data loggers were activated. On day six, the data loggers were disconnected and equipment was removed from each site. Information recorded and stored on the data loggers was downloaded into Microsoft Excel (2003) for further analysis.

Table 5. Location and study site characteristics for microclimate study.

	Site 1: THINNED	Site 2: UNTHINNED
Location	Jackson County, south of Jacksonville: T39S, R2W, Sec. 7 (3-4 acres), 30 m downslope of road 39-7-7.1 off Sterling Creek Road 787.	
Latitude/ Longitude	42.1883492; 122.9783586	
Aspect	175°	140°
Canopy Cover	0%	60%
Topography	Undulating SE to SW	
Elevation	697 m – 723 m	
Slope	36%	30%
Soils	Vannoy-Voorhies complex (60% Vannoy, 30% Voorhies) 16-18% clay	
Plant Community	Trees: <i>Quercus garryana</i> Shrubs: <i>Arctostaphylos viscida</i> , <i>Ceanothus cuneatus</i> , <i>Toxicodendron diversiloba</i> Forbs: <i>Bromus</i> spp., <i>Clarkia rhomboidea</i> , <i>Clarkia purpurea</i> , <i>Dichelostemma congestum</i> , <i>Eriophyllum lanatum</i> , <i>Daucus pusillus</i> , <i>Lotus micranthus</i> , <i>Madia</i> sp., <i>Phacelia heterophylla</i> , <i>Torilis nodosa</i>	

RESULTS

Comparisons of Spring and Fall Prescribed Fires at China Gulch and Hukill Hollow

The spring prescribed burn at China Gulch resulted in a low-intensity, patchy burn pattern leaving half (15) of the plots unburned. At Hukill Hollow, the spring prescribed burn also resulted in a patchy burn pattern leaving 6 out of 30 plots unburned (Table 6) (Appendices H and I).

Fall prescribed burn treatments at both China Gulch and Hukill Hollow were classified as moderate- to severe-intensity burns. All 30 plots were completely burned. Overstory vegetation, primarily *Quercus garryana*, was either killed or set back by the intensity and duration of fire (Table 6) (Appendices H and I). Temperatures below the soil surface recorded during fall prescribed fires at both China Gulch and Hukill Hollow ranged between 40°C-82°C. Soil surface temperatures ranged between 490°C-710°C with a flame residence time at 3.75 minutes around *Q. garryana*.

A comparison of the mean percentage of ground burned in plots between China Gulch and Hukill Hollow spring and fall prescribed fires yielded a significant difference (ANOVA two-factor: $P = 0.001$). The amount of ground burned in fall burns at both sites was the same; however, more fuel was consumed following the spring burn at Hukill Hollow than at China Gulch (Appendix D).

Table 6. Description of spring and fall prescribed fire intensity.

Treatment	Spring and Fall Prescribed Fire Intensity
China Gulch Spring Burn	LOW INTENSITY BURN: Flame length averaged 0.6-0.9 m in height. Fuels did not burn completely. Light fuel loading and green live fuels made for a patchy burn pattern.
Hukill Hollow Spring Burn	LOW INTENSITY BURN: Flame length averaged 0.6-1.2 m in height. Fuels did not burn completely. Soil moisture was high resulting in some of the slash fuels that were touching the soil to be too wet to burn. Light fuel loading and green live fuels made for a patchy burn pattern.
China Gulch Fall Burn	MODERATE TO SEVERE INTENSITY BURN: Flame length averaged 0.9-1.8 m in height. All size classes (1-hour to 10,000-hour) of fuels burned completely with intensity. Overstory vegetation was mostly killed or set back by the intensity and duration of the fire.
Hukill Hollow Fall Burn	SEVERE INTENSITY BURN: Flame length averaged 0.9-3 m in height. All size classes (1-hour to 10,000-hour) of fuels burned completely with intensity. Overstory vegetation was mostly killed or set back by the intensity and duration of the fire.

In 2008, mortality of mature *Q. garryana* trees in fall treatment blocks at China Gulch and Hukill Hollow was documented. Trees with multiple stems from one crown were counted as one individual. Stems ranged from one to six per individual tree at China Gulch and one to four stems at Hukill Hollow. At China Gulch, 60% of mature *Q. garryana* trees were killed; at Hukill Hollow 88% were killed. Additionally, two large (55.9-91.4 cm/dbh) and six smaller (25.4-50.8 cm dbh) *Pinus ponderosa* trees were killed in the Hukill Hollow fall treatment block. As an indicator of fire severity, percentage of tree mortality correlated with the moderate to severe fire intensity of the fall prescribed fires at both China Gulch and Hukill Hollow (Tables 6 and 7).

Table 7. Tree mortality of mature *Quercus garryana* trees in fall treatment blocks.

Study Site	Post-Treatment # of Live Oak Trees	Post-Treatment # of Dead Oak Trees	Total # of Oak Trees
China Gulch	25	37	62
Hukill Hollow	2	15	17

Live and Dead Fuel Loads Before and After Spring and Fall Prescribed Fires

Initial fuel loads before fall and spring burn treatments were significantly higher at Hukill Hollow than at China Gulch resulting in a higher intensity spring and fall burns at Hukill Hollow. Despite lower fuel loads at China Gulch, the fall prescribed fire still yielded a moderate- to severe-intensity burn (Table 6). Following both spring and fall burns, the remaining fuel loads were similar at China Gulch and Hukill Hollow (Table 9) (Figure 6).

Litter depth was highest in Hukill Hollow spring burn plots compared to fall and spring burn plots at China Gulch and fall burn plots at Hukill Hollow—prior to prescribed fire treatments. After the prescribed burns, litter depth was similar across all treatments (Figure 7).

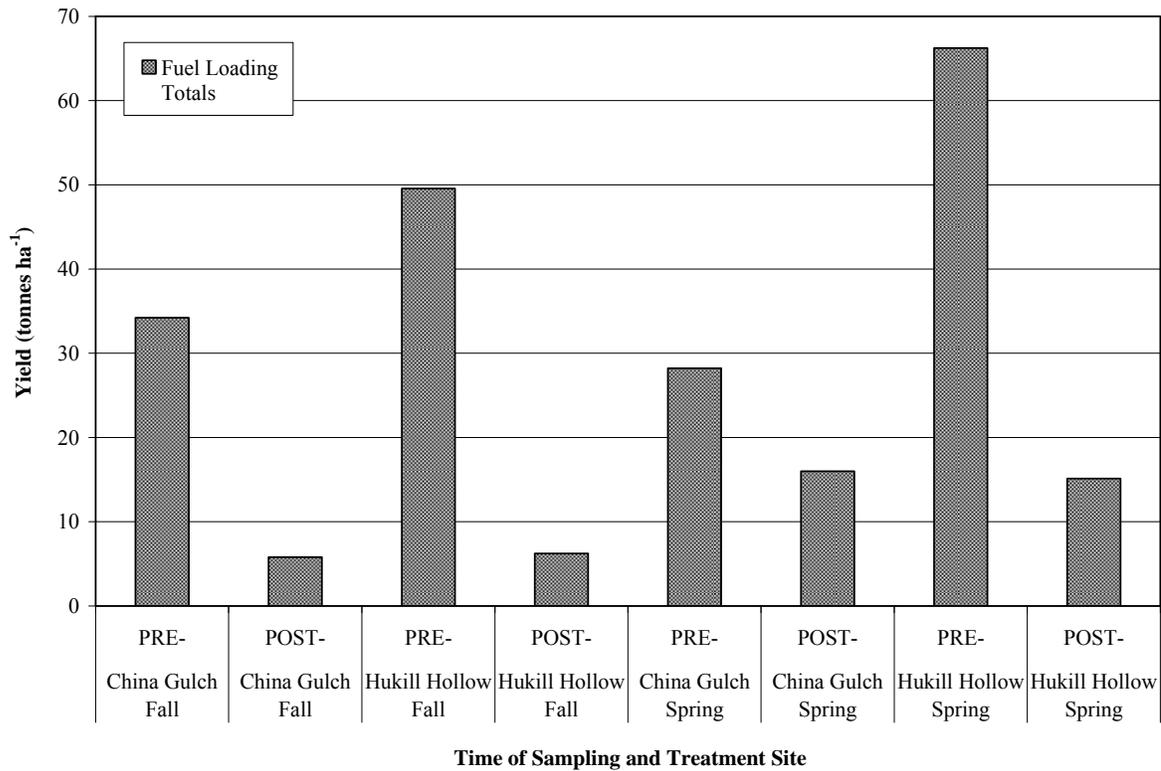


Figure 6. Fuel loading summary for pre-treatment and post-treatment spring and fall burned plots at China Gulch and Hukill Hollow.

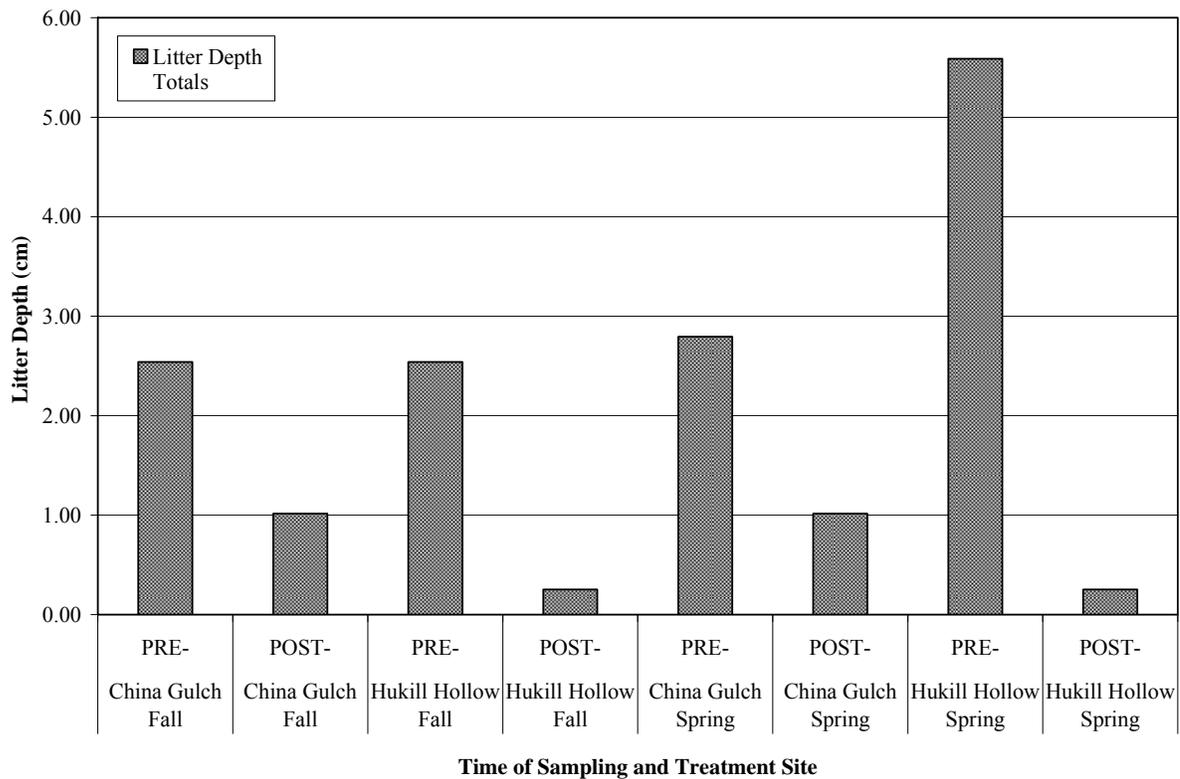


Figure 7. Litter depth totals for pre-treatment and post-treatment spring and fall burned plots at China Gulch and Hukill Hollow.

Average cover of live and dead woody species decreased in all treatment blocks following spring and fall burns at China Gulch and Hukill Hollow with the exception of live woody fuel following the spring burn at China Gulch. Transects in spring burn treatment blocks following fire at China Gulch showed a slight increase in live woody species from 6.4% to 11.7% (Table 8) (Figure 8).

Average cover of live herbaceous species did not significantly change following fall burn treatments at China Gulch and Hukill Hollow. Transects conducted after the spring burn treatment at China Gulch showed an increase in cover from 27.0% to 35.3%, while live herbaceous cover decreased at Hukill Hollow following spring prescribed fire. Dead herbaceous cover decreased following all prescribed fire treatments at China Gulch and Hukill Hollow (Figure 9).

After the spring prescribed fire at China Gulch, the average height of woody species increased from pre-treatment measurements. However, average height of woody and herbaceous species decreased following fall burn treatments at China Gulch and spring and fall burn treatments at Hukill Hollow (Figure 10).

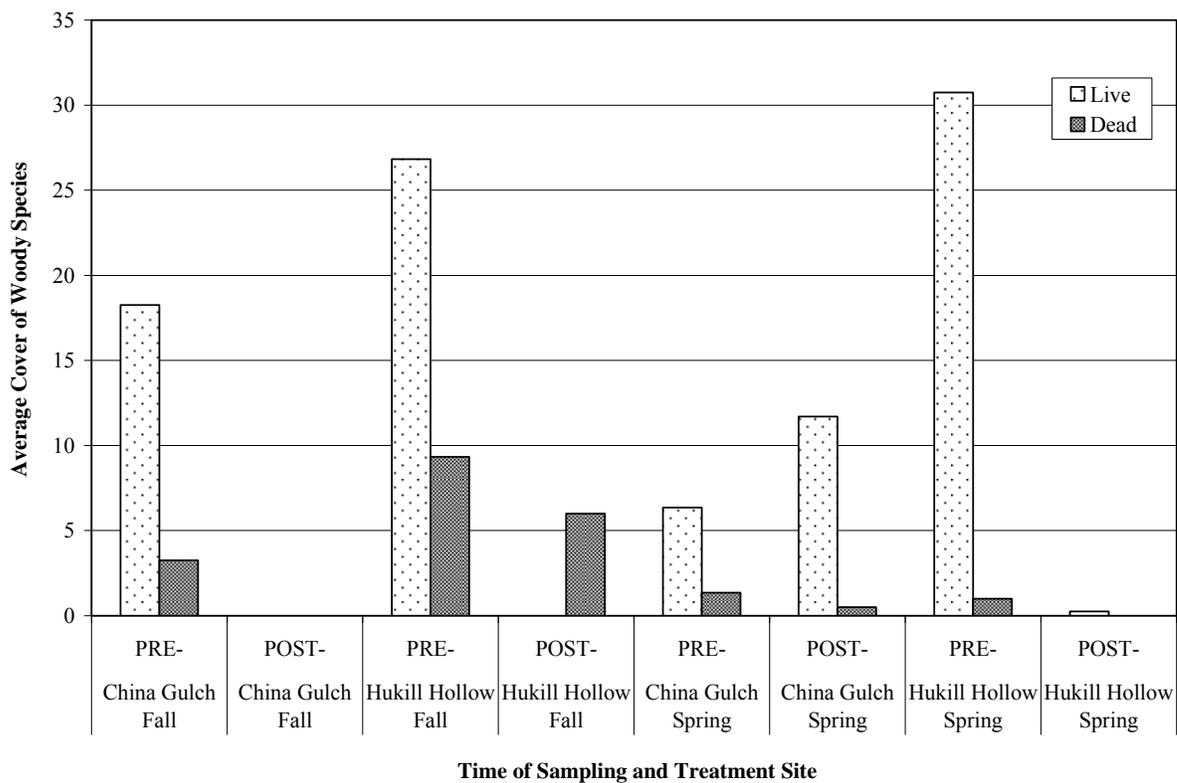


Figure 8. Average cover of woody species for pre-treatment and post-treatment spring and fall burned plots at China Gulch and Hukill Hollow.

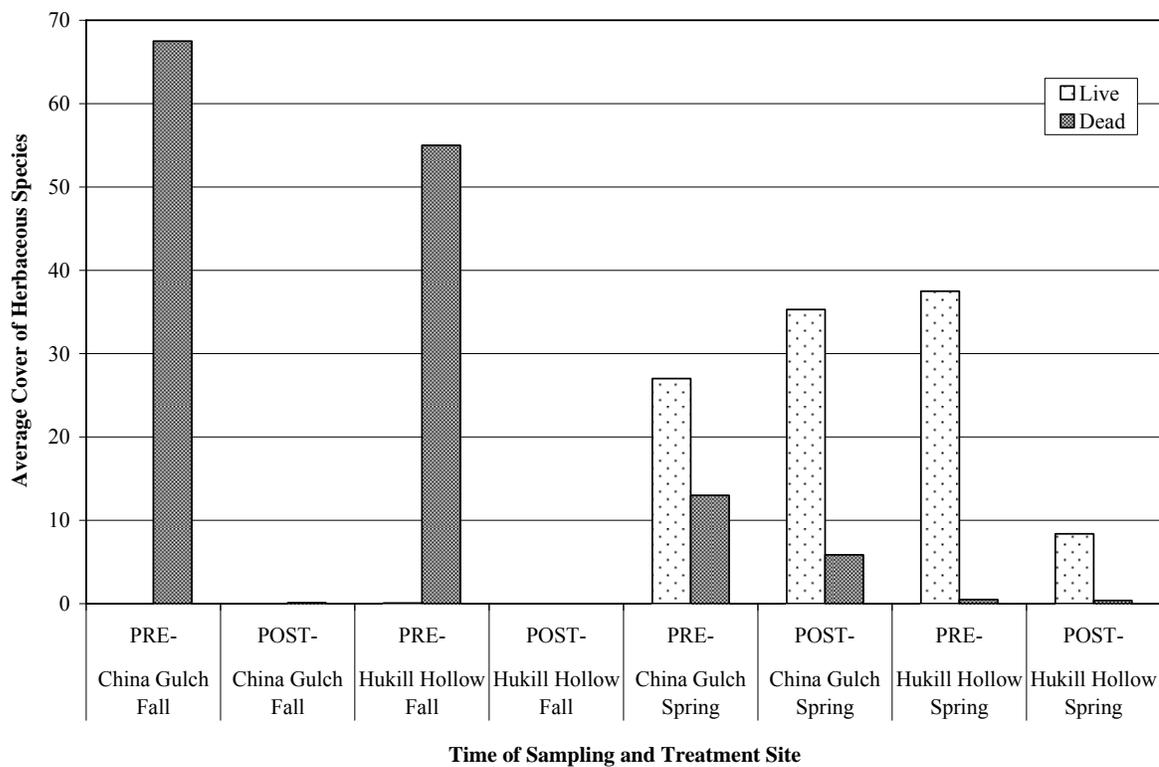


Figure 9. Average cover of herbaceous species for pre-treatment and post-treatment spring and fall burned plots at China Gulch and Hukill Hollow.

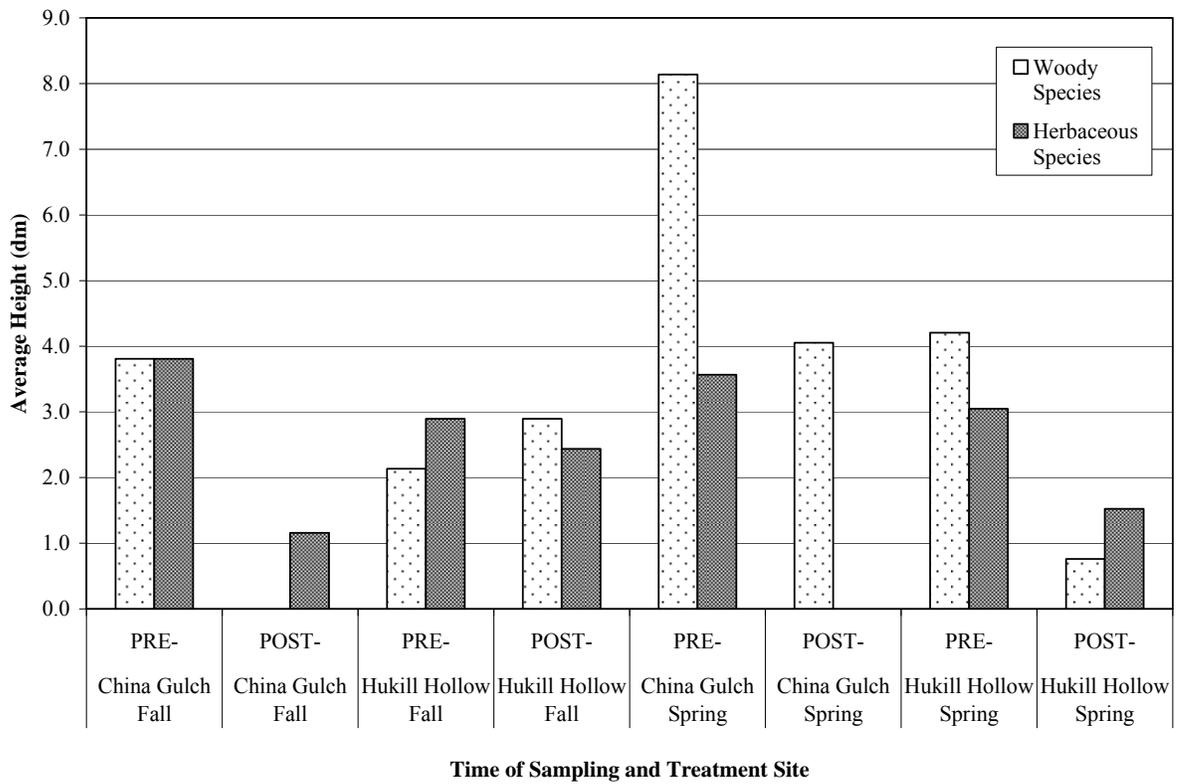


Figure 10. Average height of woody and herbaceous species for pre-treatment and post-treatment spring and fall burned plots at China Gulch and Hukill Hollow.

Table 8. Surface fuels vegetation summary.

Treatment and Site	Monitoring Status	Item	Average Cover (%)	Average Height (dm)	Average Biomass (tonnes ha ⁻¹)
China Gulch Fall Burn	PRE-TREATMENT	Dead Herbaceous	67.5	3.96	4.67
		Live Herbaceous	0	3.96	0.00
		Dead Shrub	3.3	3.96	0.49
		Live Shrub	18.3	3.96	2.87
	POST-TREATMENT	Dead Herbaceous	0.1	1.22	0.00
		Live Herbaceous	0	1.22	0.00
		Dead Shrub	0	0.00	0.00
		Live Shrub	0	0.00	0.00
China Gulch Spring Burn	PRE-TREATMENT	Dead Herbaceous	13	3.05	0.67
		Live Herbaceous	27	3.05	1.38
		Dead Shrub	1.4	2.13	0.13
		Live Shrub	6.4	2.13	0.60
	POST-TREATMENT	Dead Herbaceous	5.9	2.44	0.22
		Live Herbaceous	35.3	2.44	1.58
		Dead Shrub	0.5	3.05	0.02
		Live Shrub	11.7	3.05	1.13
Hukill Hollow Fall Burn	PRE-TREATMENT	Dead Herbaceous	55	3.66	3.56
		Live Herbaceous	0.1	3.66	0.00
		Dead Shrub	9.3	8.23	1.38
		Live Shrub	26.8	8.23	3.93
	POST-TREATMENT	Dead Herbaceous	0	0.00	0.00
		Live Herbaceous	0	0.00	0.00
		Dead Shrub	6	3.96	0.71
		Live Shrub	0	3.96	0.00
Hukill Hollow Spring Burn	PRE-TREATMENT	Dead Herbaceous	0.5	3.05	0.02
		Live Herbaceous	37.5	3.05	2.00
		Dead Shrub	1	4.27	0.09
		Live Shrub	30.8	4.27	3.09
	POST-TREATMENT	Dead Herbaceous	0.4	1.52	0.02
		Live Herbaceous	8.4	1.52	0.42
		Dead Shrub	0	0.91	0.00
		Live Shrub	0.3	0.91	0.00

Table 9. Surface fuels loading summary.

Treatment and Site	Monitoring Status: Pre- and Post-Treatment	1-hr	10 hr	100 hr	1-100 hr	1000-hr snd*	1000-hr rtn*	1-1000 hr	Duff	Litter	Total	Duff	Litter	Total
		Average tonnes ha ⁻¹									Average Depth (cm)			
China Gulch Fall Burn	PRE-	1.3	11.6	9.7	22.6			22.6	0.9	10.6	34.1	0.00	0.39	0.39
	POST-	0.1	0.0	1.1	1.2			1.2	0.0	4.7	5.9	0.00	0.16	0.16
China Gulch Spring Burn	PRE-	0.4	5.8	6.9	13.0			13.0	6.2	8.9	28.1	0.12	0.31	0.43
	POST-	0.8	2.5	8.2	11.4	0.4	0.0	11.8	0.0	4.2	16.0	0.00	0.16	0.16
Hukill Hollow Fall Burn	PRE-	0.6	5.6	20.2	26.4	11.3	1.1	38.8	0.0	10.8	49.6	0.00	0.39	0.39
	POST-	0.1	1.1	3.6	4.8	0.0	0.4	5.2	0.0	0.9	6.2	0.00	0.04	0.04
Hukill Hollow Spring Burn	PRE-	1.7	5.7	17.1	24.5	5.8	3.6	33.9	13.6	18.2	65.7	0.24	0.63	0.91
	POST-	0.0	0.7	9.2	9.9	4.4	0.0	14.4	0.0	0.7	15.0	0.00	0.04	0.04

* *snd* = sound; *rtn* = rotten

Soil Nutrient Response Before and After Spring and Fall Prescribed Fires at
China Gulch and Hukill Hollow

Soils samples were taken prior to, and approximately 48 hours after, prescribed fires. Pre-treatment and post-treatment results were analyzed using paired two-sample for means t-tests (P -value = 0.05).

Comparisons between China Gulch soil samples taken before and after spring and fall prescribed fires found no significant difference in percentage of carbon and levels of nitrogen ENR lbs/A in organic matter, potassium (ppm), sodium (ppm), and soil pH (Appendix F).

Soil samples from the fall burn at China Gulch showed higher levels of magnesium (ppm) (paired t-test: $P = 0.037$), calcium (ppm) (paired t-test: $P = 0.032$), and sulfur (ppm) (paired t-test: $P = 0.028$), but no change in levels of phosphorus (Weak Bray). By contrast, spring burn samples indicated higher levels of phosphorus (Weak Bray) (paired t-test: $P = 0.023$), with no change in levels of magnesium (ppm), calcium (ppm), and sulfur (ppm) (Appendix F).

Comparisons between Hukill Hollow soil samples taken before and after spring and fall prescribed fires resulted in no significant difference in percentage of carbon and levels of nitrogen ENR lbs/A in organic matter, phosphorus (Weak Bray), magnesium (ppm), calcium (ppm), sodium (ppm), and soil pH (Appendix F).

Soil samples from the fall burn at Hukill Hollow exhibited higher levels of potassium (ppm) (paired t-test: $P = 0.015$) and sulfur (ppm) (paired t-test: $P = 0.020$). No change was found in potassium (ppm) and sulfur (ppm) following the spring burn treatments at Hukill Hollow (Appendix F).

Soil sample comparisons between China Gulch and Hukill Hollow following spring and fall prescribed fires yielded the following results. Potassium (ppm) was significantly higher in China Gulch spring (paired t-test: $P = 0.040$) burn samples compared to Hukill Hollow spring burn samples. Magnesium (ppm) was significantly higher in Hukill Hollow spring (paired t-test: $P = 0.002$) and fall (paired t-test: $P = 0.046$) burn samples compared to China Gulch. Calcium (ppm) was significantly higher in Hukill Hollow spring (paired t-test: $P = 0.045$) burn samples compared to China Gulch spring burn samples (Appendix F).

Microclimate Comparisons of Mechanically Thinned and Unthinned

Oak-Chaparral Communities

Light intensity was noticeably higher in the thinned stand. On day 136 (JND) it is evident that it was sunny for most of the day, since the intensity of the light reached $2135.0 \mu\text{mol m}^{-2} \text{s}^{-1}$, and then slightly lower on day 138 (JND), another partly sunny day. Light intensity in the thinned stand ranged from $2135.0 - 0.0 \mu\text{mol m}^{-2} \text{s}^{-1}$ to $358.1 - 0.0 \mu\text{mol m}^{-2} \text{s}^{-1}$ in the unthinned stand (Figure 11).

Air temperature was similar on all days except on the sunniest day, 136 (JND), where the air temperature was higher in the thinned stand. Air temperature ranged from $24.8^{\circ}\text{C} - 4.7^{\circ}\text{C}$ in the thinned stand and $23.5^{\circ}\text{C} - 4.0^{\circ}\text{C}$ in the unthinned stand. However, on day 138 (JND), the partly sunny day, the air temperature was higher in the unthinned stand (Figure 12).

Relative humidity was similar on all days except for days 137 (JND) and 138 (JND), on which the humidity was higher in the unthinned stand. Relative humidity ranged from $99.2\% - 37.7\%$ in the thinned stand and $100.7\% - 42.8\%$ in the unthinned stand (Figure 13).

Leaf temperature was higher in the unthinned stand on days 136 (JND) and 138 (JND). Leaf temperature ranged from $22.9^{\circ}\text{C} - 3.1^{\circ}\text{C}$ in the thinned stand and $22.3^{\circ}\text{C} - 3.9^{\circ}\text{C}$ in the unthinned stand (Figure 14).

Soil temperature was noticeably higher in the thinned stand on all six days. Soil temperature ranged from $22.0^{\circ}\text{C} - 11.5^{\circ}\text{C}$ in the thinned stand and $17.4^{\circ}\text{C} - 10.0^{\circ}\text{C}$ in the unthinned stand (Figure 15).

Relative soil water content was higher in the unthinned stand on all six days. Relative water content ranged from $0.24\% - 0.19\%$ in the thinned stand and $0.37\% - 0.30\%$ in the unthinned stand (Figure 16).

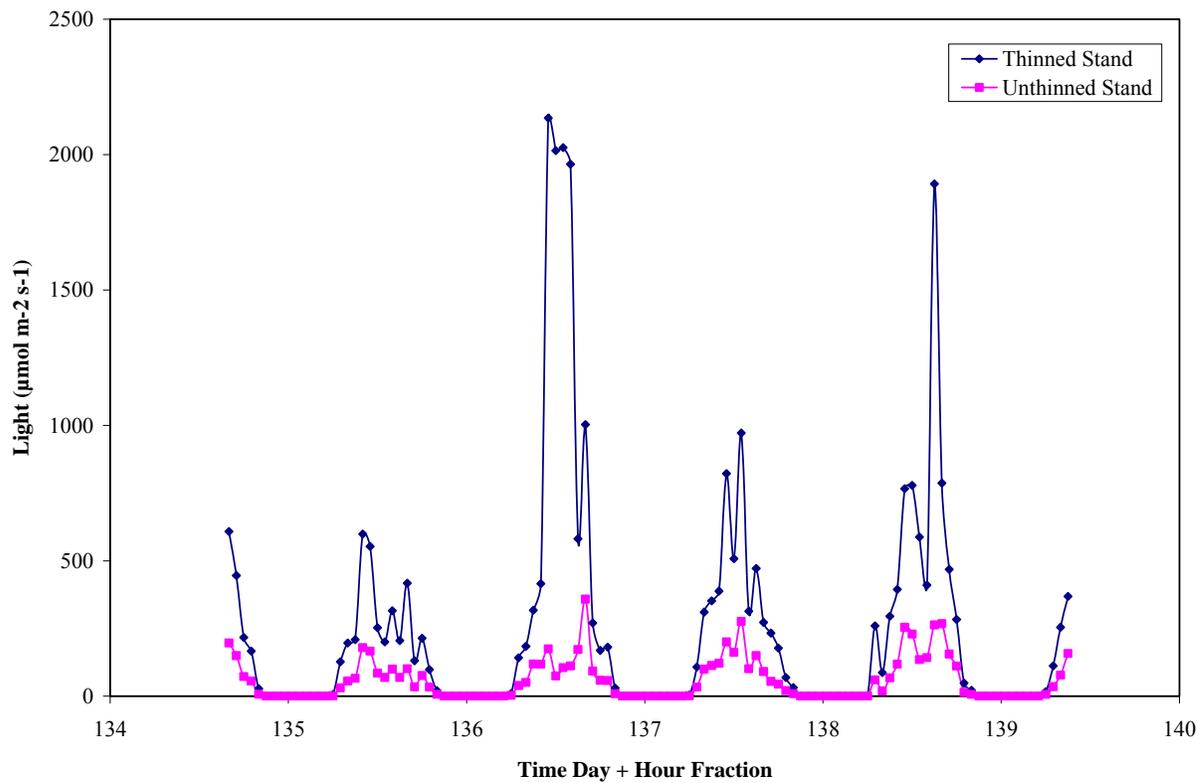


Figure 11. Light intensity in a thinned and unthinned oak-chaparral stand at Hukill Hollow.

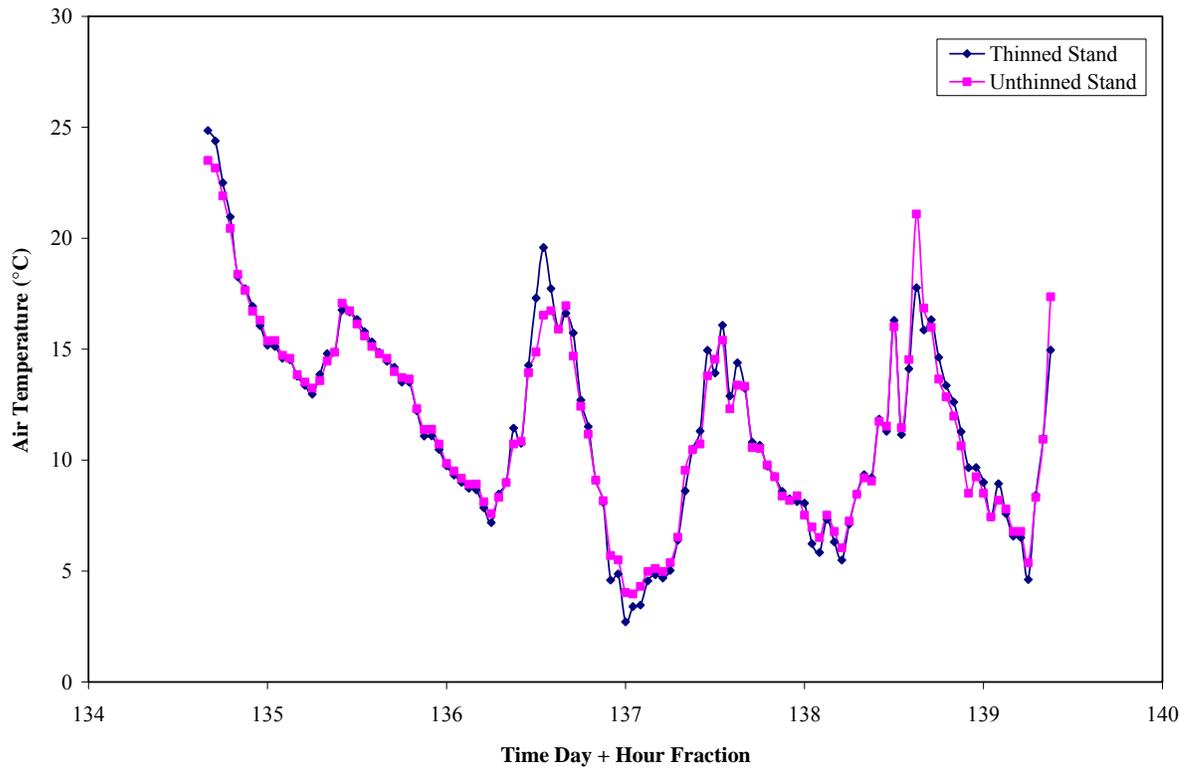


Figure 12. Air temperature in a thinned and unthinned oak-chaparral stand at Hukill Hollow.

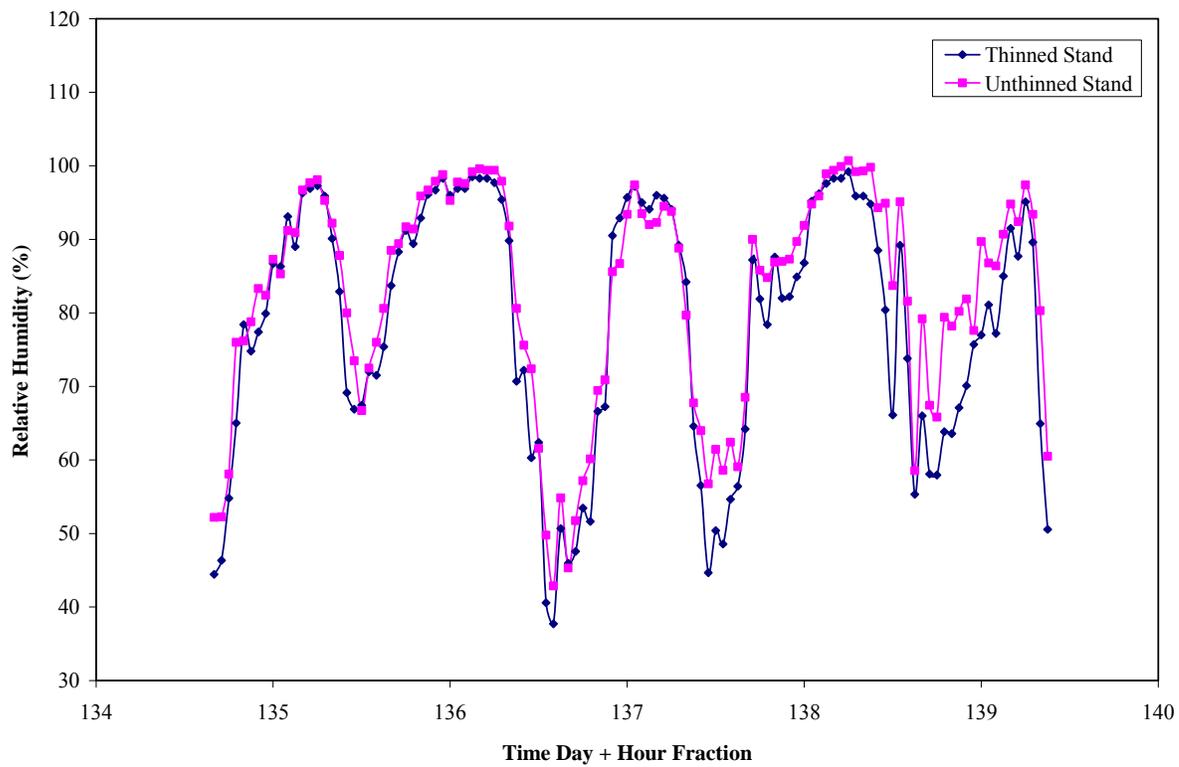


Figure 13. Relative humidity in a thinned and unthinned oak-chaparral stand at Hukill Hollow.

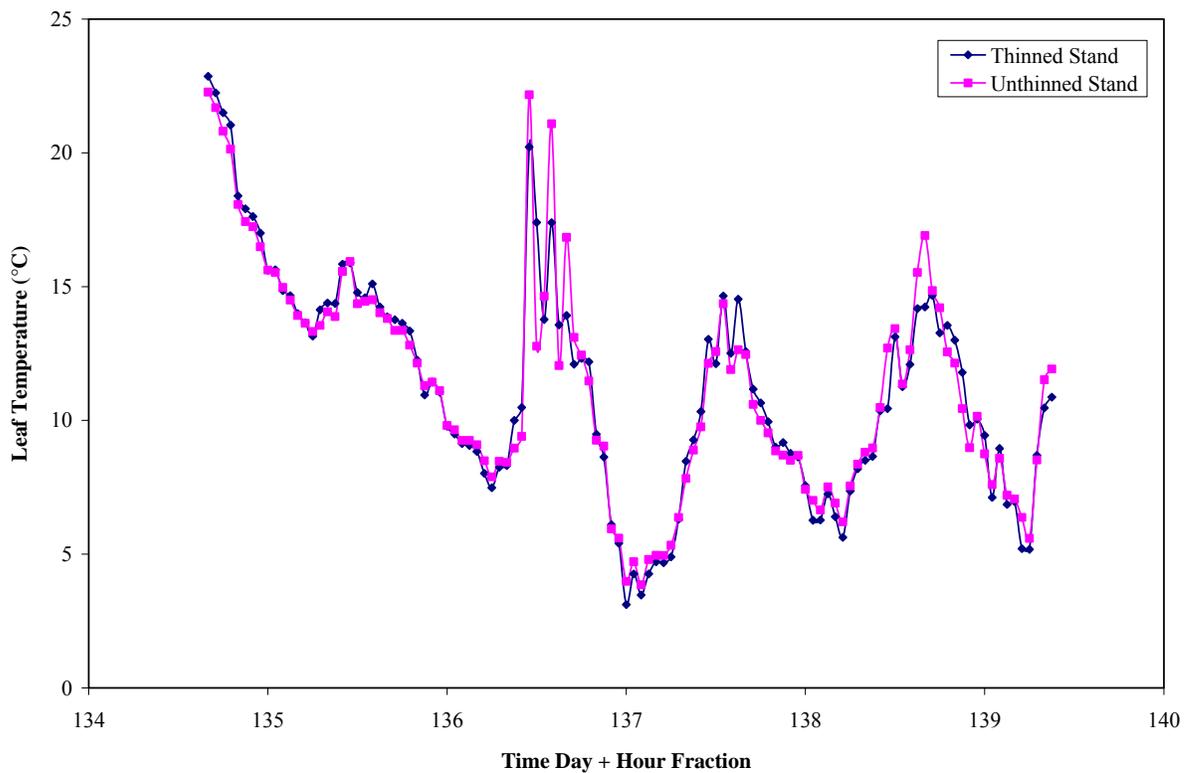


Figure 14. Leaf temperature of a *Quercus garryana* seedling in a thinned and unthinned oak-chaparral stand at Hukill Hollow.

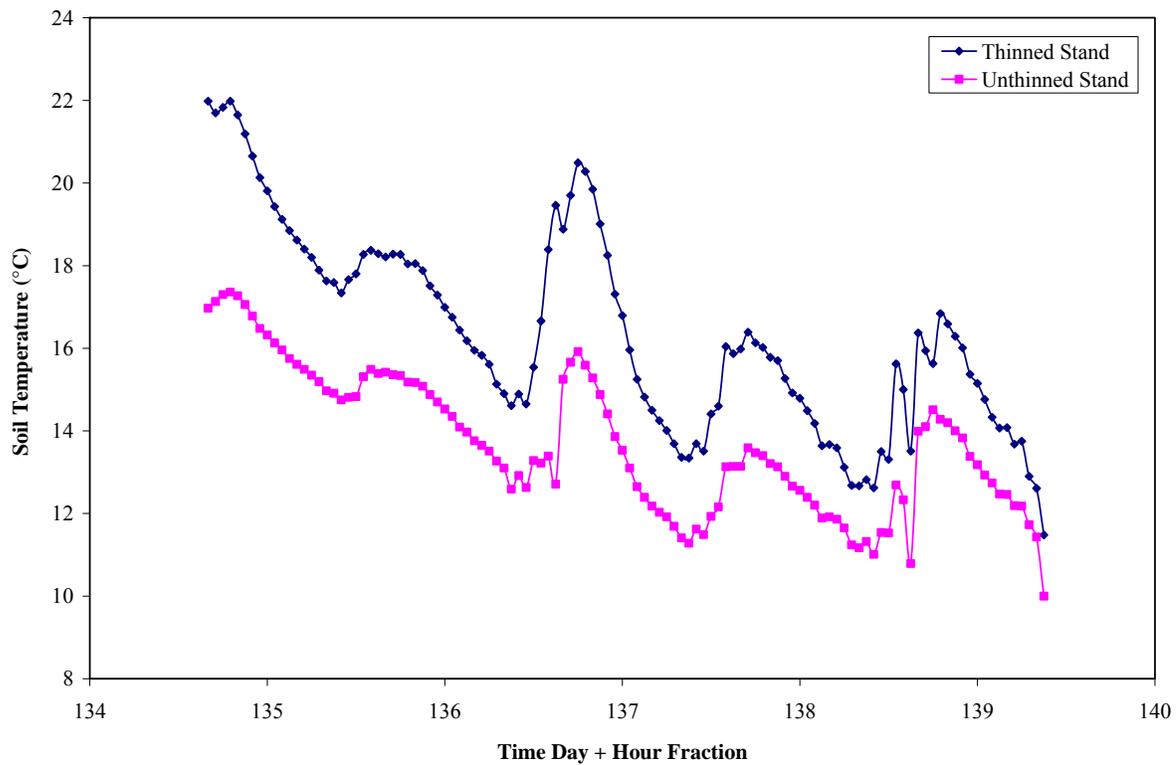


Figure 15. Soil temperature in a thinned and unthinned oak-chaparral stand at Hukill Hollow.

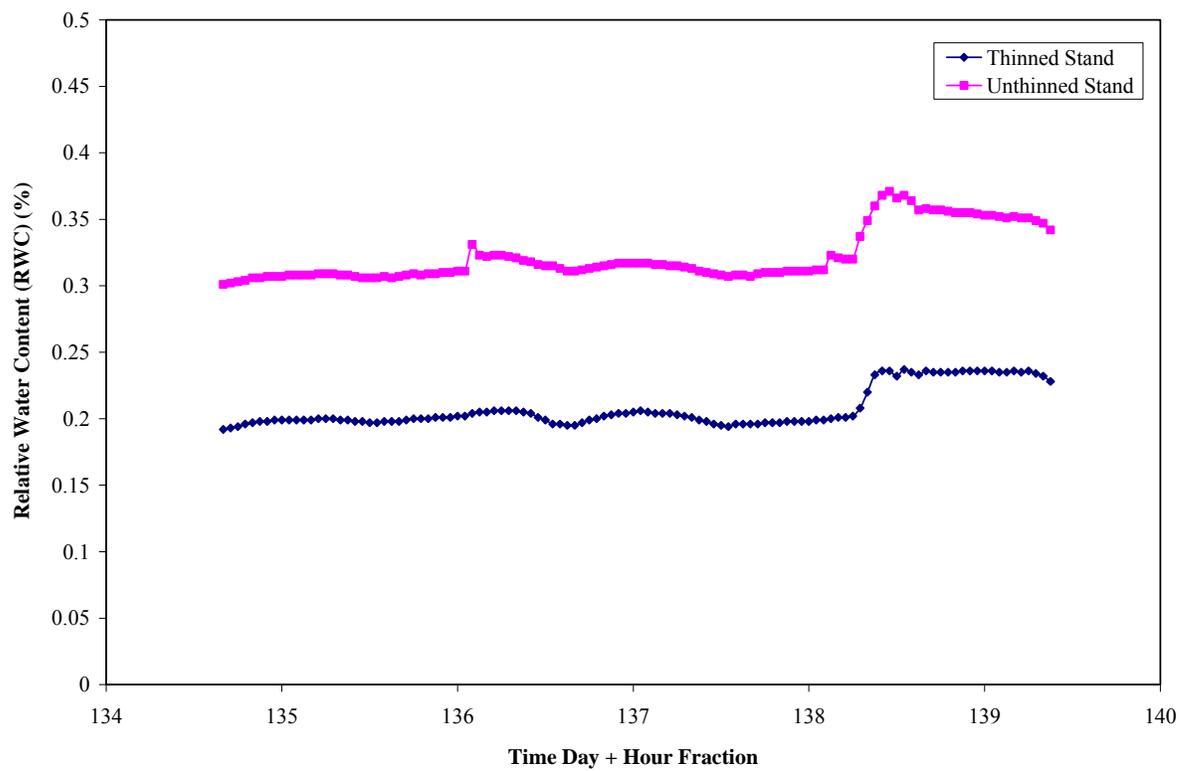


Figure 16. Relative water content in a thinned and unthinned oak-chaparral stand at Hukill Hollow.

Vegetation Response to Spring and Fall Prescribed Fires at
China Gulch and Hukill Hollow

Vegetation response to fire season and intensity was evaluated by applying increasingly finer grouping filters to elucidate patterns and trends in plant composition following prescribed fire and postfire seeding treatments. I began by exploring patterns of response in plant communities at both sites by comparing all recorded plant species grouped by pre-treatment and post-treatment year. Further comparisons were conducted by assembling all recorded individuals into life form groups. Next, just the most abundant and frequent species observed following fire were analyzed. Finally, within the most common plant species groups at both sites, added constraints were applied by grouping individuals into functional types based on how they regenerate following fire.

Grouped Comparisons of Plant Communities Before and After Spring and Fall
Prescribed Fires at China Gulch and Hukill Hollow

Grouped comparisons using the randomization (Monte Carlo) method were conducted on pre- (2005) and post- (2007) spring and fall prescribed fire treatments. Size of matrix varied according to groups evaluated. Sorensen (Bray-Curtis) was used as a distance measure and time of day was used as random number seed supplier for 1000 runs.

At China Gulch, there was no significant difference in treatment effect between pre-treatment 2005 spring burn (30 plots, 33 species) and post-treatment 2007 spring burn (30 plots, 45 species) ($r = 0.263273$, $P = 0.001$) (Appendix G). Similarity between plant abundance and diversity in spring burn treatment blocks from 2005 to 2007 was strong. Likewise, there was no significant difference in treatment effect between pre-treatment 2005 fall burn (30 plots, 33 species) and post-treatment 2007 fall burn (30 plots, 45 species) ($r = 0.126396$, $P = 0.047$) (Appendix G). In this case, there was a weak similarity in plant abundance and diversity in fall burn treatment blocks from 2005 to 2007, but still not enough to result in an effect of treatment.

At Hukill Hollow, there was a significant treatment effect between pre-treatment 2005 spring burn (30 plots, 38 species) compared to post-treatment 2007 spring burn (30 plots, 54 species), ($r = 0.132986$, $P = 0.096$) (Appendix G). Plant species abundance and diversity in spring burn treatment blocks from 2005 to 2007 were dissimilar. The same results occurred in fall burn treatment blocks, where there was a significant treatment effect between pre-treatment 2005 fall burn (30 plots, 38 species) compared to post-treatment 2007 fall burn (30 plots, 54 species), ($r = 0.097107$, $P = 0.143$) (Appendix G). Plant species abundance and diversity in post-treatment plots following the fall burn were markedly different from pre-treatment plots.

Grouped Comparisons of Life Forms Before and After Spring and Fall

Prescribed Fires at China Gulch and Hukill Hollow

Pre-treatment data (2005) and post-treatment data (2007) were compared using paired two-sample for means t-tests with a significance threshold of $P = 0.05$. Sums of cover estimates were used as the unit of measure. Individual plant species for each site were grouped by life form and then further categorized as native or exotic. For each test, five different pairwise comparisons were evaluated: (1) 2005 spring burn vs. 2007 spring burn; (2) 2005 fall burn vs. 2007 fall burn; (3) 2007 spring burn vs. 2007 spring control; (4) 2007 fall burn vs. 2007 fall control; and (5) 2007 spring burn vs. 2007 fall burn.

Table 10. Plant life form response to spring and fall prescribed fires at China Gulch and Hukill Hollow.

Response Variable	China Gulch Spring Burn	Hukill Hollow Spring Burn	China Gulch Fall Burn	Hukill Hollow Fall Burn
native perennial forbs	no change	increase	increase	increase
exotic perennial forbs	no change	no change	no change	no change
native annual forbs	increase	decrease	decrease	decrease
exotic annual forbs	increase	no change	increase	no change
native perennial grasses	no change	no change	increase	increase
exotic perennial grasses	increase	no change	no change	no change
native annual grasses	increase	decrease	decrease	no change
exotic annual grasses	no change	no change	increase	increase
native shrubs and trees	increase	no change	no change	increase

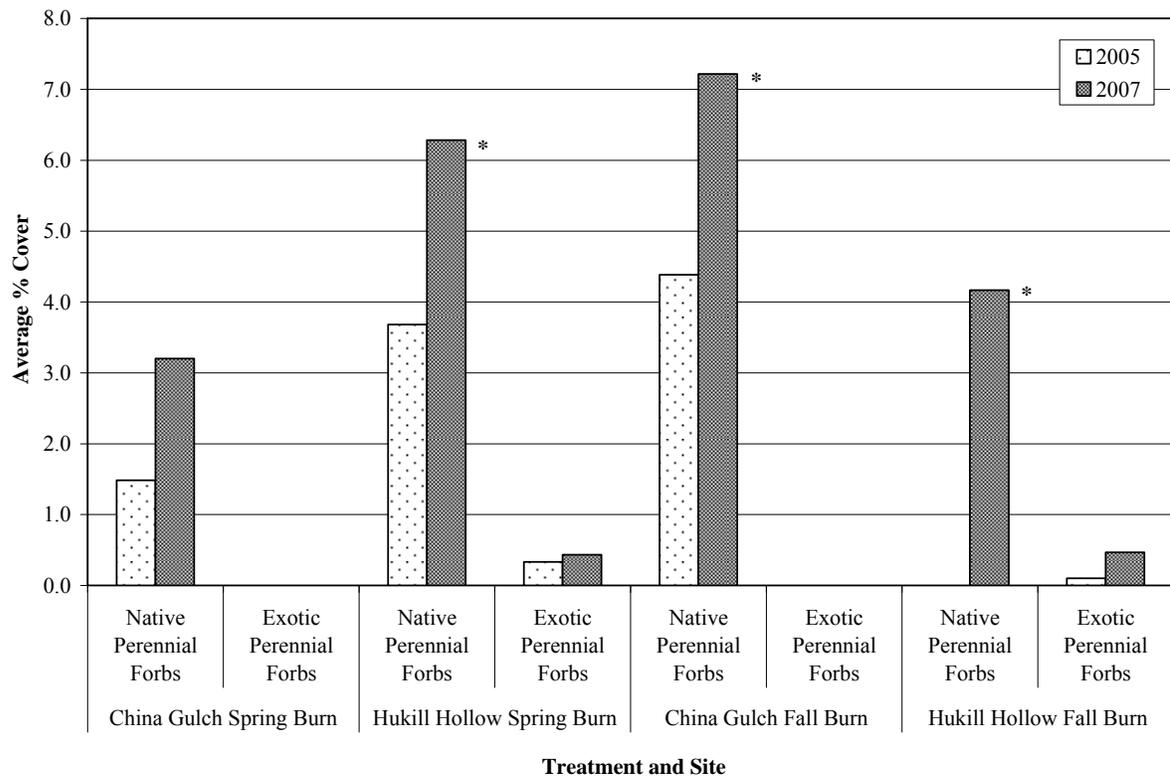


Figure 17. Average cover of native and exotic perennial forbs for China Gulch and Hukill Hollow spring and fall burn treatments (* $P \leq 0.05$; ** $P \leq 0.001$).

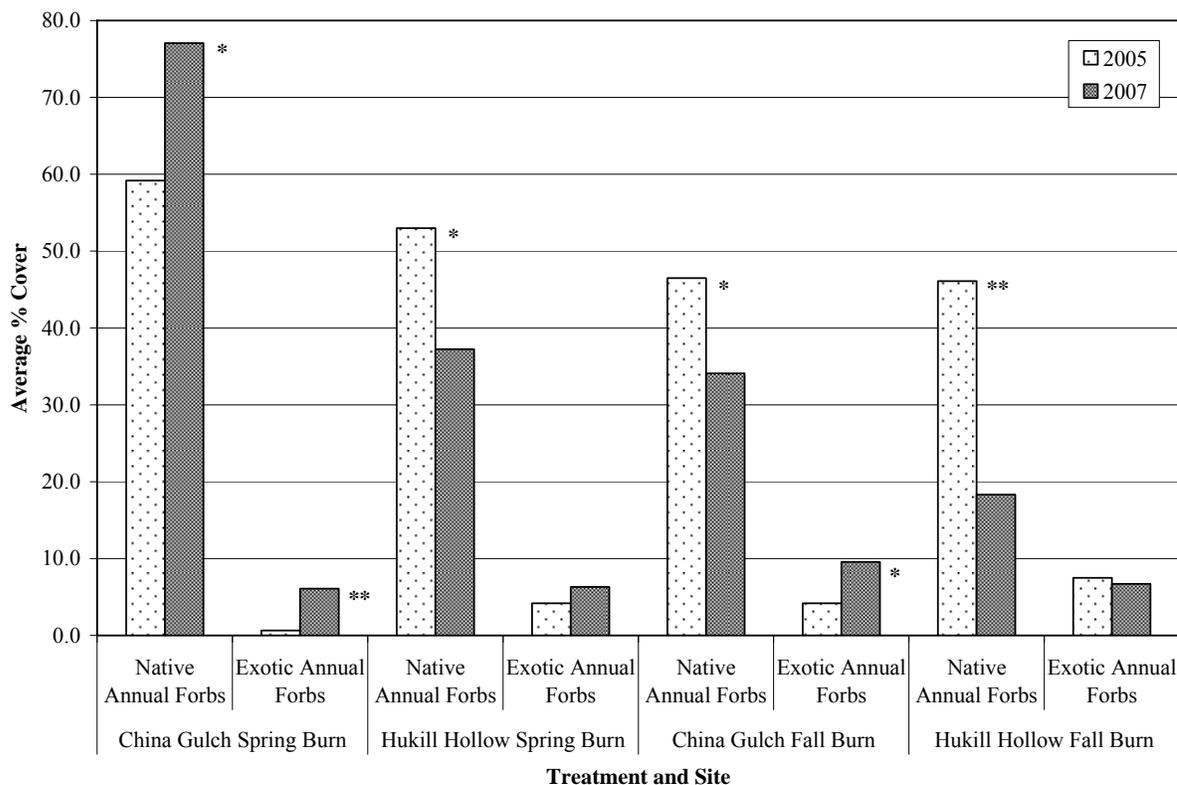


Figure 18. Average cover of native and exotic annual forbs for China Gulch and Hukill Hollow spring and fall burn treatments (* $P \leq 0.05$; ** $P \leq 0.001$).

There was no effect of treatment on native perennial forbs following spring burn treatment at China Gulch. Native perennial forbs increased following spring prescribed fire treatment at Hukill Hollow (paired t-test: $P = 0.029$). Native perennial forbs also increased following fall burn at China Gulch (paired t-test: $P = 0.013$) and Hukill Hollow (paired t-test: $P = 0.003$) (Appendix A) (Figure 17).

Exotic perennial forbs did not have a strong presence at China Gulch or Hukill Hollow (only one species, *Hypericum perforatum* L., was observed) prior to treatments.

As a result, neither site exhibited a noticeable change in the population mean of exotic perennial forbs following spring and fall burn treatments (Appendix A) (Figure 17).

Native annual forbs significantly increased in abundance following spring burn treatment at China Gulch (paired t-test: $P = 0.048$) and decreased following spring burn treatment at Hukill Hollow (paired t-test: $P = 0.016$). Native annual forbs decreased following fall burn treatments at both China Gulch (paired t-test: $P = 0.007$) and Hukill Hollow (paired t-test: $P = <0.001$) (Appendix A) (Figure 18).

Exotic annual forb abundance increased following spring burn (paired t-test: $P = <0.001$) and fall burn (paired t-test: $P = <0.038$) treatments at China Gulch. Hukill Hollow did not exhibit a significant change in exotic annual forbs following either spring or fall prescribed fire treatments (Appendix A) (Figure 18).

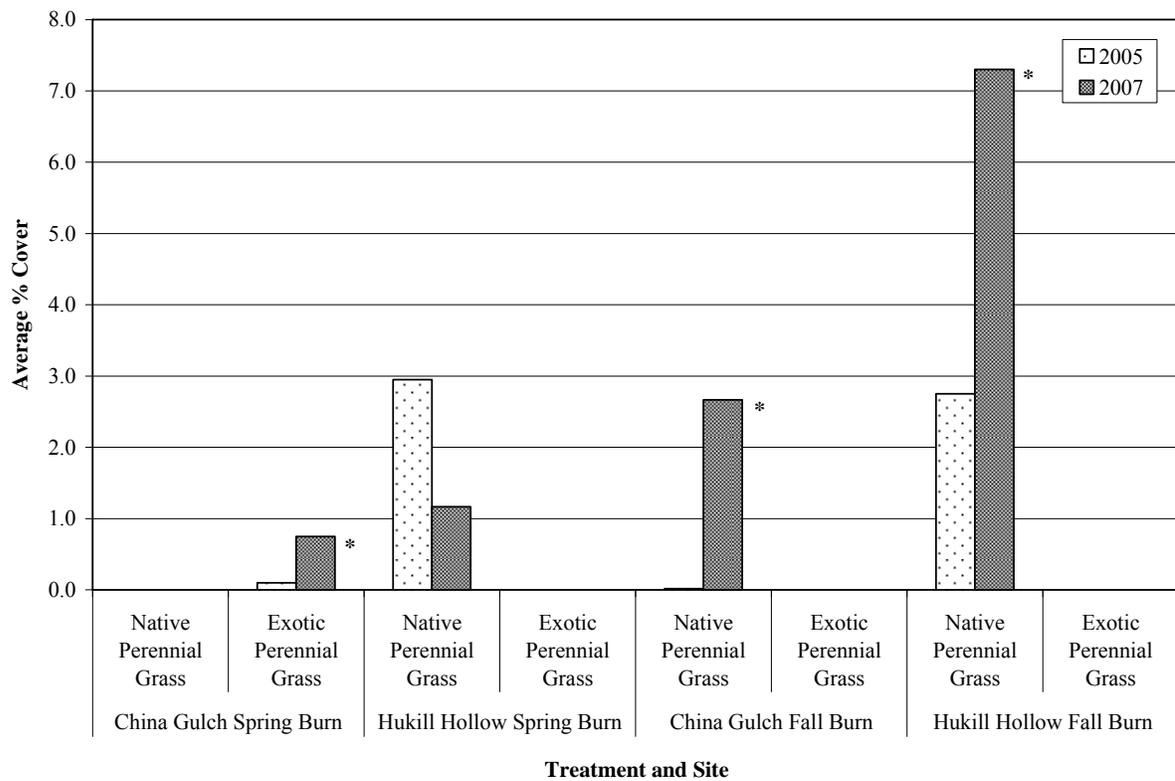


Figure 19. Average cover of native and exotic perennial grass for China Gulch and Hukill Hollow spring and fall burn treatments (* $P \leq 0.05$; ** $P \leq 0.001$).

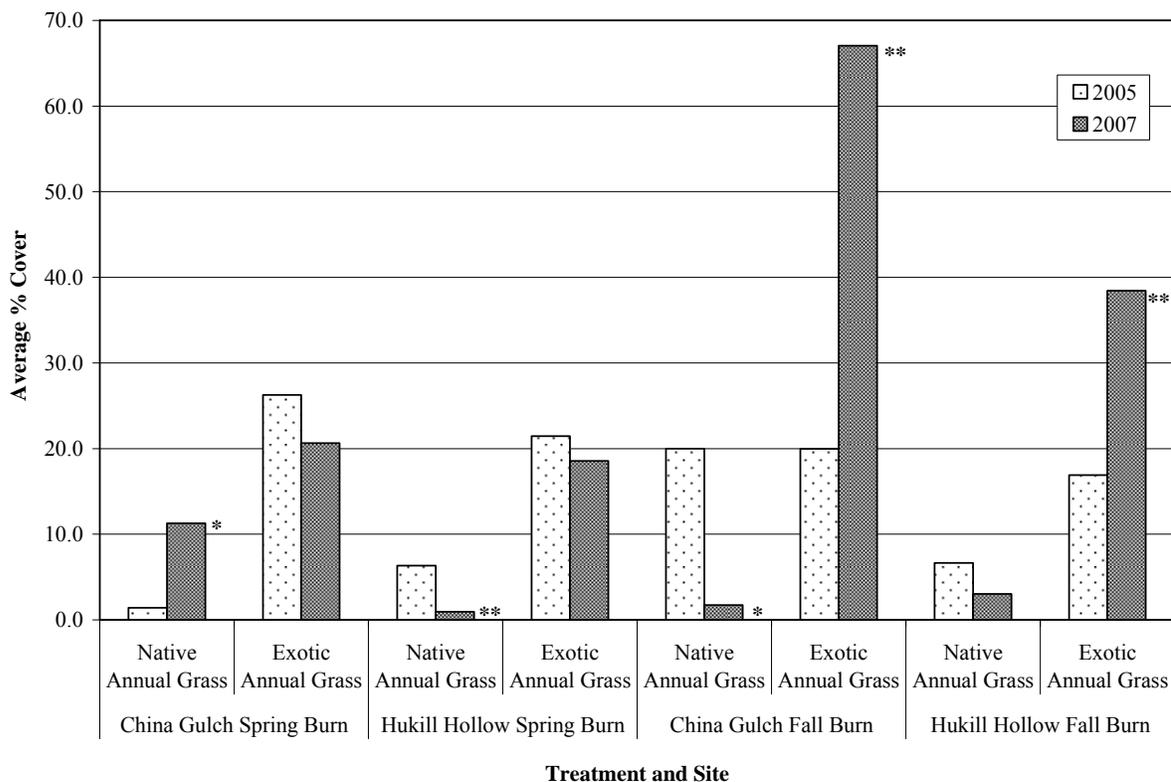


Figure 20. Average cover of native and exotic annual grass for China Gulch and Hukill Hollow spring and fall burn treatments (* $P \leq 0.05$; ** $P \leq 0.001$).

Abundance of native perennial grasses did not change significantly following spring burns at either site. Native perennial grass increased following fall burns at China Gulch (paired t-test: $P = 0.032$) and Hukill Hollow (paired t-test: $P = 0.034$). Data reflect the successful germination of native perennial bunchgrass species seeded as part of this study. Germination was not observed following spring prescribed fire but was successful following fall burn treatments at both sites (Appendix A) (Figure 19).

Exotic perennial grass increased following the spring burn at China Gulch (paired t-test: $P = 0.052$). By contrast, abundance of exotic perennial grasses did not significantly change following spring burn treatment at Hukill Hollow or following fall burn treatments at either site (Appendix A) (Figure 19).

Native annual grass increased following the spring burn at China Gulch (paired t-test: $P = 0.003$) but decreased following the spring burn at Hukill Hollow (paired t-test: $P = <0.001$). Populations of native annual grass decreased following fall burns at China Gulch (paired t-test: $P = <0.001$) but did not exhibit any detectable change at Hukill Hollow (Appendix A) (Figure 20).

Exotic annual grass increased significantly following fall burns at both China Gulch (paired t-test: $P = <0.001$) and Hukill Hollow (paired t-test: $P = 0.001$) without a significant change taking place following spring prescribed fires at either site (Appendix A) (Figure 20).

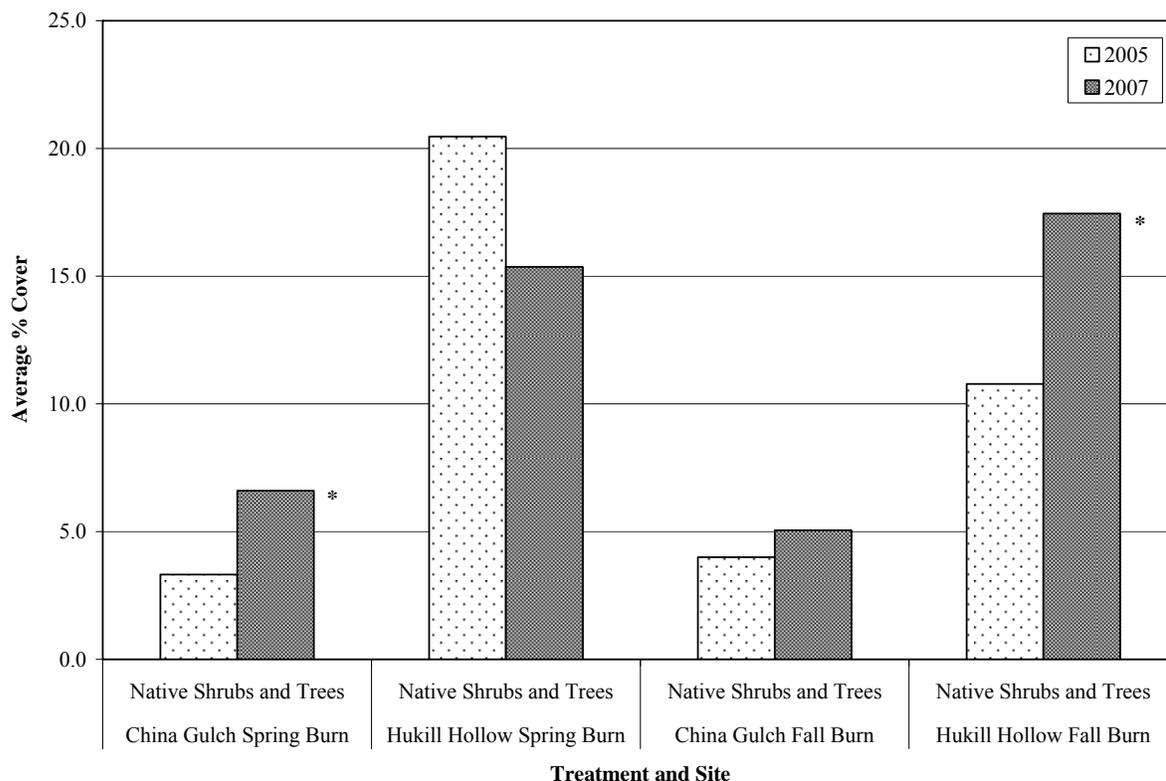


Figure 21. Average cover of native shrubs and trees for China Gulch and Hukill Hollow spring and fall burn treatments (* $P \leq 0.05$; ** $P \leq 0.001$).

Abundance of native shrubs and trees increased following the spring burn treatment at China Gulch (paired t-test: $P = 0.015$) and fall burn treatment at Hukill Hollow (paired t-test: $P = 0.030$) (Appendix A) (Figure 21). There was no significant change in abundance of native trees and shrubs following fall prescribed fire at China Gulch and spring prescribed fire at Hukill Hollow. Field observations conducted in May 2008 indicate that these results may not be reflective of the change in woody species

abundance over time. In all treatment blocks (both burned and unburned plots) seedlings of various woody shrubs were found in much greater abundance compared to 2007 data at both sites.

Grouped Comparisons of Native and Exotic Species Before and After Spring and Fall Prescribed Fires at China Gulch and Hukill Hollow

Abundance of native species increased significantly following the spring burn at China Gulch (paired t-test: $P = <0.001$). At Hukill Hollow, native species decreased significantly following spring burn (paired t-test: $P = <0.001$). The fall prescribed fire at both China Gulch (paired t-test: $P = <0.001$) and Hukill Hollow (paired t-test: $P = 0.003$) resulted in a significant decrease in native species (Appendix A) (Figure 22).

There was not a significant response of exotic species to spring burn treatments at either site. Exotic species abundance increased significantly following fall burns at both China Gulch (paired t-test: $P = <0.001$) and Hukill Hollow (paired t-test: $P = <0.001$) (Appendix A) (Figure 22).

When contrasting native versus exotic species abundance at both sites, I found that, prior to spring and fall burn treatments, native species abundance was significantly higher than exotic species abundance at both China Gulch (paired t-test: $P = <0.001$) and Hukill Hollow (paired t-test: $P = <0.001$). After spring prescribed fire treatments were applied, native species abundance was still significantly higher than exotic species abundance at both China Gulch (paired t-test: $P = <0.001$) and Hukill Hollow (paired

t-test: $P = <0.001$), while exotic species abundance was significantly higher than native species abundance following the fall burn treatment at China Gulch (paired t-test: $P = <0.001$). There was no significant difference in natives versus exotics at Hukill Hollow following fall prescribed burn (Appendix A) (Figure 22).

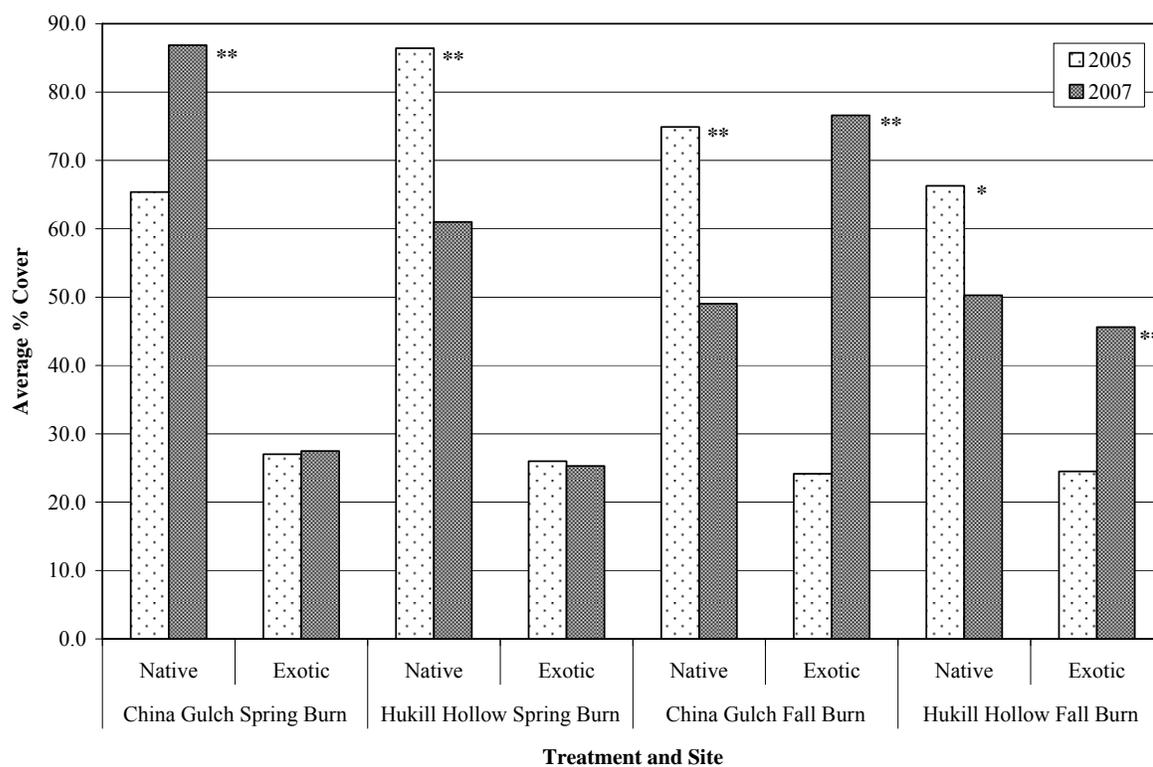


Figure 22. Average cover of native and exotic species for China Gulch and Hukill Hollow spring and fall burn treatments (* $P \leq 0.05$; ** $P \leq 0.001$).

Grouped Comparisons of Most Common Native and Exotic Species Before and After
Spring and Fall Prescribed Fires at China Gulch and Hukill Hollow

Plant species from each site were grouped by frequency and abundance into a smaller subset of the most common species for each site. Paired t-tests and Chi-square tests were used to identify significant changes in abundance and frequency following spring and fall prescribed fires at China Gulch and Hukill Hollow.

Most Common Native Species at China Gulch

Amsinckia menziesii (Lehm.) A. Nelson and J.F. Macbr. var. *intermedia* (Fisch. and C.A. Mey.) Ganders increased significantly in abundance and frequency following both the spring (paired t-test: $P = 0.005$) and fall (paired t-test: $P = <0.001$) prescribed fires (Appendices B and C).

Clarkia purpurea (W. Curtis) A. Nelson and J.F. Macbr. ssp. *quadrivulnera* (Douglas ex Lindl.) F.H. Lewis and M.E. Lewis decreased in abundance and frequency following spring (paired t-test: $P = <0.001$) and fall (paired t-test: $P = <0.003$) burn treatments (Appendices B and C).

Cryptantha flaccida (Douglas ex Lehm.) Greene increased significantly in abundance and frequency following the spring burn (paired t-test: $P = 0.005$). Fall burn treatments did not have an effect on the abundance or frequency of *Cryptantha flaccida* (Appendices B and C).

Daucus pusillus Michx. increased in abundance and frequency following the spring burn (paired t-test: $P = <0.001$). Response of *Daucus pusillus* to the fall prescribed fire was unclear, sometimes increasing and decreasing in abundance and frequency with the same set of variables when comparing paired t-test and Chi-square test results (Appendices B and C).

Spring burn treatments did not have a significant effect on *Gilia capitata* Sims abundance or frequency. By contrast, *Gilia capitata* increased significantly in abundance and frequency following the fall prescribed fire (paired t-test: $P = <0.001$) (Appendices B and C).

Changes in populations of *Lotus micranthus* Benth. to the spring and fall burn treatments were unclear. Depending on the test, *Lotus micranthus* increased and/or decreased in abundance and frequency with the same set of variables (Appendices B and C).

Madia sp. responded to the spring burn with a slight increase in abundance (paired t-test: $P = <0.024$) while decreasing significantly following the fall burn (paired t-test: $P = <0.001$). There was no detectable change in frequency of *Madia* sp. in response to both the spring and fall prescribed fire treatments (Appendices B and C).

Abundance of *Toxicodendron diversilobum* (Torr. and A. Gray) Greene increased significantly following the spring prescribed fire (paired t-test: $P = 0.020$). When fall burn plots were compared to fall control plots, an increase in abundance was found in the fall burn treatment (paired t-test: $P = 0.005$). Frequency did not change following spring and fall burn treatments (Appendices B and C).

Vulpia microstachys (Nutt.) Munro increased in abundance and frequency following the spring burn (paired t-test: $P = 0.003$) and decreased following the fall burn (paired t-test: $P = <0.001$) (Appendices B and C).

Most Common Exotic Species at China Gulch

Spring burn treatments did not have a significant effect on the abundance of *Bromus hordeaceus* but did produce an increase in frequency (Chi-square test: $P = < 0.024$). When fall burn plots were compared to fall control plots, an increase in abundance of *B. hordeaceus* was found following the fall prescribed fire (paired t-test: $P = < 0.001$). Frequency did not change following the fall burn treatments (Appendices B and C).

Bromus rubens L. increased significantly in abundance and frequency following the spring (paired t-test: $P = 0.004$) and fall (paired t-test: $P = 0.003$) prescribed fire treatments (Appendices B and C).

Bromus tectorum L. decreased in abundance following the spring burn (paired t-test: $P = <0.001$) but significantly increased in frequency (Chi-square test: $P = < 0.011$). Abundance of *Bromus tectorum* did not change following the fall prescribed fire, although frequency of this exotic annual grass significantly declined (Chi-square test: $P = < 0.001$) (Appendices B and C).

Erodium cicutarium (L.) L'Hér. ex Aiton increased in abundance and frequency in response to the spring (paired t-test: $P = 0.003$) and fall (paired t-test: $P = <0.001$) burn

treatments, with the greatest increase observed after the fall prescribed fire (Appendices B and C).

Torilis nodosa (L.) Gaertn. declined in abundance and frequency following the fall prescribed fire (paired t-test: $P = <0.026$). Response of *Torilis nodosa* to the spring prescribed fire was unclear. Abundance and frequency increased when pre-treatment and post-treatment burn plots were compared (paired t-test: $P = 0.001$), but decreased when burn plots and control plots were compared (paired t-test: $P = 0.036$) (Appendices B and C).

Spring control plots (paired t-test: $P = <0.009$) exhibited a higher abundance and frequency of *Vulpia myuros* (L.) C.C. Gmel. than the spring burn plots. *Vulpia myuros* increased significantly in abundance and frequency following the fall burn (paired t-test: $P = <0.001$) (Appendices B and C).

Most Common Native Species at Hukill Hollow

Clarkia purpurea ssp. *quadrivulnera* did not change in abundance or frequency following the spring and fall prescribed fire treatments (Appendices B and C).

Daucus pusillus did not significantly change in abundance or frequency following the spring burn treatment (Appendices B and C). *D. pusillus* decreased in abundance and frequency following the fall prescribed fire (paired t-test: $P = 0.016$).

Eriophyllum lanatum (Pursh) Forbes did not change in abundance or frequency following the spring and fall burn treatments (Appendices B and C).

Lotus micranthus declined significantly in abundance and frequency in response to the spring (paired t-test: $P = <0.001$) and fall (paired t-test: $P = <0.001$) prescribed fire treatments (Appendices B and C).

Response of *Madia* sp. to the spring burn treatment was unclear, sometimes increasing and decreasing in abundance and frequency with the same set of variables when comparing paired t-test and chi-square test results. *Madia* sp. declined significantly in abundance following the fall prescribed fire (paired t-test: $P = <0.001$) while frequency of *Madia* sp. increased (Chi-square test: $P = <0.001$) (Appendices B and C).

Toxicodendron diversilobum did not significantly change in abundance and frequency in response to the spring burn treatment. By contrast, *T. diversilobum* increased significantly in abundance and frequency following the fall prescribed fire (paired t-test: $P = <0.024$) (Appendices B and C).

Vulpia microstachys decreased significantly in abundance and frequency following the spring prescribed fire (paired t-test: $P = <0.001$). *V. microstachys* did not significantly change in abundance and frequency in response to the fall burn (Appendices B and C).

Most Common Exotic Species at Hukill Hollow

Aira caryophyllea L. did not significantly change in abundance and frequency following the spring prescribed fire. By contrast, *Aira caryophyllea* increased in abundance and frequency in response to the fall burn treatment (paired t-test: $P = 0.005$) when comparing fall burn plots to fall control plots (Appendices B and C).

Spring burn treatments did not have an effect on the abundance and frequency of *Bromus hordeaceus*. Following the fall prescribed fire, *B. hordeaceus* increased significantly in abundance (paired t-test: $P = 0.022$), although frequency did not change (Appendices B and C).

Bromus tectorum decreased in abundance (paired t-test: $P = 0.046$) and increased in frequency (Chi-square test: $P = 0.004$) following the spring burn treatment. Data showed a significant increase in abundance and frequency of *B. tectorum* following the fall prescribed fire (paired t-test: $P = <0.001$) (Appendices B and C).

Torilis nodosa abundance did not change following the spring and fall burn treatments. However, frequency of *T. nodosa* increased following the spring burn treatment (Chi-square test: $P = < 0.001$) and declined following the fall burn treatment (Chi-square test: $P = 0.005$) (Appendices B and C).

Grouped Comparisons of Functional Type Assemblages of Most Common Species
Following Spring and Fall Prescribed Fires at China Gulch and Hukill Hollow

Functional types are often used to classify postfire vegetation succession based on the way plants respond to fire cues. Classifications include postfire obligate seeder (non-resprouters specializing in postfire recruitment), facultative species (resprouters with persistent propagules), obligate resprouter (resprouters with non-persistent propagules specializing in recruitment between fires), and non-persistent species (non-persistent following fire) (Pausas et al. 2004) (Tables 11 and 12).

Table 11. Fire response of the most abundant and frequent species (Pausas et al. 2004).

Most Common Species by Abundance and Frequency for China Gulch ▪ and Hukill Hollow ◦	Species Response to Fire				Pre-Treatment Plots (2005) Present in	Post-Treatment Burned Plots (2007) Present in
	(R+ P+) Resprouters with persistent propagules	(R-P+) Non-resprouters specialize in postfire recruitment	(R+P-) Resprouters with non-persistent propagules, recruitment between fires	(R-P-) Species non-persistent following fire		
	Facultative Species	Obligate Seeder	Obligate Resprouter	Non-Persistent Species		
<i>Aira caryophylla</i> ◦		x			x	x
<i>Amsinckia menziesii</i> var. <i>intermedia</i> ▪		x			x	x
<i>Bromus hordeaceus</i> ▪◦		x			x	x
<i>Bromus rubens</i> ▪		x			x	x
<i>Bromus tectorum</i> ▪◦		x			x	x
<i>Clarkia purpurea</i> ▪◦		x			x	x
<i>Cryptantha flaccida</i> ▪		x			x	x
<i>Daucus pusillus</i> ▪◦		x			x	x

<i>Eriophyllum lanatum</i> °	X		X	X
<i>Erodium cicutarium</i> ▪		X	X	X
<i>Gilia capitata</i> ▪		X	X	X
<i>Lotus micranthus</i> ▪°		X	X	X*
<i>Madia</i> sp.▪°		X	X	X
<i>Toxicodendron diversilobum</i> ▪°	X		X	X
<i>Torilis nodosa</i> ▪°		X	X	X
<i>Vulpia microstachys</i> ▪°	X		X	X
<i>Vulpia myuros</i> ▪		X		X

* *L. micranthus* was not found in China Gulch fall burn plots

Table 12. Most abundant woody seedlings observed in 2008 (Pausas et al. 2004).

Seedlings Observed in 2008 Spring and Fall Treatment Blocks China Gulch and Hukill Hollow	2008 Woody Species Response in Burned and Control (unburned) Plots				Spring Burned Plots Present	Fall Burned Plots Present	Control (Unburned) Plots Present
	(R+ P+) Resprouters with persistent propagules	(R-P+) Non-resprouters specialize in postfire recruitment	(R+P-) Resprouters with non- persistent propagules, recruitment between fires	(R-P-) Species non- persistent following fire			
	Facultative Species	Obligate Seeder	Obligate Resprouter	Non-Persistent Species			
<i>Arctostaphylos viscida</i>		x			cg - hh - x	cg - hh - x	cg - hh - x
<i>Cercocarpus betuloides</i>		x (weak sprouter)			cg - hh -	cg - hh - x	cg - hh - x
<i>Ceanothus cuneatus</i>		x			cg - hh -	cg - hh - x	cg - x hh - x
<i>Eriodictyon californicum</i>	x				cg - hh - x	cg - x hh - x	cg - hh - x
<i>Garrya fremontii</i>			x		cg - hh -	cg - hh -	cg - x hh - x

<i>Lonicera hispidula</i>		x	cg - x hh -	cg - x hh - x	cg - x hh - x
<i>Prunus subcordata</i>	x		cg - hh - x	cg - hh - x	cg - hh - x
<i>Quercus garryana</i>	x		cg - hh -	cg - hh - x	cg - x hh - x
<i>Toxicodendron diversilobum</i>	x		cg - x hh - x	cg - x hh - x	cg - x hh - x

cg = China Gulch; *hh* = Hukill Hollow

Postfire Seeding Success Following Spring and Fall Prescribed Fires at
China Gulch and Hukill Hollow

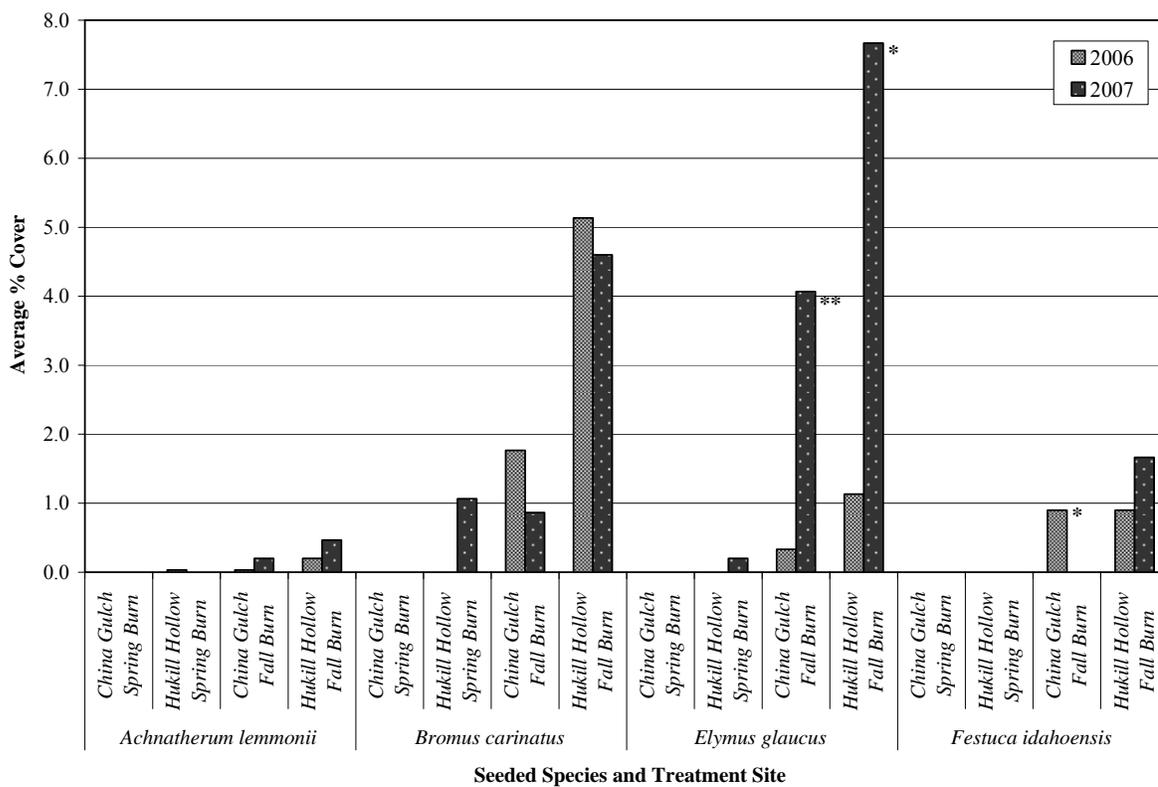


Figure 23. Average cover of seeded bunchgrass species for China Gulch and Hukill Hollow spring and fall burn treatments (* $P \leq 0.05$; ** $P \leq 0.001$).

Comparisons of Germination Success between Spring and Fall Prescribed Fires at
China Gulch and Hukill Hollow

Successful germination of seeded bunchgrass species at China Gulch and Hukill Hollow were evaluated with ANOVA tests (Two-Factor with Replication (P -value = 0.1)) using data from 2007. All results yielded a significant difference since germination did not take place following the spring prescribed fires at either China Gulch or Hukill Hollow (Appendix E) (Figure 23).

A comparison of total cover of germinants following the fall prescribed fires between China Gulch and Hukill Hollow yielded a significant difference (ANOVA: $P = 0.015$) with the highest mean cover of germinants found in the Hukill Hollow fall burn treatment (Appendix E) (Figure 23).

At China Gulch, germination success for individual species resulted in no significant difference in the total cover of *Achnatherum lemmonii* (ANOVA: $P = 0.323$) and *Festuca idahoensis* ssp. *roemeri* (ANOVA: $P =$ test invalid) between the spring and fall burns. *F. idahoensis* ssp. *roemeri* was not observed in 2007 plots (Appendix E). By contrast, there was a significant difference in the total cover of *Bromus carinatus* (ANOVA: $P = 0.007$) and *Elymus glaucus* (ANOVA: $P = 0.001$) between the spring and fall burns, with a higher mean cover recorded in the fall burn plots at China Gulch (Appendix E) (Figure 23).

At Hukill Hollow, total cover of *Achnatherum lemmonii* (ANOVA: $P = 0.048$), *Elymus glaucus* (ANOVA: $P = 0.001$), and *Festuca idahoensis* ssp. *roemeri* (ANOVA:

$P = 0.002$) was significantly higher in the fall burn plots compared to the spring burn plots. There was not a detectable difference in the total cover of *Bromus carinatus* between the spring and fall burn plots at Hukill Hollow (Appendix E) (Figure 23).

Comparisons of Germination Success between Burned and Unburned Control Plots at China Gulch and Hukill Hollow

At China Gulch, comparisons of the 2007 post-treatment burned plots versus the 2007 control (unburned) plots yielded mixed results. I did not find a significant difference in the mean cover of *Achnatherum lemmonii* and *Festuca idahoensis* ssp. *roemerii* between the burned and control plots. *F. idahoensis* ssp. *roemerii* was not observed in 2007 plots. There was a significant difference in the mean cover of *Bromus carinatus* (ANOVA: $P = 0.007$) and *Elymus glaucus* (ANOVA: $P = 0.001$) between the burned and control plots, with a higher mean cover in the burned plots (Appendix E).

At Hukill Hollow, comparisons of the 2007 post-treatment burned plots versus the 2007 control plots yielded a significantly higher mean cover of *Achnatherum lemmonii* (ANOVA: $P = 0.048$), *Elymus glaucus* (ANOVA: $P = <0.001$), and *Festuca idahoensis* ssp. *roemerii* (ANOVA: $P = 0.041$) in the burned plots. I did not find a noticeable difference in the mean cover of *Bromus carinatus* between the burned plots and the control plots (Appendix E).

Survival of Seeded Bunchgrasses Three Years Following Prescribed Fires

Cover Comparisons of Germinants 2006 to 2007

At China Gulch, there was no significant difference in the mean cover of *Achnatherum lemmonii* and *Bromus carinatus* between the 2006 and 2007 seeded fall burned plots. By contrast, there was a significant increase in the total cover of *Elymus glaucus* (ANOVA: $P = 0.013$) in 2007 when compared to 2006 in seeded fall burned plots. *Festuca idahoensis* ssp. *roemeri* (ANOVA: $P = <0.001$) was not observed in the 2007 plots, yielding a significant change in abundance from 2006 to 2007 (Appendix E) (Table 13).

At Hukill Hollow, I did not find a noticeable difference in the total cover of *Achnatherum lemmonii*, *Bromus carinatus*, and *Festuca idahoensis* ssp. *roemeri* between the 2006 and 2007 seeded fall burned plots. There was a significant change in the total cover of *Elymus glaucus* between the 2006 and 2007 seeded fall burned plots (ANOVA: $P = 0.008$), with a higher mean cover in 2007 plots (Appendix E) (Table 13).

Table 13. Seeded bunchgrass germination response to spring and fall prescribed fires at China Gulch and Hukill Hollow from 2006 to 2007 based on cover estimates.

Bunchgrass Species	China Gulch Spring Burn 2006-2007	Hukill Hollow Spring Burn 2006-2007	China Gulch Fall Burn 2006-2007	Hukill Hollow Fall Burn 2006-2007
<i>Achnatherum lemmonii</i>	no change	no change	no change	no change
<i>Bromus carinatus</i>	no change	no change	no change	no change
<i>Elymus glaucus</i>	no change	no change	increase	increase
<i>Festuca idahoensis</i> ssp. <i>roemeri</i>	no change	no change	decrease	no change

Density Comparisons of Germinants 2007 to 2008

At China Gulch, density counts of *Achnatherum lemmonii* and *Festuca idahoensis* ssp. *roemeri* increased from 2007 to 2008. *F. idahoensis* ssp. *roemeri* was not observed in 2007 plots but was observed in 2008. *Bromus carinatus* and *Elymus glaucus* decreased from 2007 to 2008. *E. glaucus* was observed to be heavily browsed in May 2008, possibly preventing seed set. Density comparisons between years yielded no significant change in the number of germinants from 2007 to 2008 (paired t-test: $P = <0.534$) (Appendix D) (Tables 14 and 15) (Figure 24).

At Hukill Hollow, density counts of *Achnatherum lemmonii*, *Bromus carinatus*, and *Festuca idahoensis* ssp. *roemerii* increased from 2007 to 2008. *Elymus glaucus* decreased from 2007 to 2008. Similar to China Gulch, *E. glaucus* was observed to be heavily browsed in May 2008. Density comparisons between years yielded no significant change in the number of germinants from 2007 to 2008 (paired t-test: $P = <0.216$) (Appendix D) (Tables 14 and 15) (Figure 24).

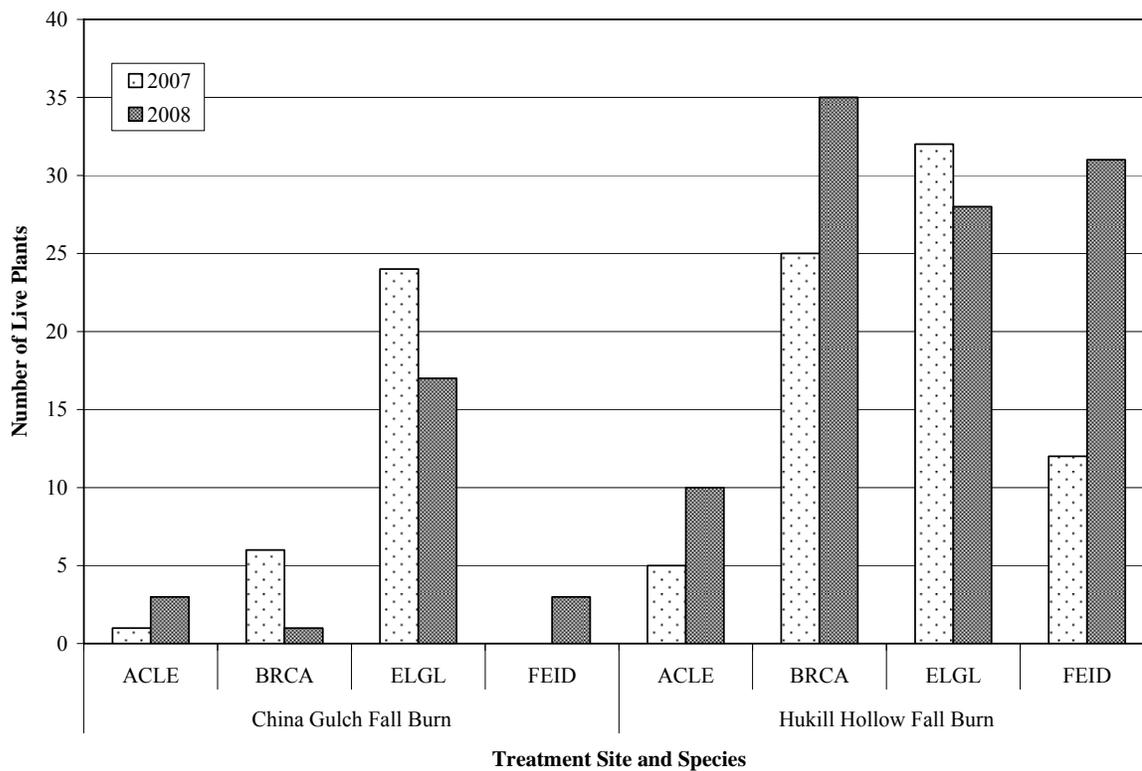
Table 14. Density counts of individual live germinants in 2007 and 2008.

Treatment and Site	Species	Average number seeds sowed per plot*	2007	2008
China Gulch Fall Burn Plots	<i>Achnatherum lemmonii</i>	46	1	3
	<i>Bromus carinatus</i>	41	6	1
	<i>Elymus glaucus</i>	54	24	17
	<i>Festuca idahoensis</i> ssp. <i>roemerii</i>	360	0	3
Hukill Hollow Fall Burn Plots	<i>Achnatherum lemmonii</i>	46	5	10
	<i>Bromus carinatus</i>	41	25	35
	<i>Elymus glaucus</i>	54	32	28
	<i>Festuca idahoensis</i> ssp. <i>roemerii</i>	360	12	31

* Number of seeds for each species is based on a total weight of 0.28 g

Table 15. Seeded bunchgrass germination response to spring and fall prescribed fires at China Gulch and Hukill Hollow from 2007 to 2008 based on density counts.

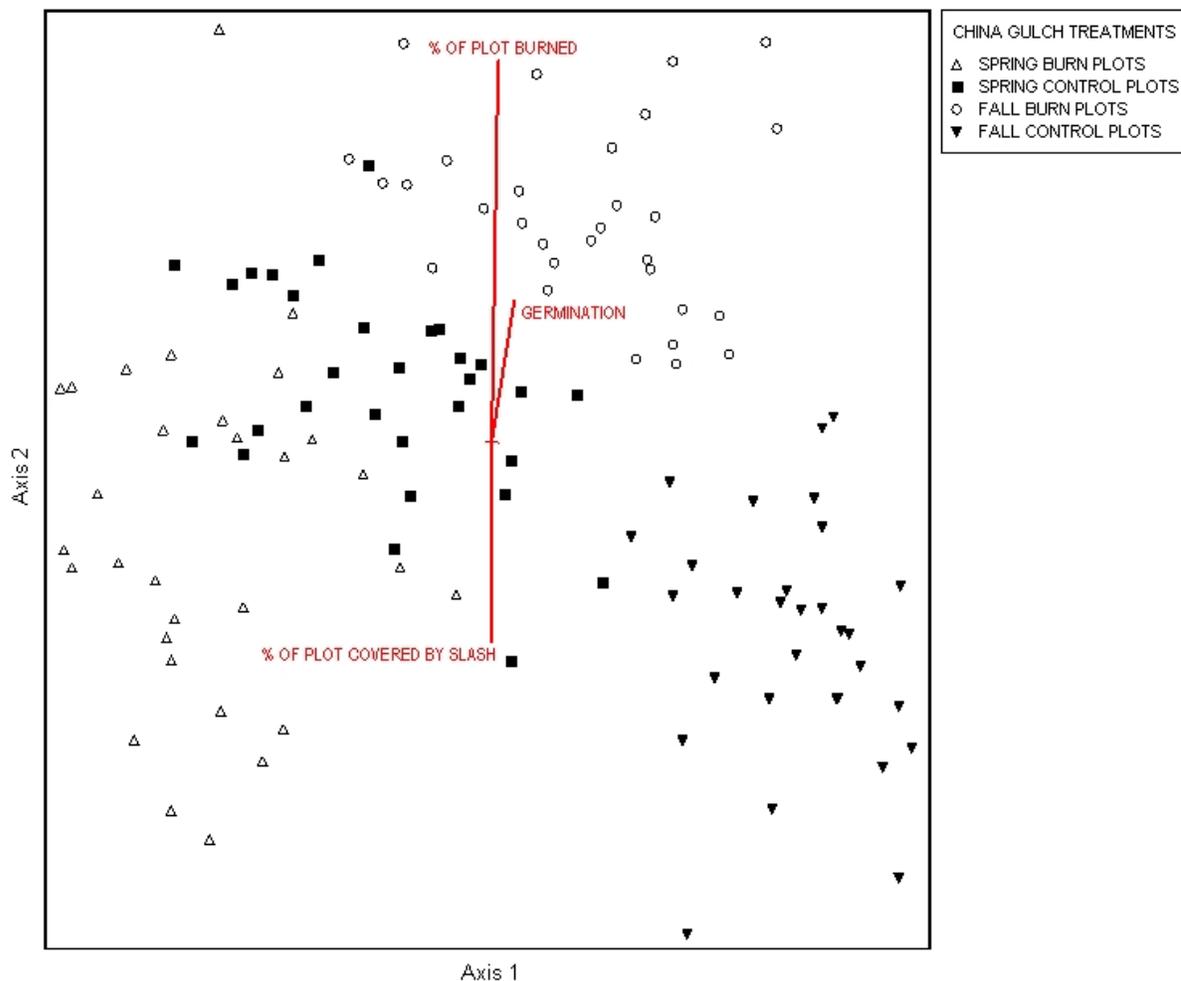
Bunchgrass Species	China Gulch Spring Burn 2007-2008	Hukill Hollow Spring Burn 2007-2008	China Gulch Fall Burn 2007-2008	Hukill Hollow Fall Burn 2007-2008
<i>Achnatherum lemmonii</i>	no change	no change	increase	increase
<i>Bromus carinatus</i>	no change	no change	decrease	increase
<i>Elymus glaucus</i>	no change	no change	decrease	decrease
<i>Festuca idahoensis</i> ssp. <i>roemeri</i>	no change	no change	increase	increase



ACLE = *Achnatherum lemmonii*; *BRCA* = *Bromus carinatus*; *ELGL* = *Elymus glaucus*; *FEID* = *Festuca idahoensis* ssp. *roemeri*

Figure 24. Density of individual live germinants in fall burn plots for China Gulch and Hukill Hollow from 2007 to 2008.

Descriptive Analyses of Environmental Variables Predicting Germination Success



3 dimensions (axes), cutoff $R^2 = 0.100$; vector scaling 200%; joint plots rotated based on treatment variable in second matrix; Sorensen (Bray-Curtis) distance measure; 50 runs; 400 iterations

Figure 25. NMS ordination of plant species and environmental variables for the China Gulch May 2007 dataset.

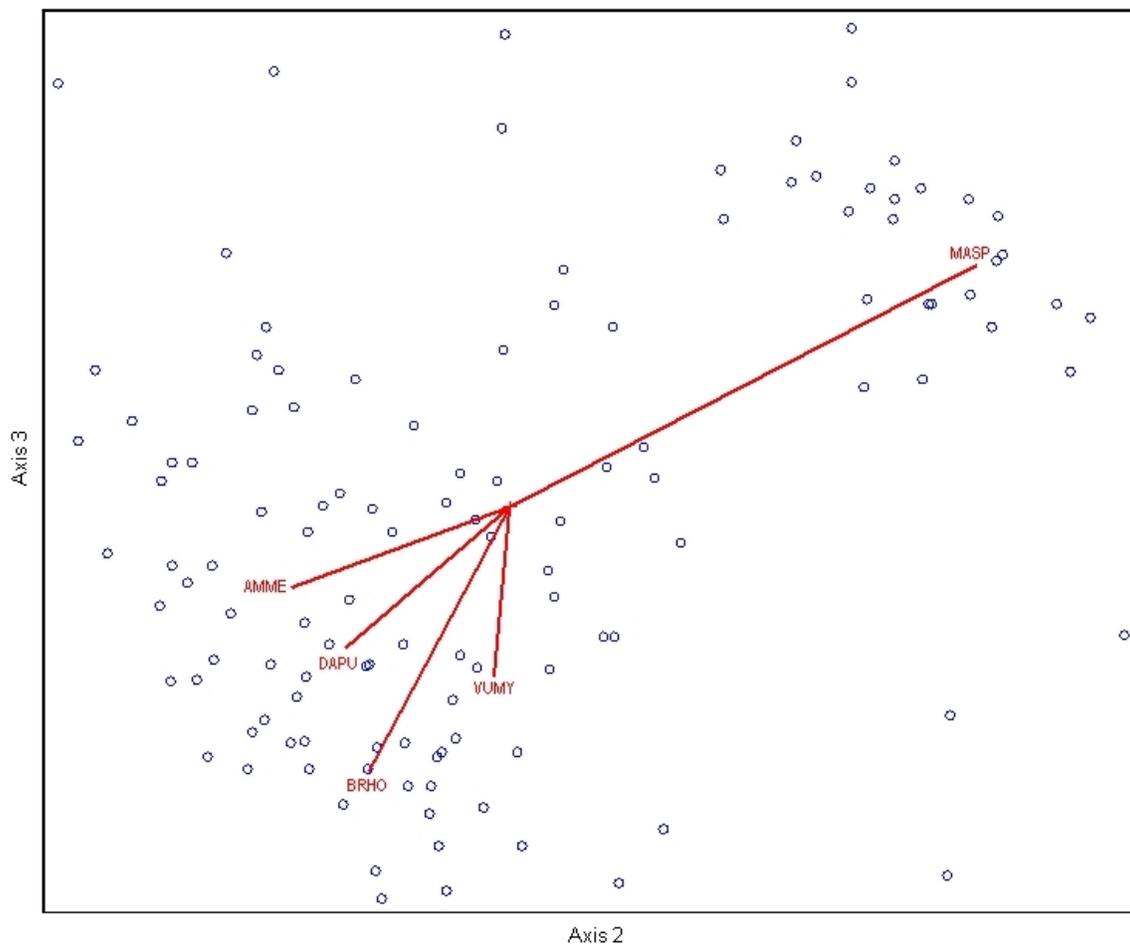
Axis 2 is most strongly correlated to the amount of plot that was burned, which is indirectly related to the fuel loads (cover of slash) prior to prescribed burn treatments.

Germination was most successful in the seeded fall burned plots where thatch and slash

were removed. Germination did not take place in the seeded spring burn plots due to (1) high ambient temperatures and low moisture following seeding and (2) lack of bare ground, since thatch was only partially removed by the patchy spring burn. Fall control plots are grouped by themselves, as they were in the pre-treatment data due to a slight change in aspect resulting in a higher abundance of certain plant species (highest percentage of oak overstory and *Madia* sp.). Those plant species with the highest abundance and frequency in plots biased the structure of the ordination (Table 16).

Table 16. Pearson's (r) correlation coefficients of environmental variables with ordination axes for China Gulch 2007 plots ordered by species abundance.

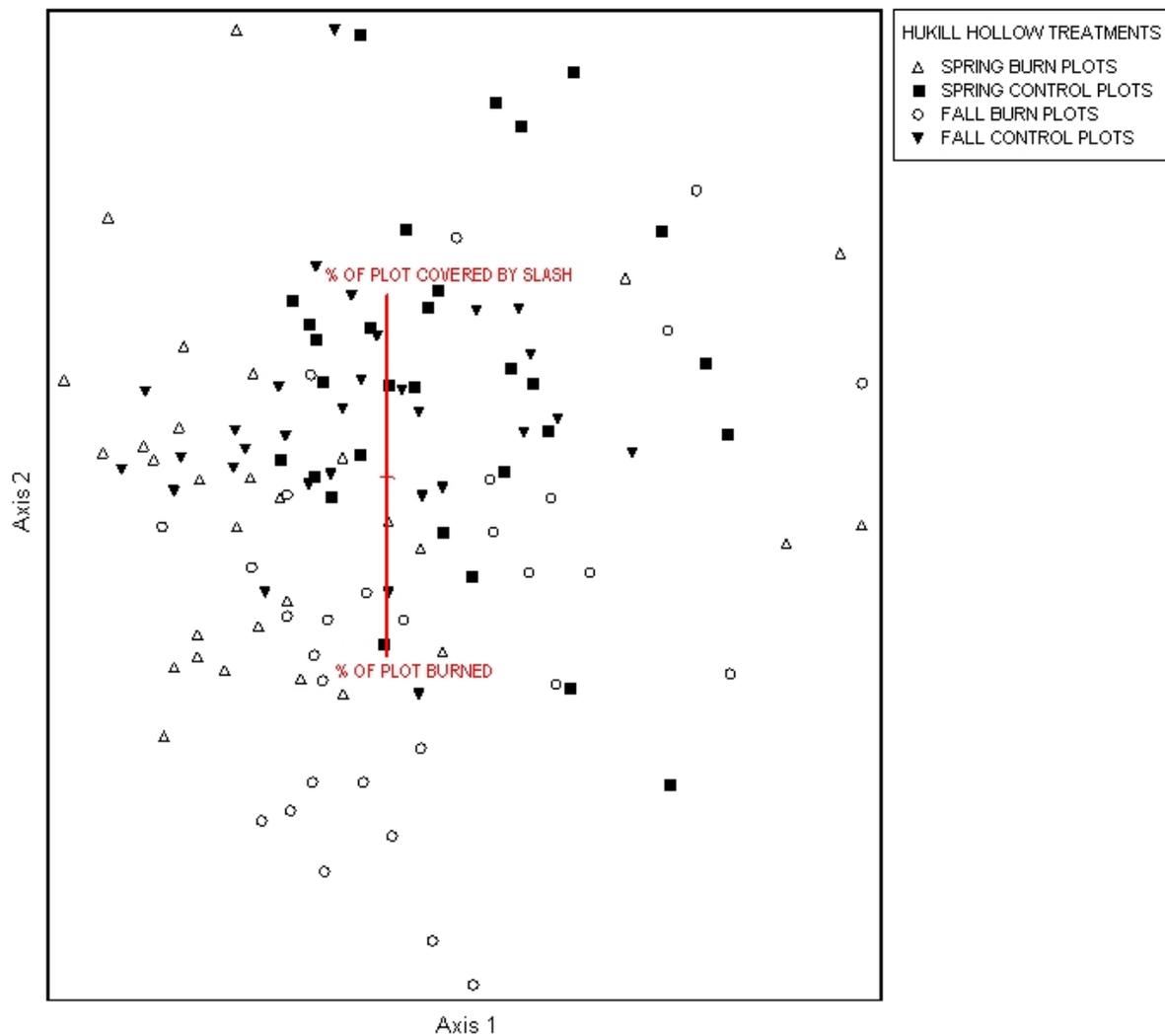
Acronym for Environmental Variable	Measurement for each Environmental Variable	Axis 1 (r)	Axis 2 (r)	Axis 3 (r)
ROCK	Cover of rock	-.170	.136	.186
THAT	Cover of thatch	.158	.062	-.280
BARE	Cover of bare soil	.199	.247	.278
SLAS	Cover of slash	.158	-.489	.061
BURN	Percentage of plot burned	-.107	.615	.111
CHAR	Cover of charcoal	-.221	-.194	-.061



3 dimensions (axes), cutoff $R^2 = 0.200$; vector scaling 150%; Sorensen (Bray-Curtis) distance measure; 50 runs; 400 iterations

Figure 26. NMS ordination of plant species abundance for the China Gulch May 2007 dataset.

Following the prescribed burns and postfire seeding, in plots where MASP (*Madia* sp.) was the most abundant, species diversity was low. AMME (*Amsinckia menziesii* var. *intermedia*), DAPU (*Daucus pusillus*), BRHO (*Bromus hordeaceus*), and VUMY (*Vulpia myuros*) dominated plots where *Madia* sp. was less abundant.



3 dimensions (axes), cutoff $R^2 = 0.100$; vector scaling 200%; joint plots rotated based on treatment variable in second matrix; Sorensen (Bray-Curtis) distance measure; 50 runs; 400 iterations

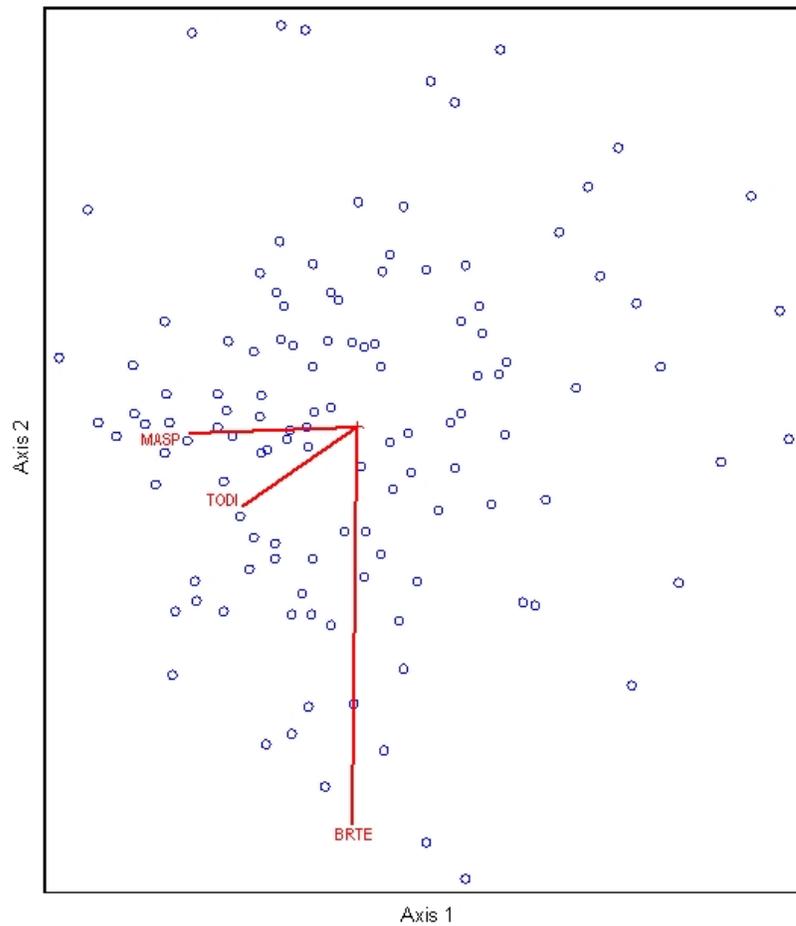
Figure 27. NMS ordination of plant species and environmental variables for the Hukill Hollow May 2007 dataset.

Results are similar to China Gulch findings where Axis 2 is most strongly correlated with the amount of plot that burned, which is indirectly related to fuel loads (cover of slash) prior to prescribed burn treatments. Germination was successful in the

seeded fall burned plots where all 30 plots were completely burned. Removal of thatch, allowing seeds to come in contact with bare soil, increased germination success with the onset of winter rains. Seeds sowed in the spring burn plots, where high fuel moisture yielded a patchy burn pattern, did not germinate. Plots are more interspersed among treatments than the China Gulch NMS ordination, indicating the effect of treatment may not have been as significant at Hukill Hollow (Table 17).

Table 17. Pearson's (*r*) correlation coefficients of environmental variables with ordination axes for Hukill Hollow 2007 plots ordered by species abundance.

Acronym for Environmental Variable	Measurement for each Environmental Variable	Axis 1 (<i>r</i>)	Axis 2 (<i>r</i>)	Axis 3 (<i>r</i>)
ROCK	Cover of rock	.020	-.165	.158
THAT	Cover of thatch	.090	.258	.289
BARE	Cover of bare soil	-.076	-.062	.229
SLAS	Cover of slash	.037	.355	.302
BURN	Percentage of plot burned	.041	-.351	.494
CHAR	Cover of charcoal	.036	-.053	.257



3 dimensions (axes), cutoff $R^2 = 0.200$; vector scaling 150%; Sorensen (Bray-Curtis) distance measure; 50 runs; 400 iterations

Figure 28. NMS ordination of plant species abundance for the Hukill Hollow May 2007 dataset.

Following the prescribed burns and postfire seeding, plots dominated by MASP (*Madia* sp.) were resilient to invasion by the invasive annual grass, BRTE (*Bromus tectorum*).

DISCUSSION

Comparisons of Spring and Fall Prescribed Fires at China Gulch and Hukill Hollow

In this study fire season and intensity significantly affected vegetation response and success of postfire seeding in oak-chaparral communities of southwestern Oregon. The patchy, low-intensity spring burns differed dramatically from the moderate- to severe-intensity fall burns at both sites. In the spring burn treatment blocks at both sites, the low-intensity burn did not consume fuels and burn all plots. By contrast, the moderate- to high-severity fire in fall treatment blocks consumed fuels and left all plots burned at both China Gulch and Hukill Hollow. Burn severity was influenced by fuel loading and moisture content of fuels and soils present at the time the prescribed fires were implemented. Collectively, the highest fuel loads and highest fire severity were observed in the Hukill Hollow fall treatment block.

Although the prescribed fires in this study coincided with the indigenous and natural fire season of the past, returning fire to a landscape wholly altered from its historic plant composition and structure cannot be viewed as the restoration of a natural process, but rather, the introduction of an additional land management action. Data from the fall prescribed fires suggest that the greater fuel loads concentrated on the ground from brush mastication resulted in longer, hotter burns differing from the historically

quick, low-intensity ground and crown fires set by indigenous populations in late summer prior to European settlement (LaLande 1995, Pullen 1996). Evaluating vegetation response to fire season becomes even more confounded by the residual effects of previous land management actions, making it difficult to predict or plan appropriate actions that support native plant establishment and deter exotic species while reducing fuels at the same time.

Soil Nutrient Response Before and After Spring and Fall Prescribed Fires at
China Gulch and Hukill Hollow

Due to the high concentration of fuel loads left on the ground, prescribed fires implemented in mechanically masticated woodlands/shrublands will burn at higher temperatures and for a longer duration. How this specific type of fire influences abiotic and biotic soil elements is critical to successful native plant establishment following prescribed burn treatments (Keeley et al. 2002). The relationship between fire and soil is considerable, since fires are fed by the combustion of organic matter directly in contact with or as part of the soil. (Agee 1993, McNabb and Cromack Jr. 1990). Consequently, the influence of fire on nutrient cycling is determined by the environment in which the fire occurs (Agee 1993). Specifically, the ability of a site to recover from an intense fire is dependent on the amount of nutrients released by the fire, the ability of remaining biota (above and below ground) to capture mobilized nutrients before they are leached out,

and the capacity of the plant community to utilize stored nutrients (Agee 1993, Borchers and Perry 1990, DeDeyn et al. 2004).

Soil samples from both sites exhibited an atypical lack of response of nitrogen to spring and fall burn treatments. Research has shown that soils heated to 300°C lost 25% of their nitrogen while soil heated to 700°C lost 64% nitrogen (Kozlowski et al. 1991). Nitrogen is the most easily volatilized soil nutrient and is wholly dependent on the intensity of the fire and how much soil litter the fire has consumed, while sulfur and phosphorous decreased to a lesser extent (McNabb and Cromack Jr. 1990). Despite the moderate- to severe-intensity fall burns, sulfur increased at both sites. Phosphorus, usually described as decreasing following fire, increased following the spring fire at China Gulch. Increases in phosphorus could be due to low surface fuel loads in spring burn plots at China Gulch compared to surface fuel loads in spring burn plots at Hukill Hollow. Fire also increases soil pH, which can ultimately change the composition of the vegetation community (Kozlowski et al. 1991). Levels of pH did not change following spring and fall burn treatments at either site.

Grouped Comparisons of Plant Communities Before and After Spring and
Fall Prescribed Fires at China Gulch and Hukill Hollow

Group comparisons yielded no significant effect the treatment at China Gulch from either spring or fall prescribed fire treatments. By contrast, at Hukill Hollow a significant treatment effect was apparent following spring and fall prescribed fire treatments.

These results differ from the NMS ordination results, where effect of treatment was clearly illustrated by the grouping pattern of plots by treatment at China Gulch. The NMS ordination for Hukill Hollow displayed the plots as interspersed, with a weak grouping between burned and unburned (control) plots. Discrepancies between the two tests can be attributed to the different variables analyzed for each test. Mantel group tests compare similarity between species abundance and diversity by plot between years. NMS ordinations compare species abundance and diversity by plot *as correlated to environmental variables* between years. Both tests tell only part of the story, while further analyses provide a more complete picture of treatment effect for both China Gulch and Hukill Hollow.

Microclimate Comparisons of Mechanically Thinned and Unthinned

Oak-Chaparral Communities

Beyond increased availability of resources provided by greater oak canopy cover, spatial variation in species composition may be strongly correlated to the change in microclimate under the shady canopy provided by a mature oak (Espeleta et al. 2004). Sikes (2005) reported that of the environmental variables measured in her study canopy cover of oak had the strongest influence on plant composition. Based on this evidence, the patchy distribution of plant species found at both China Gulch and Hukill Hollow was an effect of the variability in oak canopy cover. As a result, leave islands created in sites that are treated with mechanical mastication could alter plant composition through the alteration of oak overstory. Furthermore, leave islands may promote species diversity by increasing heterogeneity across the landscape and by acting as seed reserves for native species. Although not quantified, I observed changes in understory plant composition in areas where the canopy cover of *Quercus garryana* was greater than 10%, indicating a more mesic microclimate around mature oak trees.

The results of a week-long microclimate study conducted to determine the difference in microclimate between thinned and unthinned oak-chaparral communities confirmed that light intensity was higher in the thinned stand. Increased light intensity in mechanically thinned shrubland communities could break seed dormancy earlier than unthinned shrublands (Bradford 2005). Air temperature was similar on all days except the sunniest day, where ambient temperatures were highest in the thinned stands. Relative humidity was similar on all days except for two days where the wind lowered the relative humidity in the thinned stands. Leaf temperature was higher at the unthinned site on days where the wind played a role in lowering the leaf temperature at the more exposed thinned site. Soil temperature was higher at the thinned site, confirming that removing vegetation will leave exposed soil with a higher soil temperature. A malfunction in one of the soil probes for recording relative water content accounted for results documenting relative soil water content as higher in the unthinned stand on all six days. Relative water content should have been high in the thinned stand, since fewer large woody trees and shrubs were making use of this resource. The more mesic conditions (lower light intensity, leaf temperature, and soil temperature) in unthinned areas, where the native overstory canopy remains, may explain the mosaic pattern of plant species distribution at both sites.

Grouped Comparisons of Life Forms Before and After Spring and Fall

Prescribed Fires at China Gulch and Hukill Hollow

Abundance of native species was significantly higher than exotic species at both sites. Following the spring prescribed fire treatments, native species abundance was still significantly higher than exotic species at China Gulch, while a decrease was observed at Hukill Hollow. Abundance of exotic species was significantly higher than native species following the fall prescribed fire treatments at both China Gulch and Hukill Hollow.

Native Species Response to Spring and Fall Prescribed Fires

Spring Prescribed Fire

Abundance of native species increased following the spring burn at China Gulch, while native species decreased at Hukill Hollow. Variability in response of native species to spring prescribed fire at each site may be due to fuel loads present prior to treatment (Slocum et al. 2003). Fuel loads in spring treatment blocks at Hukill Hollow were higher than spring treatment blocks at China Gulch. The spring prescribed fire at Hukill Hollow left only 6 out of 30 plots unburned, while at China Gulch half (15) of the plots were left unburned. The understory plant community at Hukill Hollow was impacted to a greater degree by the spring burn, which left more bare ground open to colonization by exotic plant species.

Response of Native Perennial Forbs to Spring Prescribed Fire

The increase in native perennial forbs following spring prescribed fire treatment at Hukill Hollow was driven by significant increases in *Monardella odoratissima* Benth. and *Cirsium cymosum* (Greene) J.T. Howell. Neither of these two species was observed in pre-treatment surveys of spring burn plots at China Gulch. Abundance of native perennial forbs did not change following the spring burn treatment at China Gulch.

Response of Native Annual Forbs to Spring Prescribed Fire

The spring prescribed burn at China Gulch stimulated a significant response of native annual forbs dominated by *Amsinckia menziesii* var. *intermedia*, *Cryptantha flaccida*, *Daucus pusillus*, and *Madia* sp. By contrast, the spring burn decreased native annual forbs at Hukill Hollow, with the most significant declines in *Lotus micranthus*. There was also a decline in abundance and frequency of *Clarkia purpurea* at China Gulch. The patchy spring burn did not burn all of the plots; consequently, it is likely that the increase in native annual forbs at China Gulch was an effect of residual seeding from the surrounding unburned vegetation directly into the nutrient-rich bare soil. A study conducted in a mixed-conifer forest in Sequoia National Park found that the patchy spring burn implemented in mechanically thinned stands left islands of unburned vegetation, which may have provided important habitat for postfire plant colonization (Knapp et al. 2004).

A mixed response from native annual forbs following spring burn treatments may be the result of a longer, hotter spring burn at Hukill Hollow, where fuel loads were higher than at China Gulch. LeFer and Clarke (2005) found severe impacts on native annual forbs, specializing in first-year postfire establishment, due to the lethal effects of soil heating from wet spring burns.

Response of Native Perennial Grasses to Spring Prescribed Fire

Abundance of native perennial grasses did not significantly change following spring burns at either site. These species were a minor component or absent in pre-treatment surveys. The seeded native perennial bunchgrasses used in the postfire seeding study did not germinate following the spring prescribed fires, leaving this life form group with a negligible presence in spring treatment blocks at China Gulch and Hukill Hollow.

Response of Native Annual Grasses to Spring Prescribed Fire

Vulpia microstachys is the only native annual grass with a substantial population at China Gulch. Following the spring burn at China Gulch, the abundance and frequency of this species increased, possibly due to residual seeding from unburned individuals. *V. microstachys* decreased significantly in abundance and frequency following spring

burn treatment at Hukill Hollow. This native annual grass is a facultative species that resprouts from persistent seeds following fire. Soil moisture heating from the wet spring burn could have killed seeds.

Response of Native Shrubs and Trees to Spring Prescribed Fire

Abundance of native shrubs and trees increased following the spring burn treatment at China Gulch due to a prolific resprouting response of *Toxicodendron diversilobum*. Of the two woody species found in spring burn plots at Hukill Hollow, *T. diversilobum* and *Eriophyllum lanatum*, no change in abundance and frequency was detected for either species following spring prescribed fire. Maret and Wilson (2005) in the Willamette Valley of Oregon found an increase in seedlings following prescribed burn treatments. Field observations conducted in May 2008 indicate that these results may not be reflective of the change in woody species abundance over time. In all treatment blocks (both burned and unburned plots), seedlings of various woody shrubs were found in much greater abundance throughout both sites, with highest diversity observed at the more mesic Hukill Hollow site.

Fall Prescribed Fire

Fall prescribed fire treatments at both sites resulted in a significant decrease in native plant species. These results contradict the findings that fall prescribed fire increases native species diversity (LeFer and Parker 2005). In east-central Arizona, the abundance of non-native species did not significantly change between high- and low-severity fires (Kuenzi et al. 2008). Studies have found that high-severity fires can be lethal to native seedbanks. LeFer and Parker (2005) observed that germination significantly decreased when seeds were exposed to temperatures reaching 120°C. In southern California chaparral communities, seeds are destroyed when exposed to temperatures greater than 120°C (Keeley 1977). Temperatures below the soil surface recorded at both China Gulch and Hukill Hollow fall prescribed fires ranged between 40°C-82°C. It is not likely that the weak native species response to fall burn treatments was a result of damage to the seedbank. The reasons for the poor regeneration of native species following fall prescribed burns at China Gulch and Hukill Hollow, beyond competition from invasive plant species, are unclear. Keeley et al. (2005a) conducted a study in a chaparral shrubland of the Sierra Nevada Mountains of southern California and found that chaparral communities excluded from fire for more than 100 years were equally successful in postfire recovery as chaparral shrublands experiencing fire within the historic fire-return interval of 50-60 years.

Response of Native Perennial Forbs to Fall Prescribed Fire

Native perennial forbs increased following the fall burn at both China Gulch and Hukill Hollow. At China Gulch, the increase was due to a vigorous response of the native thistle, *Cirsium cymosum*. At Hukill Hollow, the increase was due to a vigorous response of the native herb, *Phacelia heterophylla* Pursh. Laughlin and Fulé (2008) found an increase in *Phacelia heterophylla* following fire in a wildland fire effects study in the Grand Canyon National Park.

The geophyte, *Dichelostemma congestum* (Sm.) Kunth, sprouted robustly following fire at Matney Gulch in the Applegate Valley (Hosten et al. *in prep.*). When comparing pre-treatment data with two years postfire data *Dichelostemma congestum* does not exhibit a significant change in abundance or frequency. However, if we isolate the data to include just the first postfire year, then a significant change in abundance and frequency of *D. congestum* following the fall prescribed fire at China Gulch and Hukill Hollow is evident, conforming to historical observations at Matney Gulch (Hosten et al. *in prep.*). Geophytes also dominate peak abundance in the first postfire year in California chaparral communities (Keeley et al. 2006b).

At China Gulch, pre-treatment (2005) data showed that 13 % of plots contained *D. congestum* with an average cover of 0.1%. One year following the fall prescribed fire, 80 % of the plots contained *D. congestum* with an average cover of 0.4%. In 2007, two years postfire, 43% of the plots contained *D. congestum* with an average cover of 0.4%.

At Hukill Hollow, pre-treatment (2005) data showed 0 % of plots with *D. congestum*. One year following the fall prescribed fire, 53 % of the plots contained *D. congestum* with an average cover of 0.3%. In 2007, two years postfire, 13% of the plots contained *D. congestum* with an average cover of 0.1%.

Response of Native Annual Forbs to Fall Prescribed Fire

Overall, native annual forbs declined following fall burn treatments at China Gulch and Hukill Hollow owing to a decrease in *Madia* sp. Prior to treatments, this species dominated fall burn plots, averaging 30% cover. Two years postfire data find the average cover of *Madia* sp. at 12.6%. Nonetheless, *Madia* sp. was the only native annual forb to increase in frequency following the fall burn treatment at Hukill Hollow. Observations in 2008 indicated that this species is recovering from prescribed fire treatments.

Among the most common individuals at China Gulch, *Amsinckia menziesii* var. *intermedia* and *Gilia capitata* significantly increased in abundance and frequency following the fall prescribed fire.

Other species exhibiting a decline include *Clarkia purpurea*, which decreased in abundance at China Gulch following fall burn treatments. *Daucus pusillus* and *Lotus micranthus* decreased in abundance and frequency following fall burn treatments at

Hukill Hollow. A study on the effect of seasonality of burn on seed germination found that *D. pusillus* was negatively affected by temperatures from both dry and moist fuel loads (LeFer and Parker 2005).

Lotus micranthus was a strong first-year postfire obligate seeder in similar plant communities of the Applegate Valley following mechanical thinning and prescribed fire (Hosten *in prep.*). This study differed at Hukill Hollow, where *L. micranthus* showed a slight but not significant increase in 2006 (one year following fire) and a further decline two years following fire in 2007.

Response of Native Perennial Grasses to Fall Prescribed Fire

Native perennial grasses increased following fall burns at both China Gulch and Hukill Hollow, driven by the strong germination response of *Elymus glaucus*, one of four species seeded into plots as part of the postfire seeding study.

Response of Native Annual Grasses to Fall Prescribed Fire

At China Gulch, *Vulpia microstachys* decreased, while at Hukill Hollow there was no detectable change in abundance or frequency following fall prescribed fire treatment. *V. microstachys* was observed as a dominant pioneer species following fall fire in a related study (Hosten et al. 2007). *V. microstachys* regenerates solely from

soil-stored seed following fire. Although several studies have cited *V. microstachys* as a common annual grass in the first postfire year, fire has a weak-to-no effect on this species (FEIS 2008).

Response of Native Shrubs and Trees to Fall Prescribed Fire

In general, there was no significant change in the abundance of native trees and shrubs following fall prescribed fire at China Gulch. Conversely, at Hukill Hollow, the increase in abundance and frequency of native shrubs and trees was strongly influenced by the vigorous response of *Toxicodendron diversilobum* to the fall burn treatment.

Exotic Species Response to Spring and Fall Prescribed Fire

Three factors directly and indirectly influenced the success of non-native plant species at China Gulch and Hukill Hollow: woody canopy closure, extent of the exotic seedbank, and fire intensity (Keeley et al. 2005b). Prior to burn treatments the woody canopy was significantly reduced by mechanical brush mastication. The more open, recently disturbed environment from fire provides an optimal habitat for invasive species to colonize.

In addition, both study sites were situated within 30 m of a road where invasive roadside weeds are prolific—contributing to the extent of the surrounding exotic seedbank (Alston and Richardson 2006). Fuel breaks and roads are known conduits for

exotic plant invasion (Merriam et al. 2006). In this study, road access was required for the implementation of prescribed burns.

The postfire abundance of invasive plant species was highest in the fall burn treatments at both China Gulch and Hukill Hollow. These results may be cause for concern since native species are limited in their ability to exclude exotic species in disturbed habitats. Furthermore, once non-native plant communities are established, they are difficult to displace (Kulmatiski 2006).

Spring Prescribed Fire

There was no statistically significant response of exotic species to spring burn treatments at either site. Other research, conversely, has found that non-native species decrease in abundance following late spring prescribed fire (LeFer and Parker 2005). Spring prescribed burn treatments are thought to have a detrimental effect on flowering invasive plant species. Increases in exotic annual grasses following spring prescribed fire can delay the germination of native species until the following spring (LeFer and Parker 2005). Delayed native annual plant responses leave bare ground open to colonization by exotic annual grasses (Keeley 2002). Since invasive plants capitalize on frequent disturbance, prescribed fire has not been shown to consistently decrease abundance of invasive plant species (Keeley 2002).

Response of Exotic Perennial Forbs to Spring Prescribed Fire

Hypericum perforatum was the only exotic perennial forb observed at China Gulch and Hukill Hollow and neither site exhibited a noticeable change in abundance of this species following spring burn treatments. Other studies have documented a vigorous response of *H. perforatum* to fire by sprouting from undamaged roots and soil-stored seeds (FEIS 2008).

Response of Exotic Annual Forbs to Spring Prescribed Fire

Erodium cicutarium clearly dominated the significant increase in exotic annual forbs at China Gulch following spring prescribed fire. There was not a significant change in this exotic annual forb following spring prescribed fire at Hukill Hollow. Frequent fire favors populations of *E. cicutarium* by stimulating the seedbank. Seeds buried below the litter layer can survive a moderate-intensity fire (FEIS 2008).

Response of *Torilis nodosa* to spring burn treatments was unclear. Abundance and frequency at China Gulch increased when pre-treatment and post-treatment burn plots were compared but decreased when burn plots and control plots were compared. Conflicting results could be due to the patchy growth pattern of this species. At Hukill Hollow, *T. nodosa* abundance did not significantly change while frequency increased following spring burn treatments.

Response of Exotic Perennial Grasses to Spring Prescribed Fire

Exotic perennial grasses increased following the spring burn at China Gulch due to an increase in *Poa bulbosa* L. This perennial grass has a patchy distribution at China Gulch. Also, the time of year (late May) that the plant surveys were conducted may have resulted in an underestimation of *Poa bulbosa* over three years, depending on the amount of rainfall occurring that spring. Abundance of exotic perennial grasses did not significantly change following spring burn treatment at Hukill Hollow. *P. bulbosa* is typically top-killed by fire, though surviving bulbs may resprout if uncovered by the consumption of the litter layer by fire (FEIS 2008).

Response of Exotic Annual Grasses to Spring Prescribed Fire

Overall, exotic annual grass species did not significantly change in abundance at either site following spring prescribed fires. A closer examination of the most common exotic annual grasses at China Gulch show that the spring burn treatments did not have a significant effect on *Bromus hordeaceus* abundance, but did increase its frequency. Spring prescribed fire decreased the abundance of *B. tectorum*, while increasing its frequency at both sites. *B. rubens* increased significantly in abundance and frequency following spring burn treatments at China Gulch.

Fall Prescribed Fire

Exotic species abundance and frequency increased significantly following fall burns at both China Gulch and Hukill Hollow. Burn severity was highest in fall prescribed fire treatments at both sites. In May of 2006, eight months following the burns, an average of 15% of each plot in the fall burn block was recorded as bare ground at China Gulch, and an average of 17% of each plot was bare ground in the fall burn block at Hukill Hollow. The high-severity burn may have postponed native plant response, leaving bare ground open for the invasion of exotic annual grasses. Abundance of exotic species was higher in the second postfire year compared to the first postfire year.

Response of Exotic Perennial Forbs to Fall Prescribed Fire

Hypericum perforatum was the only exotic perennial forb observed at China Gulch and Hukill Hollow. Neither site exhibited a noticeable change in abundance of this forb following fall burn treatments.

Response of Exotic Annual Forbs to Fall Prescribed Fire

Erodium cicutarium was responsible for the increase in abundance and frequency in exotic annual forbs following the fall burn treatment at China Gulch, while fall burn plots at Hukill Hollow did not demonstrate a significant increase or decrease in *E. cicutarium*. This annual forb was one of several exotics consistently found during the first postfire year by Keeley et al. (1981).

Torilis nodosa decreased in both abundance and frequency following the fall prescribed fire at China Gulch. At Hukill Hollow, *T. nodosa* abundance did not significantly change, though frequency decreased following fall burn treatment.

Response of Exotic Perennial Grasses to Fall Prescribed Fire

Abundance of exotic perennial grasses did not significantly change following fall burn treatments at either site.

Response of Exotic Annual Grasses to Fall Prescribed Fire

Noticeable increases in exotic annual grasses following fall burns at China Gulch were driven by *Bromus rubens* and *Vulpia myuros*. The only species found following prescribed fire that was not present prior to burn treatments was *V. myuros*. This annual grass is top-killed by fire but germinates from the seed bank, allowing it to colonize bare

ground in the first postfire year (FEIS 2008). The absence of *V. myuros* in pre-treatment surveys could be due to the late time summer when plots were sampled. The abundance of this species following prescribed fire treatments suggests it was present prior to treatments, though low in abundance. Both *B. rubens* and *V. myuros* can establish from off-site seed sources following fire (FEIS 2008). The proximity of weed-infested roads to study plots makes this a likely scenario at both China Gulch and Hukill Hollow.

Likewise, Hukill Hollow exhibited a significant increase in exotic annual grasses dominated by *Bromus tectorum* and *Vulpia myuros* following fall prescribed fire.

At China Gulch, when fall burn plots were compared to fall control plots an increase in abundance of *Bromus hordeaceus* was found, though frequency did not significantly change following fall burn treatments. Similarly, abundance of *B. tectorum* did not significantly change following fall burn, while frequency significantly decreased following fall burn treatment at China Gulch

At Hukill Hollow, *Aira caryophyllea* increased in abundance and frequency following fall burn when comparing fall burn plots to fall control plots.

Annual grasses are often described as ruderal species adapted to frequent disturbance with low stress (high nutrients and low competition) (Kuenzi et al. 2008). As a cool-season grass that germinates at the onset of fall rains, *B. tectorum* is able to monopolize soil nutrients and moisture, depriving native grasses of these resources during their growth in the spring (FEIS 2008). Natural disturbances, such as fire, provide ideal conditions for exotic annual grasses to establish (Kuenzi et al. 2008). Keeley et al. (1981) cite examples where *Bromus tectorum* increased from almost zero cover in the

first postfire year to several hundred milacre ($\sim 4 \text{ m}^2$) plots by the fourth postfire season. *B. tectorum* has also been observed to change fuel properties by increasing the frequency, extent, and continuity of fire on the landscape (Brooks et al. 2004). Non-native annual grasses thriving along roads or in canopy gaps may take several years to disperse their seeds into burned areas, but their eventual dominance is broadly documented (Keeley et al. 1981). The establishment of an invasive plant community following fire provides information about the plant community prior to treatment. Low numbers and/or decreased viability of the pre-fire native seedbank, or the poor survival of native seeds during prescribed fire treatments, suggest that the plant community has been degraded (Keeley et al. 1981). Related research in the Applegate Valley has described plant communities like those found at China Gulch and Hukill Hollow as a disturbance-mediated woodland/chaparral (Pfaff 2007). Multiple disturbances have compromised the ability of the native seedbank to respond to fire.

Increased nutrient availability in plant communities and level of plant diversity are positively correlated with the presence of soil biota (De Deny et al. 2004). In the absence of root-feeding invertebrates that are killed by fire, combined with the high nutrient availability present immediately following fire, fast-growing annual grasses like *B. tectorum* can invade recently burned bare soil. In cases where soil biota are still present and nutrient levels are initially low, slower-growing species are able to become

established, resulting in a more diverse plant community (De Deyn et al. 2004). Increase growth in grasses, whether exotic or not, is a primary factor in nutrient retention in soils (Kaye et al. 1999).

Grouped Comparisons of Functional Type Assemblages of the Most Common Species Before and After Spring and Fall Prescribed Fires at China Gulch and Hukill Hollow

Mixed chaparral communities are typically dominated by fire-dependent species whose seeds require fire cues such as heat and smoke (LeFer and Parker 2005). The remaining plant species in a chaparral community are considered fire-independent and do not require fire for germination. Further categories can be applied by dividing fire-dependent species into functional types based on the way plants respond to fire cues. Classifications include postfire obligate seeder (non-resprouters specializing in postfire recruitment), facultative species (resprouters with persistent propagules), and obligate resprouter (resprouters with non-persistent propagules specializing in recruitment between fires) (Pausas et al. 2004).

Native annual obligate seeders dominated postfire recovery at both China Gulch and Hukill Hollow. Facultative species that resprout following fire followed in abundance and frequency.

Postfire Seeding Success Following Spring and Fall Prescribed Fires at
China Gulch and Hukill Hollow

Native bunchgrass species have a direct influence on the extent of native species diversity (Maslovat 2002). Bunchgrasses increase resource availability and decrease soil-surface temperatures, allowing other native forbs to thrive (Maslovat 2002). Postfire seeding has shown variable success throughout the Pacific Northwest. Many studies have cited the failure of seeding treatments to prevent erosion of rock and soil, while postfire seeding can sometimes be too successful and prevent the germination and survival of native forbs (Keeley 1996, Keeley et al. 2006a).

Success of postfire seeding can also be influenced by the water-repellent (hydrophobic) soil surface left behind after fire (Agee 1993). Coarse-textured soils covered with a resinous leaf litter are most vulnerable to this process. The water-repellent surface is caused by hydrophobic organic substances released from the leaf litter and surrounding fungi that coat the surface of the soil particle (Agee 1993). The result is a significant decrease in soil infiltration capacity (Kozlowski et al. 1991). Low-intensity fires have been associated with minor water-repellent layers (Agee 1993). Postfire soils in chaparral watersheds are a mosaic of repellent and non-repellent surfaces, lessening the impacts of fire on the hydrologic response of the postfire soil (Hubbert et al. 2006). Furthermore, high surface temperatures and low water infiltration capacity may desiccate sowed seeds.

Germination did not take place in spring burn plots at either study site. There was a significant difference in the total cover of germinants following spring and fall prescribed fires between China Gulch and Hukill Hollow, with the highest mean cover of germinants observed in Hukill Hollow fall burn plots. These results agree with other studies citing a relative increase in germination rates following fall prescribed burns when compared to spring prescribed burns (LeFer and Parker 2005). Concurrently, the temperature of the soil following a fire is typically altered. Daytime temperatures increase due to blackened surfaces and the absence of vegetation to insulate the soil, and nighttime temperatures decrease from the lack of organic matter to trap heat (Agee 1993). Changes in surface soil temperatures can promote or inhibit seedling germination (McNabb and Commack Jr. 1990).

The dramatic difference in germination success between spring and fall treatments may have been influenced by temperature and precipitation patterns occurring during the treatment year. Poor germination was observed in a higher-than-average rainfall year (2006) following spring prescribed fire, and greater germination success was observed in a lower-than-average rainfall year (2005) following fall prescribed fire. Therefore, the data suggest that seeded bunchgrass response to spring and fall prescribed burns is not dependent on the amount of precipitation or temperature extremes in a given year but, instead, on whether or not the stratification requirements are met soon after seeds have been sowed. All four bunchgrass species, except *Elymus glaucus*, require stratification by cool, wet winter months and subsequent increases in soil temperature and light intensity to germinate. The hot, dry weather following spring burn treatments during 2006 may

have offset the stratification requirements for seeds sowed after the spring prescribed fires, increasing the chance of desiccation before the onset of fall rains. Implementing prescribed fires during spring months may compromise the ability of seeds to germinate if stratification requirements are not met (Knox and Clarke 2006).

Herbivore and avian predation may have influenced the germination and survival of seeded bunchgrass species as well. Mills (1986) found that small mammals, ants, and phloem-feeding insects altered the number of *Ceanothus* sp. seeds in chaparral communities. At both sites, seeds were sown by scattering them on top of newly burnt soil, making them an easy target for seed predation.

None of the species seeded in control plots germinated, with the exception of *Festuca idahoensis* ssp. *roemeri*. One individual was observed in a fall control plot growing out of a gopher mound. Data suggest that the thick layer of unburned thatch in control plots impedes the germination of sowed seeds. Gopher activity may have created the disturbance needed by creating bare ground for this species to germinate (Adams and Weitkamp 1992).

Variability in the germination success among the four seeded bunchgrasses was observed. *Elymus glaucus* was the most abundant and frequent germinant recorded in fall burn treatment blocks at both sites. *E. glaucus* does not require stratification or scarification to germinate, only the onset of fall precipitation (FEIS 2008). *Bromus carinatus* was second in abundance and frequency.

Like *Elymus glaucus*, *Bromus carinatus* germinates in the fall once the cool, rainy weather begins. *B. carinatus* germinates slowly, which may place it at a disadvantage to faster growing species such as *B. tectorum*, which germinates at the same time of year.

Festuca idahoensis ssp. *roemerii* was third in abundance and frequency relative to the other seeded species. *F. idahoensis* ssp. *roemerii* germinates once winter precipitation has sufficiently infiltrated soils. Unlike the other seeded bunchgrasses, *F. idahoensis* ssp. *roemerii* requires an after-ripening period of six months to germinate. This delay in germination may account for the increase in germinants observed from 2006 to 2008 at both China Gulch and Hukill Hollow.

Achnatherum lemmonii exhibited the lowest germination rate both in the laboratory and in the field. Seed analysis recorded germination rate and viability at 9%, while tetrazolium tests showed that 88% of seeds tested were viable. The low germination rate recorded during seed testing was likely due to the amount of time seeds were exposed to cold stratification. *A. lemmonii* requires 60-80 days of cold, moist chilling, but seeds were stratified for only seven days at 10°C in the lab (NRCS. 2008). In the field, *A. lemmonii* has high seed dormancy and delayed germination, which reduces its competitive ability to germinate following disturbance (NRCS. 2008).

Survival of Seeded Bunchgrasses Three Years Following Prescribed Fires

Overall, there was no significant change in density between 2007 and 2008 at either site. Frequency of *Achnatherum lemmonii* and *Festuca idahoensis* ssp. *roemerii* increased at China Gulch. *A. lemmonii*, *Bromus carinatus*, and *F. idahoensis* ssp. *roemerii* increased at Hukill Hollow. *Elymus glaucus* decreased at both sites and appeared to be heavily browsed.

Despite the continued presence of seeded bunchgrasses in fall burned plots—since 15 plots were seeded and 15 plots were left unseeded for comparison in each treatment block—invasive annual grasses are dominating unseeded plots. Consequently, the experimental design for this study created a matrix effect, with islands of seeded bunchgrass amid the spread of non-native annual grass.

Plant Community Composition Three Years Following Treatments

Both spring burn treatment blocks at China Gulch and Hukill Hollow are recovering to abundance and diversity levels observed prior to prescribed fire treatments. While the fall burn treatment blocks at both sites are still impacted by large populations of exotic annual grasses, across all treatment blocks at both sites woody species are beginning to re-establish.

Data from this study were subjected to increasingly finer grouping filters to elucidate patterns and trends in plant composition following prescribed fire and postfire

seeding treatments. Life form and functional type groups have been broadly used throughout the literature and have provided land managers with valuable information regarding vegetation response to fire. Many studies document the role of the seed bank in preserving species diversity in fire-prone ecosystems (Parker and Keeley 1989, LeFer and Parker 2005). However, recent findings by Keeley et al. (2006) suggest that these broadly used categories may fail to capture key interactions taking place among individual species as plant succession unfolds in the years following fire. To make sense of post-treatment successional trajectories at both sites, results from each grouping level (plant community, life form, functional type, and individual species) must be considered as a whole.

Plant diversity often fluctuated in the first five years following prescribed fire treatments in southern California chaparral, where species diversity peaked in the first or second postfire year, and then peaked again in the fifth year (Keeley et al. 2006*b*). Species composition differed between the first and fifth postfire year. A species approach rather than a functional type approach shows that the set of species driving diversity immediately following fire are replaced by a different set of species in year five (Keeley et al. 2006*b*). In southern California chaparral communities, the first peak in diversity is dominated by species that specialize in postfire conditions, such as shade-intolerant, short-lived species (Keeley et al. 2006*b*, Franklin et al. 2004). Most of these species are herbaceous, while others may be woody subshrubs that thrive in the open, high-nutrient environment left behind after fire (Franklin et al. 2004). The second peak appears to be driven by new species colonizing open spaces left behind by senescing first- and

second-year postfire species from unburned communities surrounding the fire (Keeley et al. 2006b). As discussed earlier, geophyte abundance can peak during the first postfire year, with subsequent declines in the following years (Keeley et al. 2006b). By contrast, bunchgrass species will thrive as long as conditions are conducive to their survival (Keeley et al. 2006b). Annual species are postfire obligate seeders typically specializing in first postfire year recruitment (Keeley et al. 2006b). These species may spend the majority of their life cycle as dormant seeds waiting for the next fire event (Keeley et al. 2006b).

In this study, species diversity was highest in the first postfire year. At both sites, the most abundant and frequent species were native annual obligate seeders that will likely be replaced by other late-successional species. Indeed, a species approach (examining the response of individual species) may be the surest way of determining biodiversity patterns following natural and human-induced disturbance regimes (Keeley et al. 2006b).

Woody plants can and will displace invasive plant species (Keeley et al. 2005b). Keeley (2002) does not recommend fire as a restoration tool in chaparral shrublands. When the management objective is to deter exotic plant invasion, closed-canopy, mature chaparral communities are relatively resistant to invasion by exotic species. By contrast, chaparral shrublands subjected to frequent fire are converted to sparse shrublands, dominated by a dense understory of non-native annual grasses and forbs within newly created interstitial spaces (Keeley 2002). Thus, woody species recovery is compromised by more frequent fire events (Keeley 2001).

Regeneration of woody species at both sites three years following spring and fall prescribed fire treatments suggests that these historic shrublands have not yet been type converted from woodland/shrublands to exotic annual grass communities. Several species of woody seedlings regenerated in greater abundance than in previous postfire years. Understanding the impacts of prescribed fire on the targeted plant community is critical to the restoration of those individual species. Woody vegetation lacking a mechanism for persistence or postfire recruitment is at risk of being extirpated from burned sites (Keeley 2003). Among those species, the following woody taxa will be discussed based on their frequent occurrence in oak-chaparral communities of southwestern Oregon.

Arctostaphylos viscida and *Ceanothus cuneatus* are obligate seeders that were once co-dominants at both China Gulch and Hukill Hollow. Unlike obligate resprouters that depend on thick bark or underground storage organs for postfire regeneration, obligate seeders rely solely on the persistence of their seeds following fire (Keeley 1975). Studies have shown that despite the increased carbohydrate reserves of resprouters, growth rates are similar between seeders and sprouters following fire (Schwilk and Ackerly 2005).

LeFer and Parker (2005) found that the effect of soil moisture heating can harm soft-coated seeds while benefiting hard-coated seeds like *Ceanothus* spp. and *Arctostaphylos* spp. Differential impacts to dormant seeds can shift species composition following spring prescribed fire by favoring those species with hard-coated seeds (LeFer and Parker 2005). Local observations have found that *C. cuneatus* regenerates quickly following high-intensity fire while *A. viscida* seeds are more easily killed. Typically,

these two chaparral shrubs germinate within two years following fire (FEIS 2008). The weak response of both species in the first two postfire years to prescribed fire treatments and the strong response of seedlings observed in control (unburned) treatments indicate that the moderate- to severe-intensity fires may have caused seed mortality for both *A. viscida* and *C. cuneatus*. Related studies have observed germination of *C. cuneatus* in areas exposed to intense solar heating (Perchemlides et al. 2008, Sikes 2005). My findings are consistent with those results and suggest that *A. viscida* is also capable of germinating in response to heat and light.

Cercocarpus betuloides Nutt. is an obligate seeder found regenerating at Hukill Hollow. *Cercocarpus betuloides* is an evergreen shrub with a plumose tailed achene that disperses in summer (Keeley 1987). *C. montanus* sprouts vigorously following low-intensity fire, with seeds have been killed by high-intensity fire (FEIS 2008). Regeneration of *C. betuloides* was observed in fall burn treatment blocks at Hukill Hollow where the most severe-intensity fire was recorded. There are no adult plants observed in the study area to suggest seedling establishment, so I assume the seedlings are resprouting from surviving root crowns.

Eriodictyon californicum (Hook. and Arn.) Torr., a facultative species found at both China Gulch and Hukill Hollow, has been observed to germinate most successfully under low soil moisture conditions found following fall prescribed burns due to their soft-seed coat (FEIS 2008). Soil moisture heating produced from wet spring fuels can be detrimental to soft-coated seeds. If facultative species, such as *Eriodictyon californicum*, that either resprout or germinate from persistent propagules in response to fire are limited

to resprouting only due to fire intensity, over time the population will decline in abundance. *E. californicum* can also increase the flammability of leaf litter. The leaves of this shrub secrete flammable resin that builds up with age. When these leaves drop to the ground in late summer they can produce highly flammable litter (FEIS 2008).

Garrya fremontii Torr. and *Lonicera hispidula* (Lindl.) Douglas ex Torr. and A. Gray are both obligate resprouters. *Garrya fremontii* was observed resprouting in sparse numbers at China Gulch, while *Lonicera hispidula* was found resprouting vigorously at both sites.

Prunus subcordata Benth. is a facultative species. In May 2008, seedlings were found in abundance around the outside perimeters of the fall burn treatment block at Hukill Hollow. Other species of *Prunus* sp. have been documented as sprouting rapidly and prolifically from root crowns while the aboveground stems are killed by fire (FEIS 2008).

Toxicodendron diversilobum, observed across all treatment blocks at both China Gulch and Hukill Hollow, sprouts vigorously following fire from the root crown or surviving rhizomes. Seedlings emerge in the first postfire year and will continue to do so for several years following fire disturbance (FEIS 2008).

The most significant woody taxon response recorded at the study sites was the survival of mature stands of *Quercus garryana*. In 2008, mortality of mature *Q. garryana* trees in fall treatment blocks at China Gulch and Hukill Hollow was documented. Oak mortality did not take place in spring burn treatment blocks. In fall burn treatment blocks at China Gulch, 60% of standing mature oaks showed no sign of regeneration; at Hukill

Hollow, 88% of mature oaks are presumed dead. *Q. garryana* is a facultative species adapted to frequent low-intensity fire regimes. The bark of mature *Q. garryana* makes this species resistant to flashy low-intensity fire (Agee 1993). Intensity and duration of fall prescribed fire treatments may have influenced the capacity of *Q. garryana* to persist at both sites following fire. The moderate- to high-intensity fall burn, resulting from increased fuel loads from mechanical mastication treatments combined with the low moisture in fuels and soil, produced a residence time of 3.75 minutes around the base of standing oak trees. Kauffman and Martin (1987) found higher mortality rates in *Q. kelloggii* when exposed to fire in fire-suppressed communities where greater fuel loads had accumulated. Oak mortality following natural or prescribed fire is not well-documented. The intensity and duration of fire needed to kill a tree species is a function of bark thickness and the diameter of the tree bole (Agee 1993). Based on an equation developed by Peterson and Ryan (1986) assuming a temperature of 500°C, a species with bark thickness of 0.6 cm could survive for approximately one minute. A tree species with bark thickness of 2.6 cm could survive for about 20 minutes. The *Q. garryana* trees at both study sites ranged from 12.7-30.5 cm in diameter of bole at breast height with a bark thickness estimated at 1-2 cm. *Q. kelloggii* has been found to be susceptible to cambial injury due to its thin bark (Stephens and Finney 2002). A temperature of 60°C for one minute is lethal to plant tissues (Stephens and Finney 2002). Bova and Dickinson (2005) found that if the necrosis is greater than the thickness of the bark by a small amount, vascular cambium will be damaged or killed. Tree mortality is evident based on the extent of damage to leaves, buds, stems, cambium, and fine roots (Stephens and Finney

2002). Fire intensity (defined by fuel consumption) and flame-residence time around tree bole are the two most significant factors predicting tree mortality (Bova and Dickinson 2005, Fryer 2007).

It is possible that trees that appear dead in 2008 will eventually resprout from lateral roots (Agee 1993). The ability of oak species to resprout following fire is dependent on age and size, and mature *Q. garryana* can produce sprouts from lateral roots after the death of the main stem (Agee 1993).

CONCLUSIONS

Results from this study did not support the hypotheses that (1) native species richness increases following fall prescribed fire and (2) abundance of invasive plant species decreases following spring prescribed fire. Instead, the data show that native species significantly decreased at both sites following fall prescribed fire and the abundance of invasive species following spring prescribed fire treatments did not markedly change. The data did support the hypothesis that (3) seeded bunchgrass species exhibit higher germination rates in prescribed fire treatment blocks. Despite lack of germination following spring prescribed fire, postfire seeding following fall prescribed burns was successful, with seeded plots still dominated by seeded bunchgrasses three years following treatments at both China Gulch and Hukill Hollow.

Oak-chaparral communities dominate the wildland-urban interface of the Applegate Valley in southwestern Oregon. Years of ground disturbance have culminated in these oak-chaparral communities becoming more susceptible to an undesirable type conversion, defined as a permanent community shift from native perennials to non-native annuals. Fire exclusion, fuel-reduction treatments, and the edge effect created by roads and adjacent private lands have left these communities degraded and less resilient

(Dodson and Fieldler 2006). Consequently, land managers are faced with the related, but often conflicting, tasks of reducing hazardous fuels and restoring historic plant composition and structure.

This study involved both fuel-reduction (brush mastication followed by fire) and restoration (postfire seeding of native bunchgrasses). In the short term, these two management activities have resulted in an increase in exotic annual grasses and a decrease in native perennials. In the long term, however, as woody seedlings mature and begin to shade out understory species, the balance between native and non-native plants may shift. Resampling study plots five years following treatments will provide more conclusive data regarding the balance between native and non-native plant populations at both sites (Keeley et al. 2005*b*).

Efficiency in policymaking (i.e., developing a single fuels-reduction prescription) has proven unsuccessful in promoting plant diversity and structural complexity (Brunson and Shindler 2004). Dodson et al. (2007) suggest that high biodiversity and species richness can be achieved by implementing a mosaic of different treatments across the landscape (Dodson et al. 2007, Brown et al. 2004). Plant communities with high species diversity are more resistant to non-native plant invasions (MacDougall 2005). Keeley and Fotheringham (2001) recommend the adoption of adaptive management strategies that focus on the careful placement (where and when) of prescribed burns while remaining open to other options for fuel reduction in shrubland-wildland-urban interface lands.

While several studies show that prescribed fire may be beneficial to native plant communities, other studies cite negative impacts (Korb et al. 2003; Keeley and Bond 2001). Data from my study suggest that when postfire seeding does not follow mechanical mastication and prescribed fire, the synergistic effect of these two treatments can promote the invasion of exotic species. Abundance and frequency of exotic species at China Gulch and Hukill Hollow were significantly less in plots subjected to brush mastication alone. A related study, by contrast, found that exotic species were significantly greater in mechanically treated sites when compared to untreated sites (Perchemlides et al. 2008). Each additional ground disturbing activity appears to increase the abundance of invasive plant species incrementally. In my study, exotic annual grasses established in the second year following prescribed burns and invaded only those areas where seeded bunchgrasses did not dominate. For that reason, broadcast application of native seed after prescribed burn treatments could minimize exotic plant invasion.

Discrepancies in the results of both spring and fall prescribed fire treatments analyzed in this study show that the effects of prescribed fire on plant communities are variable and unpredictable. The patchy, low-intensity spring burns were dramatically different from the moderate- to severe-intensity fall burns at both sites. Adjusting the time of burning may produce dramatically different responses in both native and non-native species at China Gulch and Hukill Hollow because of changes in fuel moisture. Burning in late instead of early October or burning in late instead of early April could change the fire severity enough to favor native species. Furthermore, spring burning in areas where a native bunchgrass community is already established has proven successful

in decreasing invasive species by invigorating native perennial grasses through tillering or resprouting (Keeley 2001, Keeley 2002).

A restored chaparral shrubland in the wildland-urban interface will eventually demand further hazardous fuel-reduction treatments. It may not be possible to maintain the historic oak-chaparral communities that once thrived in the valley. Wildfire risk has overruled plant conservation in many areas where public and private lands are intermixed. A type conversion, then, becomes a compelling choice for local public land managers working in oak-chaparral sites with high fuel loads. This study suggests that broadcast seeding of native bunchgrass species following fall, as opposed to spring, prescribed fire in brush masticated oak-chaparral may achieve a desirable type conversion—one that maintains native species richness by establishing a native herbaceous understory community that protects soils and reduces fuels by taking the place of woody species.

Further study is needed to elucidate the potential harmful effects of introducing prescribed fire to mechanically masticated oak-chaparral communities. Beyond the numerous benefits oak trees provide to wildlife, previous arguments cite the key ecological services provided by oak trees to other plant species—primarily, increased resource (i.e., soil moisture and nutrient) availability and acquisition. The elimination of oak species in degraded woodland chaparral systems could lead to the establishment of exotic herbaceous species bringing about the transformation of native to non-native plant communities.

Finally, one of the objectives of this study was to reduce fuel loads in oak-chaparral communities and thereby eliminate the risk of crown fires that kill mature oak trees. Implementing prescribed burn treatments in mechanically masticated oak-chaparral during the drier months of fall (early October) seems to have produced the very result we sought to avoid: significant oak mortality.

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APPENDIX A

PAIRED T-TEST RESULTS FOR LIFE FORMS
AT CHINA GULCH AND HUKILL HOLLOW

PERENNIAL FORBS

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
native perennial forbs	2005 spring burn	2007 spring burn ◦	$P = 0.337$	$P = 0.029$
exotic perennial forbs	2005 spring burn	2007 spring burn	—	$P = 0.702$
native perennial forbs	2005 fall burn	2007 fall burn ▪ ◦	$P = 0.088$	$P = < 0.001$
exotic perennial forbs	2005 fall burn	2007 fall burn	—	$P = 0.154$
native perennial forbs	2007 spring burn ◦	2007 spring control	$P = 0.154$	$P = 0.030$
exotic perennial forbs	2007 spring burn	2007 spring control	—	$P = 0.738$
native perennial forbs	2007 fall burn ▪ ◦	2007 fall control	$P = 0.013$	$P = 0.003$
exotic perennial forbs	2007 fall burn	2007 fall control	$P = 0.161$	$P = 0.488$
native perennial forbs	2007 spring burn	2007 fall burn	$P = 1.699$	$P = 0.211$
exotic perennial forbs	2007 spring burn	2007 fall burn	—	$P = 0.947$

▪ Indicates the group with the highest mean % cover for China Gulch

◦ Indicates the group with the highest mean % cover for Hukill Hollow

ANNUAL FORBS

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
native annual forbs	2005 spring burn ◦	2007 spring burn ▪	$P = 0.048$	$P = 0.016$
exotic annual forbs	2005 spring burn	2007 spring burn ▪	$P = < 0.001$	$P = 0.158$
native annual forbs	2005 fall burn ▪◦	2007 fall burn	$P = 0.007$	$P = < 0.001$
exotic annual forbs	2005 fall burn	2007 fall burn ▪	$P = 0.038$	$P = 0.729$
native annual forbs	2007 spring burn ▪◦	2007 spring control	$P = 0.001$	$P = 0.009$
exotic annual forbs	2007 spring burn	2007 spring control	$P = 0.541$	$P = 0.159$
native annual forbs	2007 fall burn	2007 fall control ▪◦	$P = 0.080$	$P = < 0.001$
exotic annual forbs	2007 fall burn ◦	2007 fall control	$P = 0.693$	$P = 0.067$
native annual forbs	2007 spring burn ▪◦	2007 fall burn	$P = < 0.001$	$P = < 0.001$
exotic annual forbs	2007 spring burn	2007 fall burn	$P = 0.108$	$P = 0.869$

▪ Indicates the group with the highest mean % cover for China Gulch

◦ Indicates the group with the highest mean % cover for Hukill Hollow

PERENNIAL GRASS

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
native perennial grass	2005 spring burn ◦	2007 spring burn	—	$P = 0.086$
exotic perennial grass	2005 spring burn	2007 spring burn ▪	$P = 0.084$	—
native perennial grass	2005 fall burn	2007 fall burn ▪◦	$P = 0.032$	$P = 0.070$
exotic perennial grass	2005 fall burn	2007 fall burn	—	—
native perennial grass	2007 spring burn	2007 spring control ◦	—	$P = < 0.001$
exotic perennial grass	2007 spring burn	2007 spring control	$P = 0.189$	—
native perennial grass	2007 fall burn ◦	2007 fall control	$P = 0.025$	$P = 0.034$
exotic perennial grass	2007 fall burn	2007 fall control	$P = 0.326$	—
native perennial grass	2007 spring burn	2007 fall burn ▪◦	$P = 0.031$	$P = 0.023$
exotic perennial grass	2007 spring burn ▪	2007 fall burn	$P = 0.052$	—

▪ Indicates the group with the highest mean % cover for China Gulch

◦ Indicates the group with the highest mean % cover for Hukill Hollow

ANNUAL GRASS

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
native annual grass	2005 spring burn ◦	2007 spring burn ▪	$P = 0.003$	$P = < 0.001$
exotic annual grass	2005 spring burn	2007 spring burn	$P = 0.253$	$P = 0.526$
native annual grass	2005 fall burn ▪◦	2007 fall burn	$P = < 0.001$	$P = 0.071$
exotic annual grass	2005 fall burn	2007 fall burn ▪◦	$P = < 0.001$	$P = 0.001$
native annual grass	2007 spring burn	2007 spring control ◦	$P = 0.225$	$P = 0.001$
exotic annual grass	2007 spring burn	2007 spring control ▪	$P = < 0.001$	$P = 0.632$
native annual grass	2007 fall burn	2007 fall control ◦	$P = 0.165$	$P = 0.008$
exotic annual grass	2007 fall burn ▪◦	2007 fall control	$P = < 0.001$	$P = < 0.001$
native annual grass	2007 spring burn ▪	2007 fall burn ◦	$P = 0.004$	$P = 0.007$
exotic annual grass	2007 spring burn	2007 fall burn ▪◦	$P = < 0.001$	$P = 0.013$

▪ Indicates the group with the highest mean % cover for China Gulch

◦ Indicates the group with the highest mean % cover for Hukill Hollow

NATIVE SHRUBS and TREES

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
native shrubs and trees	2005 spring burn ◦	2007 spring burn ▪	$P = 0.015$	$P = 0.064$
native shrubs and trees	2005 fall burn	2007 fall burn ◦	$P = 0.419$	$P = 0.030$
native shrubs and trees	2007 spring burn ▪◦	2007 spring control	$P = 0.013$	$P = 0.033$
native shrubs and trees	2007 fall burn ◦	2007 fall control	$P = 0.205$	$P = 0.021$
native shrubs and trees	2007 spring burn	2007 fall burn	$P = 0.495$	$P = 0.591$

▪ Indicates the group with the highest mean % cover for China Gulch

◦ Indicates the group with the highest mean % cover for Hukill Hollow

NATIVE and EXOTIC SPECIES

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
native species	2005 spring burn ◦	2007 spring burn ▪	$P = 0.020$	$P = < 0.001$
exotic species	2005 spring burn	2007 spring burn	$P = 0.922$	$P = 0.875$
native species	2005 fall burn ▪◦	2007 fall burn	$P = < 0.001$	$P = 0.003$
exotic species	2005 fall burn	2007 fall burn ▪◦	$P = < 0.001$	$P = 0.001$
native species	2007 spring burn ▪	2007 spring control	$P = < 0.001$	$P = 0.277$
exotic species	2007 spring burn	2007 spring control ▪	$P = < 0.001$	$P = 0.946$
native species	2007 fall burn	2007 fall control ◦	$P = 0.957$	$P = 0.007$

exotic species	2007 fall burn ^{▪◦}	2007 fall control	$P = < 0.001$	$P = < 0.001$
native species	2007 spring burn ^{▪◦}	2007 fall burn	$P = < 0.001$	$P = 0.085$
exotic species	2007 spring burn	2007 fall burn ^{▪◦}	$P = < 0.001$	$P = 0.012$

▪ Indicates the group with the highest mean % cover for China Gulch

◦ Indicates the group with the highest mean % cover for Hukill Hollow

NATIVE versus EXOTIC SPECIES

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
native ^{▪◦} vs. exotic	2005 spring burn	2005 spring burn	$P = < 0.001$	$P = < 0.001$
native ^{▪◦} vs. exotic	2005 fall burn	2005 fall burn	$P = < 0.001$	$P = < 0.001$
native ^{▪◦} vs. exotic	2007 spring burn	2007 spring burn	$P = < 0.001$	$P = < 0.001$
native vs. exotic [▪]	2007 fall burn	2007 fall burn	$P = < 0.001$	$P = 0.547$

▪ Indicates the group with the highest mean % cover for China Gulch

◦ Indicates the group with the highest mean % cover for Hukill Hollow

APPENDIX B

PAIRED T-TEST RESULTS FOR MOST COMMON SPECIES
AT CHINA GULCH AND HUKILL HOLLOW

MOST COMMON SPECIES by abundance and frequency

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Aira caryophyllea</i>	2005 spring burn	2007 spring burn	—	$P = 0.512$
	2005 fall burn	2007 fall burn	—	$P = 0.830$
	2007 spring burn	2007 spring control ◦	—	$P = 0.009$
	2007 fall burn ◦	2007 fall control	—	$P = 0.005$
	2007 spring burn	2007 fall burn ◦	—	$P = 0.001$
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Amsinckia menziesii</i> var. <i>intermedia</i>	2005 spring burn	2007 spring burn ▪	$P = < 0.001$	—
	2005 fall burn	2007 fall burn ▪	$P = 0.005$	—
	2007 spring burn	2007 spring control ◦	$P = 0.402$	—
	2007 fall burn ▪◦	2007 fall control	$P = < 0.001$	—
	2007 spring burn	2007 fall burn ◦	$P = 0.103$	—

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Bromus hordeaceus</i>	2005 spring burn	2007 spring burn	$P = 0.894$	$P = 0.221$
	2005 fall burn	2007 fall burn \bullet°	$P = 0.063$	$P = 0.022$
	2007 spring burn	2007 spring control \bullet	$P = 0.002$	$P = 0.242$
	2007 fall burn \bullet°	2007 fall control	$P = < 0.001$	$P = 0.011$
	2007 spring burn	2007 fall burn	$P = 0.328$	$P = 0.164$
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Bromus rubens</i>	2005 spring burn	2007 spring burn \bullet	$P = 0.004$	—
	2005 fall burn	2007 fall burn \bullet	$P = 0.003$	—
	2007 spring burn	2007 spring control	$P = 0.631$	—
	2007 fall burn \bullet	2007 fall control	$P = 0.047$	—
	2007 spring burn	2007 fall burn \bullet	$P = 0.030$	—
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Bromus tectorum</i>	2005 spring burn \bullet°	2007 spring burn	$P = < 0.001$	$P = 0.046$
	2005 fall burn	2007 fall burn \circ	$P = 0.500$	$P = 0.019$
	2007 spring burn \circ	2007 spring control	$P = 0.174$	$P = 0.025$

<i>Bromus tectorum</i>	2007 fall burn °	2007 fall control	$P = 0.614$	$P = < 0.001$
	2007 spring burn	2007 fall burn ■°	$P = 0.021$	$P = 0.034$
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Clarkia purpurea</i>	2005 spring burn ■	2007 spring burn	$P = < 0.001$	$P = 0.261$
	2005 fall burn ■	2007 fall burn	$P = 0.003$	$P = 0.083$
	2007 spring burn ■	2007 spring control	$P = 0.001$	$P = 0.348$
	2007 fall burn	2007 fall control	$P = 1.000$	$P = 0.167$
	2007 spring burn ■	2007 fall burn	$P = 0.001$	$P = 0.955$
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Cryptantha flaccida</i>	2005 spring burn	2007 spring burn ■	$P = 0.005$	—
	2005 fall burn	2007 fall burn	$P = 0.138$	—
	2007 spring burn ■	2007 spring control	$P = 0.081$	—
	2007 fall burn	2007 fall control	$P = 0.302$	—
	2007 spring burn	2007 fall burn	$P = 0.138$	—

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Daucus pusillus</i>	2005 spring burn	2007 spring burn ▪	$P = < 0.001$	$P = 0.552$
	2005 fall burn ▪°	2007 fall burn	$P = < 0.001$	$P = 0.016$
	2007 spring burn ▪	2007 spring control	$P = 0.004$	$P = 0.163$
	2007 fall burn ▪	2007 fall control	$P = 0.047$	$P = 0.222$
	2007 spring burn ▪	2007 fall burn	$P = < 0.001$	$P = 0.271$
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Eriophyllum lanatum</i>	2005 spring burn °	2007 spring burn	—	$P = 0.075$
	2005 fall burn	2007 fall burn °	—	$P = 0.076$
	2007 spring burn	2007 spring control	—	$P = 0.109$
	2007 fall burn	2007 fall control	—	$P = 0.149$
	2007 spring burn	2007 fall burn	—	$P = 0.419$
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Erodium cicutarium</i>	2007 spring burn ▪	2007 spring control	$P = 0.003$	—
	2007 fall burn ▪	2007 fall control	$P = < 0.001$	—
	2007 spring burn	2007 fall burn ▪	$P = 0.007$	—

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Gilia capitata</i>	2005 spring burn	2007 spring burn	$P = 0.281$	—
	2005 fall burn	2007 fall burn ▪	$P = < 0.001$	—
	2007 spring burn	2007 spring control ▪	$P = 0.058$	—
	2007 fall burn ▪	2007 fall control	$P = < 0.001$	—
	2007 spring burn	2007 fall burn ▪	$P = 0.004$	—
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Lotus micranthus</i>	2005 spring burn ◦	2007 spring burn	$P = 0.109$	$P = < 0.001$
	2005 fall burn ◦	2007 fall burn	$P = 0.000$	$P = < 0.001$
	2007 spring burn ▪	2007 spring control	$P = < 0.001$	$P = 0.138$
	2007 fall burn ◦	2007 fall control	$P = 0.000$	$P = 0.003$
	2007 spring burn ▪	2007 fall burn	$P = < 0.001$	$P = 0.644$
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Madia</i> sp.	2005 spring burn	2007 spring burn	$P = 0.943$	$P = 0.375$
	2005 fall burn ▪◦	2007 fall burn	$P = < 0.001$	$P = < 0.001$
	2007 spring burn ▪◦	2007 spring control	$P = 0.024$	$P = 0.048$

<i>Madia</i> sp.	2007 fall burn	2007 fall control ■°	$P = < 0.001$	$P = < 0.001$
	2007 spring burn °	2007 fall burn	$P = 0.257$	$P = 0.004$
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Toxicodendron diversilobum</i>	2005 spring burn °	2007 spring burn ■	$P = 0.020$	$P = 0.097$
	2005 fall burn	2007 fall burn °	$P = 0.445$	$P = 0.024$
	2007 spring burn ■°	2007 spring control	$P = 0.013$	$P = 0.060$
	2007 fall burn ■°	2007 fall control	$P = 0.005$	$P = < 0.001$
	2007 spring burn	2007 fall burn	$P = 0.964$	$P = 0.450$
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Torilis nodosa</i>	2005 spring burn	2007 spring burn ■	$P = 0.001$	$P = 0.816$
	2005 fall burn ■°	2007 fall burn	$P = 0.026$	$P = 0.052$
	2007 spring burn	2007 spring control ■°	$P = 0.036$	$P = 0.021$
	2007 fall burn	2007 fall control ■	$P = < 0.001$	$P = 0.855$
	2007 spring burn ■	2007 fall burn	$P = 0.020$	$P = 0.435$

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Vulpia microstachys</i>	2005 spring burn ◦	2007 spring burn ▪	$P = 0.003$	$P = < 0.001$
	2005 fall burn ▪◦	2007 fall burn	$P = < 0.001$	$P = 0.071$
	2007 spring burn	2007 spring control ◦	$P = 0.225$	$P = < 0.001$
	2007 fall burn	2007 fall control ◦	$P = 0.165$	$P = 0.008$
	2007 spring burn ▪	2007 fall burn ◦	$P = 0.004$	$P = 0.007$
Response Variable	Treatment and Year		China Gulch	Hukill Hollow
<i>Vulpia myuros</i>	2007 spring burn	2007 spring control ▪	$P = < 0.001$	—
	2007 fall burn ▪	2007 fall control	$P = < 0.001$	—
	2007 spring burn	2007 fall burn ▪	$P = < 0.001$	—

▪ Indicates the group with the highest mean % cover for China Gulch

◦ Indicates the group with the highest mean % cover for Hukill Hollow

APPENDIX C

CHI-SQUARE TEST RESULTS FOR MOST COMMON SPECIES BY ABUNDANCE
AND FREQUENCY AT CHINA GULCH AND HUKILL HOLLOW

SPRING BURN 2005, 2006, 2007

Species	China Gulch		Hukill Hollow	
	<i>P</i> -value	Frequency	<i>P</i> -value	Frequency
<i>Aira caryophyllea</i>	—	—	0.586	—
<i>Amsinckia menziesii</i> var. <i>intermedia</i>	< 0.001	increased	—	—
<i>Bromus hordeaceus</i>	0.024	increased	0.557	—
<i>Bromus rubens</i>	< 0.001	increased	—	—
<i>Bromus tectorum</i>	0.011	increased	0.004	increased
<i>Clarkia purpurea</i>	0.051	decreased	0.677	—
<i>Cryptantha flaccida</i>	0.003	increased	—	—
<i>Daucus pusillus</i>	0.003	increased	0.131	—
<i>Eriophyllum lanatum</i>	—	—	0.708	—
<i>Erodium cicutarium</i>	—	—	—	—
<i>Gilia capitata</i>	0.355	—	—	—
<i>Lotus micranthus</i>	0.186	—	< 0.001	decreased
<i>Madia</i> sp.	0.298	—	0.004	decreased

<i>Toxicodendron diversilobum</i>	0.730	—	0.554	—
<i>Torilis nodosa</i>	< 0.001	increased	< 0.001	increased
<i>Vulpia microstachys</i>	0.007	increased	< 0.001	decreased
<i>Vulpia myuros</i>	—	—	—	—

FALL BURN 2005, 2006, 2007

Species	China Gulch		Hukill Hollow	
	<i>P</i> -value	Frequency	<i>P</i> -value	Frequency
<i>Aira caryophyllea</i>	—	—	0.015	increased
<i>Amsinckia menziesii</i> var. <i>intermedia</i>	0.012	increased	—	—
<i>Bromus hordeaceus</i>	0.186	—	0.108	—
<i>Bromus rubens</i>	< 0.001	increased	—	—
<i>Bromus tectorum</i>	< 0.001	decreased	0.002	increased
<i>Clarkia purpurea</i>	0.001	decreased	0.708	—
<i>Cryptantha flaccida</i>	0.074	—	—	—
<i>Daucus pusillus</i>	< 0.001	decreased	0.067	—
<i>Eriophyllum lanatum</i>	—	—	0.919	—
<i>Erodium cicutarium</i>	—	—	—	—
<i>Gilia capitata</i>	< 0.001	increased	—	—
<i>Lotus micranthus</i>	1.000	—	0.010	decreased
<i>Madia</i> sp.	< 0.001	increased	< 0.001	increased
<i>Toxicodendron diversilobum</i>	0.835	—	0.383	—
<i>Torilis nodosa</i>	0.093	—	0.005	decreased
<i>Vulpia microstachys</i>	< 0.001	decreased	0.142	—
<i>Vulpia myuros</i>	—	—	—	—

SPRING BURN versus SPRING CONTROL 2007

Species	China Gulch		Hukill Hollow	
	P-value	Frequency*	P-value	Frequency*
<i>Aira caryophylla</i>	—	—	—	—
<i>Amsinckia menziesii</i> var. <i>intermedia</i>	0.197	—	—	—
<i>Bromus hordeaceus</i>	0.313	—	0.297	—
<i>Bromus rubens</i>	0.176	—	—	—
<i>Bromus tectorum</i>	0.317	—	0.348	—
<i>Clarkia purpurea</i>	1.000	—	0.572	—
<i>Cryptantha flaccida</i>	0.067	—	—	—
<i>Daucus pusillus</i>	0.076	—	0.791	—
<i>Eriophyllum lanatum</i>	—	—	0.245	—
<i>Erodium cicutarium</i>	< 0.001	decrease	—	—
<i>Gilia capitata</i>	< 0.001	increase	—	—
<i>Lotus micranthus</i>	< 0.001	decrease	0.191	—
<i>Madia</i> sp.	0.185	—	0.639	—
<i>Toxicodendron diversilobum</i>	< 0.001	decrease	0.445	—
<i>Torilis nodosa</i>	0.063	—	0.137	—
<i>Vulpia microstachys</i>	0.001	increase	< 0.001	increase
<i>Vulpia myuros</i>	< 0.001	increase	—	—

* Decrease or increase in frequency refers to changes in density in spring control plots

FALL BURN versus FALL CONTROL 2007

Species	China Gulch		Hukill Hollow	
	P-value	Frequency*	P-value	Frequency*
<i>Aira caryophylla</i>	—	—	—	—
<i>Amsinckia menziesii</i> var. <i>intermedia</i>	< 0.001	decrease	—	—
<i>Bromus hordeaceus</i>	< 0.001	decrease	0.002	decrease
<i>Bromus rubens</i>	0.001	decrease	—	—
<i>Bromus tectorum</i>	0.008	increase	0.166	—
<i>Clarkia purpurea</i>	1.000	—	0.284	—
<i>Cryptantha flaccida</i>	0.718	—	—	—
<i>Daucus pusillus</i>	0.005	decrease	0.114	—
<i>Eriophyllum lanatum</i>	—	—	0.445	—
<i>Erodium cicutarium</i>	< 0.001	decrease	—	—
<i>Gilia capitata</i>	< 0.001	decrease	—	—
<i>Lotus micranthus</i>	1.000	—	< 0.001	decrease
<i>Madia</i> sp.	0.150	—	1.000	—
<i>Toxicodendron diversilobum</i>	0.003	decrease	0.560	—
<i>Torilis nodosa</i>	< 0.001	increase	0.425	—
<i>Vulpia microstachys</i>	0.002	increase	0.222	—
<i>Vulpia myuros</i>	0.150	—	—	—

* Decrease or increase in frequency refers to changes in density in fall control plots

APPENDIX D

PAIRED T-TEST RESULTS FOR SEEDED BUNCHGRASS
SPECIES AT CHINA GULCH AND HUKILL HOLLOW

DENSITY of LIVE GERMINANTS

Response Variable	Treatment and Year		China Gulch	Hukill Hollow
number of live germinants	2007 fall burn plots	2008 fall burn plots	$P = 0.534$	$P = 0.216$

APPENDIX E

ANOVA: TWO-FACTOR AND SINGLE-FACTOR RESULTS
FOR CHINA GULCH AND HUKILL HOLLOW

CHINA GULCH: Sum % cover of germinants in seeded burn plots compared to seeded control plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of seeded grass species	0.023
Between Columns	
H ₀ : There is no difference in the mean % cover of seeded grass species between burned plots and control plots	0.023
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and burned versus control plots on mean % cover of seeded grass species	0.023

x = independent variable 1st factor (burned versus control); 2nd factor (seasonality of burn – spring versus fall); y = dependent variable (% of germinants)

CHINA GULCH: Sum % cover of *Achnatherum lemmonii* in seeded burn plots compared to seeded control plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Achnatherum lemmonii</i>	0.322
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Achnatherum lemmonii</i> between burned plots and control plots	0.322

 Interaction Between Samples and Columns

H₀: There is no interaction between seasonality of burn and burned versus control plots on mean % cover of *Achnatherum lemmonii* 0.322

x = independent variable 1st factor (burned versus control); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Achnatherum lemmonii*)

CHINA GULCH: Sum % cover of *Bromus carinatus* in seeded burn plots compared to seeded control plots

Source of Variation	P-value
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 Between Samples

H₀: There is no effect of seasonality of burn on the mean % cover of *Bromus carinatus* 0.015

 Between Columns

H₀: There is no difference in the mean % cover of *Bromus carinatus* between burned plots and control plots 0.015

 Interaction Between Samples and Columns

H₀: There is no interaction between seasonality of burn and burned versus control plots on mean % cover of *Bromus carinatus* 0.015

x = independent variable 1st factor (burned versus control); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Bromus carinatus*)

CHINA GULCH: Sum % cover of *Elymus glaucus* in seeded burn plots compared to seeded control plots

Source of Variation	P-value
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 Between Samples

H₀: There is no effect of seasonality of burn on the mean % cover of *Elymus glaucus* 0.044

 Between Columns

H₀: There is no difference in the mean % cover of *Elymus glaucus* between burned plots and control plots 0.044

 Interaction Between Samples and Columns

H₀: There is no interaction between seasonality of burn and burned versus control plots on mean % cover of *Elymus glaucus* 0.044

x = independent variable 1st factor (burned versus control); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Elymus glaucus*)

CHINA GULCH: Sum % cover of *Festuca idahoensis* ssp. *roemeri* in seeded burn plots compared to seeded control plots

Source of Variation	<i>P</i> -value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Festuca idahoensis</i> ssp. <i>roemeri</i>	— (no FERO)
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Festuca idahoensis</i> ssp. <i>roemeri</i> between burned plots and control plots	— (no FERO)

 Interaction Between Samples and Columns

H₀: There is no interaction between seasonality of burn and burned versus control plots on mean % cover of *Festuca idahoensis* ssp. *roemeri* —
(no FERO)

x = independent variable 1st factor (burned versus control); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Festuca idahoensis* ssp. *roemeri*)

CHINA GULCH: Sum % cover of *Achnatherum lemmonii* in 2006 seeded burn plots compared to 2007 seeded burn plots

Source of Variation	<i>P</i> -value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Achnatherum lemmonii</i>	0.255
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Achnatherum lemmonii</i> between 2006 and 2007	0.415

 Interaction Between Samples and Columns

H₀: There is no interaction between seasonality of burn and 2006 versus 2007 on mean % cover of *Achnatherum lemmonii* 0.415

x = independent variable 1st factor (2006 versus 2007); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Achnatherum lemmonii*)

CHINA GULCH: Sum % cover of *Bromus carinatus* in 2006 seeded burn plots compared to 2007 seeded burn plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Bromus carinatus</i>	0.001
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Bromus carinatus</i> between 2006 and 2007	0.247
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and 2006 versus 2007 on mean % cover of <i>Bromus carinatus</i>	0.247

x = independent variable 1st factor (2006 versus 2007); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Bromus carinatus*)

CHINA GULCH: Sum % cover of *Elymus glaucus* in 2006 seeded burn plots compared to 2007 seeded burn plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Elymus glaucus</i>	0.030
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Elymus glaucus</i> between 2006 and 2007	0.065

 Interaction Between Samples and Columns

H₀: There is no interaction between seasonality of burn and 2006 versus 2007 on mean % cover of *Elymus glaucus* 0.065

x = independent variable 1st factor (2006 versus 2007); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Elymus glaucus*)

CHINA GULCH: Sum % cover of *Festuca idahoensis* ssp. *roemeri* in 2006 seeded burn plots compared to 2007 seeded burn plots

Source of Variation	P-value
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 Between Samples

H₀: There is no effect of seasonality of burn on the mean % cover of *Festuca idahoensis* ssp. *roemeri* 0.003

 Between Columns

H₀: There is no difference in the mean % cover of *Festuca idahoensis* ssp. *roemeri* between 2006 and 2007 0.003

 Interaction Between Samples and Columns

H₀: There is no interaction between seasonality of burn and 2006 versus 2007 on mean % cover of *Festuca idahoensis* ssp. *roemeri* 0.003

x = independent variable 1st factor (2006 versus 2007); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Festuca idahoensis* ssp. *roemeri*)

CHINA GULCH: % cover of plot burned in spring burn plots compared to fall burn plots

Source of Variation	P-value
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 Between Groups

H₀: There is no effect of seasonality of burn on the mean % cover of ground burned in plot < 0.001

x = independent variable single factor (seasonality of burn – spring versus fall);
y = dependent variable (% cover of plot burned)

HUKILL HOLLOW: Sum % cover of germinants in seeded burn plots compared to seeded control plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of seeded grass species	0.060
Between Columns	
H ₀ : There is no difference in the mean % cover of seeded grass species between burned plots and control plots	0.058
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and burned versus control plots on mean % cover of seeded grass species	< 0.001

x = independent variable 1st factor (burned versus control); 2nd factor (seasonality of burn – spring versus fall); y = dependent variable (% of germinants)

HUKILL HOLLOW: Sum % cover of *Achnatherum lemmonii* in seeded burn plots compared to seeded control plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Achnatherum lemmonii</i>	0.089
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Achnatherum lemmonii</i> between burned plots and control plots	0.089
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and burned versus control plots on mean % cover of <i>Achnatherum lemmonii</i>	0.089

*x = independent variable 1st factor (burned versus control); 2nd factor (seasonality of burn – spring versus fall); y = dependent variable (% cover of *Achnatherum lemmonii*)*

HUKILL HOLLOW: Sum % cover of *Bromus carinatus* in seeded burn plots compared to seeded control plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Bromus carinatus</i>	0.612
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Bromus carinatus</i> between burned plots and control plots	0.653
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and burned versus control plots on mean % cover of <i>Bromus carinatus</i>	< 0.001

x = independent variable 1st factor (burned versus control); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Bromus carinatus*)

HUKILL HOLLOW: Sum % cover of *Elymus glaucus* in seeded burn plots compared to seeded control plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Elymus glaucus</i>	0.013
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Elymus glaucus</i> between burned plots and control plots	0.009
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and burned versus control plots on mean % cover of <i>Elymus glaucus</i>	0.013

x = independent variable 1st factor (burned versus control); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Elymus glaucus*)

HUKILL HOLLOW: Sum % cover of *Festuca idahoensis* ssp. *roemeri* in seeded burn plots compared to seeded control plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Festuca idahoensis</i> ssp. <i>roemeri</i>	0.012
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Festuca idahoensis</i> ssp. <i>roemeri</i> between burned plots and control plots	0.054
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and burned versus control plots on mean % cover of <i>Festuca idahoensis</i> ssp. <i>roemeri</i>	0.054

x = independent variable 1st factor (burned versus control); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Festuca idahoensis* ssp. *roemeri*)

HUKILL HOLLOW: Sum % cover of *Achnatherum lemmonii* in 2006 seeded burn plots compared to 2007 seeded burn plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Achnatherum lemmonii</i>	0.027
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Achnatherum lemmonii</i> between 2006 and 2007	0.407
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and 2006 versus 2007 on mean % cover of <i>Achnatherum lemmonii</i>	0.287

x = independent variable 1st factor (2006 versus 2007); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Achnatherum lemmonii*)

HUKILL HOLLOW: Sum % cover of *Bromus carinatus* in 2006 seeded burn plots compared to 2007 seeded burn plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Bromus carinatus</i>	0.001
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Bromus carinatus</i> between 2006 and 2007	0.833
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and 2006 versus 2007 on mean % cover of <i>Bromus carinatus</i>	0.528

x = independent variable 1st factor (2006 versus 2007); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Bromus carinatus*)

HUKILL HOLLOW: Sum % cover of *Elymus glaucus* in 2006 seeded burn plots compared to 2007 seeded burn plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Elymus glaucus</i>	0.006
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Elymus glaucus</i> between 2006 and 2007	0.028
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and 2006 versus 2007 on mean % cover of <i>Elymus glaucus</i>	0.038

x = independent variable 1st factor (2006 versus 2007); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Elymus glaucus*)

HUKILL HOLLOW: Sum % cover of *Festuca idahoensis* ssp. *roemeri* in 2006 seeded burn plots compared to 2007 seeded burn plots

Source of Variation	P-value
Between Samples	
H ₀ : There is no effect of seasonality of burn on the mean % cover of <i>Festuca idahoensis</i> ssp. <i>roemeri</i>	0.002
Between Columns	
H ₀ : There is no difference in the mean % cover of <i>Festuca idahoensis</i> ssp. <i>roemeri</i> between 2006 and 2007	0.330
Interaction Between Samples and Columns	
H ₀ : There is no interaction between seasonality of burn and 2006 versus 2007 on mean % cover of <i>Festuca idahoensis</i> ssp. <i>roemeri</i>	0.330

x = independent variable 1st factor (2006 versus 2007); 2nd factor (seasonality of burn – spring versus fall); *y* = dependent variable (% cover of *Festuca idahoensis* ssp. *roemeri*)

HUKILL HOLLOW: % cover of plot burned in spring burn plots compared to fall burn plots

Source of Variation	P-value
Between Groups	
H ₀ : There is no effect of seasonality of burn on the mean % cover of ground burned in plot	< 0.001

x = independent variable single factor (seasonality of burn – spring versus fall);
y = dependent variable (% cover of plot burned)

APPENDIX F

PAIRED T-TEST RESULTS FOR SOIL NUTRIENTS
AT CHINA GULCH AND HUKILL HOLLOW

Response Variable	Treatment		China Gulch	Hukill Hollow
Organic Matter (% Carbon)	fall unburned samples	fall burned samples	$P = 0.415$	$P = 0.544$
Organic Matter (% Carbon)	spring unburned samples	spring burned samples	$P = 0.480$	$P = 0.378$
Response Variable	Treatment		China Gulch	Hukill Hollow
Organic Matter Nitrogen (ENR* lbs/A)	fall unburned samples	fall burned samples	$P = 0.407$	$P = 0.532$
Organic Matter Nitrogen (ENR* lbs/A)	spring unburned samples	spring burned samples	$P = 0.488$	$P = 0.367$
Response Variable	Treatment		China Gulch	Hukill Hollow
Phosphorus ** (Weak Bray) (ppm)	fall unburned samples	fall burned samples °	$P = 0.315$	$P = 0.074$
Phosphorus (Weak Bray) (ppm)	spring unburned samples	spring burned samples ▪	$P = 0.023$	$P = 0.949$
Response Variable	Treatment		China Gulch	Hukill Hollow
Potassium*** (ppm)	fall unburned samples	fall burned samples °	$P = 0.155$	$P = 0.015$
Potassium*** (ppm)	spring unburned samples	spring burned samples	$P = 0.581$	$P = 0.868$
Response Variable	Treatment		China Gulch	Hukill Hollow
Magnesium**** (ppm)	fall unburned samples	fall burned samples ▪°	$P = 0.037$	$P = 0.094$

Magnesium**** (ppm)	spring unburned samples	spring burned samples	$P = 0.116$	$P = 0.959$
Response Variable	Treatment		China Gulch	Hukill Hollow
Calcium (ppm)	fall unburned samples	fall burned samples [▪] [◦]	$P = 0.032$	$P = 0.073$
Calcium (ppm)	spring unburned samples	spring burned samples	$P = 0.827$	$P = 0.330$
Response Variable	Treatment		China Gulch	Hukill Hollow
Sodium (ppm)	fall unburned samples	fall burned samples	$P = 0.103$	$P = 0.159$
Sodium (ppm)	spring unburned samples	spring burned samples	$P = 0.181$	$P = 0.449$
Response Variable	Treatment		China Gulch	Hukill Hollow
Soil pH	fall unburned samples	fall burned samples	$P = 0.769$	$P = 0.689$
Soil pH	spring unburned samples	spring burned samples	$P = 0.624$	$P = 0.516$
Response Variable	Treatment		China Gulch	Hukill Hollow
Sulfur (ppm)	fall unburned samples	fall burned samples [▪] [◦]	$P = 0.028$	$P = 0.020$
Sulfur (ppm)	spring unburned samples	spring burned samples	$P = 0.116$	$P = 0.213$

▪ Indicates the group with the highest mean for China Gulch

◦ Indicates the group with the highest mean for Hukill Hollow

* ENR = estimated nitrogen release

** multiply the results in ppm (parts per million)
by 4.6 to convert to pounds per acre P_2O_5

*** multiply the results in ppm (parts per million)
by 2.4 to convert to pounds per acre K_2O

**** multiply the results in ppm (parts per million)
by 2 to convert to pounds per acre to the elemental form

SOIL ANALYSES comparing China Gulch versus Hukill Hollow

Response Variable	Treatment		<i>P</i> -value
Organic Matter (% Carbon)	China Gulch fall burned samples	Hukill Hollow fall burned samples	0.951
Organic Matter (% Carbon)	China Gulch spring burned samples	Hukill Hollow spring burned samples	0.781
Response Variable	Treatment		<i>P</i> -value
Organic Matter Nitrogen (ENR* lbs/A)	China Gulch fall burned samples	Hukill Hollow fall burned samples	0.945
Organic Matter Nitrogen (ENR lbs/A)	China Gulch spring burned samples	Hukill Hollow spring burned samples	0.806
Response Variable	Treatment		<i>P</i> -value
Phosphorus** (Weak Bray) (ppm)	China Gulch fall burned samples	Hukill Hollow fall burned samples	0.785
Phosphorus (Weak Bray) (ppm)	China Gulch spring burned samples	Hukill Hollow spring burned samples	0.362
Response Variable	Treatment		<i>P</i> -value
Potassium*** (ppm)	China Gulch fall burned samples	Hukill Hollow fall burned samples	0.314
Potassium (ppm)	China Gulch spring burned samples	Hukill Hollow spring burned samples	0.040
Response Variable	Treatment		<i>P</i> -value
Magnesium***** (ppm)	China Gulch fall burned samples	Hukill Hollow fall burned samples	0.046
Magnesium (ppm)	China Gulch spring burned samples	Hukill Hollow spring burned samples	0.002

Response Variable	Treatment		P-value
Calcium (ppm)	China Gulch fall burned samples	Hukill Hollow fall burned samples	0.384
Calcium (ppm)	China Gulch spring burned samples	Hukill Hollow spring burned samples °	0.045
Response Variable	Treatment		P-value
Sodium (ppm)	China Gulch fall burned samples	Hukill Hollow fall burned samples	0.305
Sodium (ppm)	China Gulch spring burned samples	Hukill Hollow spring burned samples	0.245
Response Variable	Treatment		P-value
Soil pH	China Gulch fall burned samples	Hukill Hollow fall burned samples °	0.092
Soil pH	China Gulch spring burned samples	Hukill Hollow spring burned samples	0.504
Response Variable	Treatment		P-value
Sulfur (ppm)	China Gulch fall burned samples	Hukill Hollow fall burned samples	0.195
Sulfur (ppm)	China Gulch spring burned samples	Hukill Hollow spring burned samples	0.320

▪ Indicates the group with the highest mean for China Gulch

° Indicates the group with the highest mean for Hukill Hollow

* ENR = estimated nitrogen release

** multiply the results in ppm (parts per million)

by 4.6 to convert to pounds per acre P_2O_5

*** multiply the results in ppm (parts per million)

by 2.4 to convert to pounds per acre K_2O

**** multiply the results in ppm (parts per million)

by 2 to convert to pounds per acre to the elemental form

APPENDIX G

MANTEL TEST RESULTS FOR CHINA GULCH AND HUKILL HOLLOW

CHINA GULCH 2005 pre-treatment versus 2007 post-treatment

Pre-Treatment 2005 (120 plots, 33 species) compared to Post-Treatment 2007 (120 plots, 45 species)	Standardized Mantel statistic = r	<i>P</i> -value
H_0 : No correlation between plant abundance or diversity between pre-treatment and post-treatment	0.433711	0.001

HUKILL HOLLOW 2005 pre-treatment versus 2007 post-treatment

Pre-Treatment 2005 (120 plots, 38 species) compared to Post-Treatment 2007 (120 plots, 54 species)	Standardized Mantel statistic = r	<i>P</i> -value
H_0 : No correlation between plant abundance or diversity between pre-treatment and post-treatment	0.154471	0.001

CHINA GULCH 2005 spring burn versus 2007 spring burn

Pre-Treatment 2005 spring burn (30 plots, 33 species) compared to Post-Treatment 2007 spring burn (30 plots, 45 species)	Standardized Mantel statistic = r	<i>P</i> -value
H_0 : No correlation between plant abundance or diversity between pre-treatment and post-treatment	0.263273	0.001

 CHINA GULCH 2005 fall burn versus 2007 fall burn

Pre-Treatment 2005 fall burn (30 plots, 33 species) compared to Post-Treatment 2007 fall burn (30 plots, 45 species)	Standardized Mantel statistic = r	<i>P</i> -value
H ₀ : No correlation between plant abundance or diversity between pre-treatment and post-treatment	0.126396	0.047

 HUKILL HOLLOW 2005 spring burn versus 2007 spring burn

Pre-Treatment 2005 spring burn (30 plots, 38 species) compared to Post-Treatment 2007 spring burn (30 plots, 54 species)	Standardized Mantel statistic = r	<i>P</i> -value
H ₀ : No correlation between plant abundance or diversity between pre-treatment and post-treatment	0.132986	0.096

 HUKILL HOLLOW 2005 fall burn versus 2007 fall burn

Pre-Treatment 2005 fall burn (30 plots, 38 species) compared to Post-Treatment 2007 fall burn (30 plots, 54 species)	Standardized Mantel statistic = r	<i>P</i> -value
H ₀ : No correlation between plant abundance or diversity between pre-treatment and post-treatment	0.097107	0.143

APPENDIX H

CHINA GULCH SITE PHOTOS



View from Spring Burn treatment block at China Gulch



Fall prescribed fire at China Gulch (2005)



Spring control treatment block at China Gulch (2005)



Fall prescribed fire at China Gulch (2005)



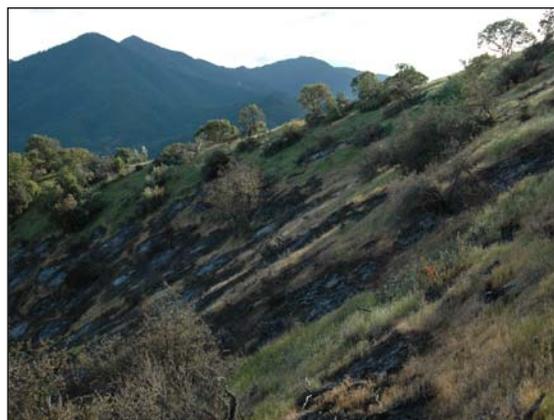
Spring burn treatment block at China Gulch (2005)



Fall prescribed fire at China Gulch (2005)



Fall prescribed fire at China Gulch (2005)



Spring prescribed fire at China Gulch (2006)



Sowing bunchgrass seeds following fall prescribed fire at China Gulch (2006)



Spring prescribed fire at China Gulch (2006)



Spring prescribed fire at China Gulch (2006)



Spring prescribed fire at China Gulch (2006)



Sowing bunchgrass seeds following spring prescribed fire at China Gulch (2006)



Vegetation response to spring prescribed fire at China Gulch (2006)



Gopher mound in fall treatment block at China Gulch (2008)



Verbascum thapsus and *Dichelostemma congestum* response to fall prescribed fire at China Gulch (2007)



Cirsium cymosum
response to fall
prescribed fire at
China Gulch (2008)



Cynoglossum grande response to fall
prescribed fire at China Gulch (2008)



Plagiobothrys nothofulvus
response to spring prescribed
fire at China Gulch (2008)



Quercus garryana
response to fall
prescribed fire
at China Gulch
(2007)



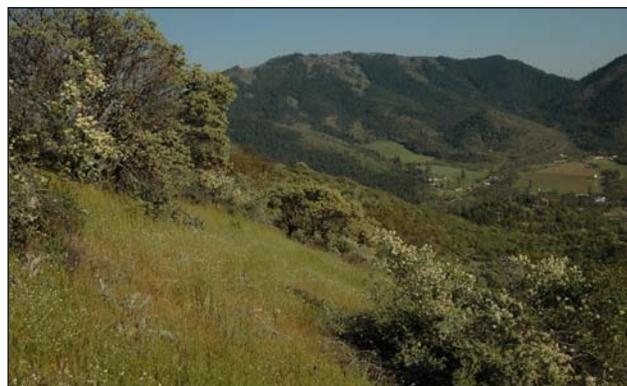
Amsinckia menziesii var. *intermedia*
response to spring and fall prescribed
fires at China Gulch (2008)



Bromus tectorum response to fall
prescribed fire at China Gulch (2008)



Spring burn treatment block at
China Gulch (2008)



Spring control treatment block at China Gulch
(2008)



Quercus garryana mortality in fall treatment block
at China Gulch (2008)

APPENDIX I

HUKILL HOLLOW SITE PHOTOS



View from spring control treatment block at Hukill Hollow



Fall prescribed fire at Hukill Hollow (2005)



Fall control treatment block at Hukill Hollow (2005)



Fall prescribed fire at Hukill Hollow (2005)



Slash left behind from brush mastication at Hukill Hollow (2005)



Fall prescribed fire at Hukill Hollow (2005)



Spring prescribed fire at Hukill Hollow (2006)



Fall prescribed fire at Hukill Hollow (2005)



Spring prescribed fire at Hukill Hollow (2006)



Germinants in seeded fall burn plot at Hukill Hollow (2006)



Spring prescribed fire at Hukill Hollow (2006)



Fall treatment block at Hukill Hollow (2008)



Garrya fremontii response to fall prescribed fire at Hukill Hollow (2008)



Quercus garryana response to fall prescribed fire at Hukill Hollow (2008)



Eriodictyon californicum response to fall prescribed fire at Hukill Hollow (2008)



Eriophyllum lanatum response to fall prescribed fire at Hukill Hollow (2008)



Prunus subcordata response to fall prescribed fire at Hukill Hollow (2008)



Toxicodendron diversilobum response to fall prescribed fire at Hukill Hollow (2008)



Bromus tectorum response to fall prescribed fire at Hukill Hollow (2008)



Woody species and pocket gopher response to brush mastication in fall control treatment block at Hukill Hollow (2008)



Arctostaphylos viscida response to brush mastication in spring control treatment block at Hukill Hollow (2008)



Quercus garryana mortality in fall treatment block at Hukill Hollow (2008)

APPENDIX J

PLANT SPECIES LIST FOR SAMPLED PLOTS

Taxon name: scientific name according to USDA Plants Database (2008) (* indicates plant observed outside plot); Family: plant family of taxon; Common Name: according to USDA Plants Database; Life Form: (pf) – perennial forb (includes biennial forbs), (af) – annual forb, (pg) – perennial grass, (ag) – annual grass, (w) – woody subshrub, shrub or tree; Non-Native: taxa classified as non-native according to USDA Plants Database; Treated As: taxa that were grouped with other species for analysis, or analyzed under a synonym.

Taxon Name	Family	Common Name	Study Sites		Life Form	Non-Native	Treated As
			China Gulch	Hukill Hollow			
<i>Achnatherum (Vasey) Barkworth lemmonii</i>	Poaceae	Lemmon's needlegrass	x	x	pg		
<i>Achillea millefolium</i> L.	Asteraceae	common yarrow		x	pf		
<i>Aira caryophylla</i> L.	Poaceae	silver hairgrass	x	x	ag	x	

<i>Amsinckia menziesii</i> (Lehm.) A. Nelson and J.F. Macbr. var. <i>intermedia</i> (Fisch. and C.A. Mey.) Ganders	Boraginaceae	common fiddleneck	x	x	af	
<i>Arctostaphylos viscida</i> Parry	Ericaceae	sticky whiteleaf manzanita	x	x	w	
<i>Arbutus menziesii</i> Pursh*	Ericaceae	Pacific madrone	x	x	w	
<i>Anthoxanthum aristatum</i> Boiss.	Poaceae	annual vernalgrass	x		ag	x
<i>Anthriscus caucalis</i> M. Bieb.	Apiaceae	bur chervil		x	af	x
<i>Avena fatua</i> L.	Poaceae	wild oat	x		ag	x
<i>Bombycilaena californica</i> (Fisch. and C.A. Mey.) Holub var. <i>californica</i>	Asteraceae	q-tips	x	x	af	<i>Micropus californicus</i>
<i>Bromus carinatus</i> Hook. and Arn.	Poaceae	California brome	x	x	pg	

<i>Bromus hordeaceus</i> L.	Poaceae	soft brome	x	x	ag	x	
<i>Bromus arvensis</i> L.	Poaceae	field brome		x	ag	x	<i>Bromus japonicus</i>
<i>Bromus rubens</i> L.	Poaceae	red brome	x	x	ag	x	<i>Bromus madritensis</i>
<i>Bromus tectorum</i> L.	Poaceae	cheat grass	x	x	ag	x	
<i>Calochortus tolmiei</i> Hook. and Arn.	Liliaceae	Tolmie startulip	x	x	pf		
<i>Calystegia occidentalis</i> (A. Gray) Brummitt	Convolvulaceae		x	x	pf		
<i>Castilleja attenuata</i> (A. Gray) T.I. Chuang and Heckard	Scrophulariaceae	attenuate Indian paintbrush			af		

<i>Cercocarpus betuloides</i> Nutt.	Rosaceae	mountain mahogany		x	w
<i>Ceanothus cuneatus</i> (Hook.) Nutt	Rhamnaceae	buckbrush	x	x	w
<i>Ceanothus integerrimus</i> Hook. and Arn.	Rhamnaceae	deerbrush		x	w
<i>Centaurea solstitialis</i> L.	Asteraceae	yellow starthistle	x		af x
<i>Cirsium cymosum</i> (Greene) J.T. Howell	Asteraceae	peregrine thistle	x	x	pf
<i>Clarkia purpurea</i> (W. Curtis) A. Nelson and J.F. Macbr. ssp. <i>quadrivulnera</i> (Douglas ex Lindl.) F.H. Lewis and M.E. Lewis	Onagraceae	winecup clarkia	x	x	af
<i>Clarkia rhomboidea</i> Douglas ex Hook.	Onagraceae	diamond clarkia	x	x	af

<i>Collomia grandiflora</i> Douglas ex Lindl.	Polemoniaceae	grand collomia	x	x	af	
<i>Collinsia linearis</i> A. Gray	Scrophulariaceae	narrowleaf blue eyed Mary		x	af	
<i>Collinsia parviflora</i> Lindl.	Scrophulariaceae	maiden blue eyed Mary	x	x	af	
<i>Conyza canadensis</i> (L.) Cronquist *	Asteraceae	Canadian horseweed		x	af	x
<i>Cryptantha flaccida</i> (Douglas ex Lehm.) Greene	Boraginaceae	weakstem cryptantha	x	x	af	
<i>Claytonia perfoliata</i> Donn ex Willd.	Portulacaceae	miner's lettuce	x	x	af	<i>Claytonia</i> sp.
<i>Cynosurus echinatus</i> L.	Poaceae	bristly dogstail grass	x	x	ag	x
<i>Cynoglossum grande</i> Douglas ex Lehm.	Boraginaceae	Pacific hound's tongue	x	x	pf	

<i>Daucus pusillus</i> Michx.	Apiaceae	American wild carrot	x	x	af	
<i>Delphinium menziesii</i> DC.*	Ranunculaceae	Menzies' larkspur	x	x	pf	
<i>Dichelostemma congestum</i> (Sm.) Kunth	Liliaceae	ookow	x		pf	
<i>Elymus glaucus</i> Buckley	Poaceae	blue wildrye	x	x	pg	
<i>Erodium cicutarium</i> (L.) L'Hér. ex Aiton	Geraniaceae	redstem stork's bill	x	x	af	x
<i>Eriodictyon californicum</i> (Hook. and Arn.) Torr. *	Hydrophyllaceae	California yerba santa	x	x	w	
<i>Eriophyllum lanatum</i> (Pursh) Forbes	Asteraceae	common woolly sunflower	x	x	w	

<i>Eschscholzia californica</i> Cham. *	Papaveraceae	California poppy			af	
<i>Festuca californica</i> Vasey	Poaceae	California fescue		x	pg	
<i>Festuca occidentalis</i> Hook.	Poaceae	western fescue		x	pg	
<i>Festuca idahoensis</i> Elmer ssp. <i>roemeri</i> (Pavlick) S. Aiken	Poaceae	Roemer's fescue	x	x	pg	
<i>Fritillaria recurva</i> Benth.	Liliaceae	scarlet fritillary		x	pf	
<i>Galium ambiguum</i> W. Wight	Rubiaceae	Yolla Bolly bedstraw	x	x	pf	
<i>Galium aparine</i> L.	Rubiaceae	stickywilly	x	x	af	
<i>Galium parisiense</i> L.	Rubiaceae	wall bedstraw	x	x	af	x

<i>Garrya fremontii</i> Torr.	Garryaceae	bearbrush	x	x	w	
<i>Gayophytum diffusum</i> Torr. and A. Gray	Onagraceae	spreading ground-smoke	x	x	af	
<i>Gilia capitata</i> Sims	Polemoniaceae	bluehead gilia	x	x	af	
<i>Hypericum perforatum</i> L.	Hypericaceae	common St. Johnswort	x	x	pf	x
<i>Koeleria macrantha</i> (Ledeb.) Schult.	Poaceae	prairie Junegrass		x	pg	
<i>Lactuca serriola</i> L.	Asteraceae	prickly lettuce		x	af	x
<i>Leptosiphon bicolor</i> Nutt.	Polemoniaceae	true babystars	x	x	af	<i>Linanthus bicolor</i>
<i>Lonicera hispidula</i> (Lindl.) Douglas ex Torr. and A. Gray	Caprifoliaceae	pink honeysuckle	x	x	w	

<i>Lonicera interrupta</i> Benth.	Caprifoliaceae	chaparral honeysuckle	x		w	
<i>Lotus micranthus</i> Benth.	Fabaceae	desert deervetch	x	x	af	
<i>Madia citriodora</i> Greene	Asteraceae	lemon-scented madia	x	x	af	<i>Madia</i> sp.
<i>Madia exigua</i> (Sm.) A. Gray	Asteraceae	small tarweed	x	x	af	<i>Madia</i> sp.
<i>Madia gracilis</i> (Sm.) D.D. Keck	Asteraceae	grassy tarweed	x	x	af	<i>Madia</i> sp.
<i>Madia madioides</i> (Nutt.) Greene	Asteraceae	woodland madia	x	x	af	<i>Madia</i> sp.
<i>Monardella odoratissima</i> Benth.	Lamiaceae	mountain monardella	x	x	pf	
<i>Nemophila parviflora</i> Douglas ex Benth.	Hydrophyllaceae	smallflower nemophila		x	af	

<i>Orobanche fasciculata</i> Nutt.*	Orobanchaceae	clustered broomrape	x	x	af	
<i>Penstemon deustus</i> Douglas ex Lindl.	Scrophulariaceae	scabland penstemon	x		pf	
<i>Phacelia heterophylla</i> Pursh	Hydrophyllaceae	varileaf phacelia		x	pf	
<i>Pinus ponderosa</i> C. Lawson*	Pinaceae	ponderosa pine			w	
<i>Plagiobothrys nothofulvus</i> (A. Gray) A. Gray	Boraginaceae	rusty popcorn- flower	x	x	pf	
<i>Plectritis congesta</i> (Lindl.) DC.	Valerianaceae	shortspur seablush		x	af	
<i>Poa bulbosa</i> L.	Poaceae	bulbous bluegrass	x	x	pg	x
<i>Potentilla glandulosa</i> Lindl.	Rosaceae	sticky cinquefoil		x	af	

<i>Prunus subcordata</i> Benth.	Rosaceae	Klamath plum		x	w	
<i>Quercus garryana</i> Douglas ex Hook.	Fagaceae	Garry oak	x	x	w	
<i>Sanicula graveolens</i> Poepp. ex DC.	Apiaceae	northern sanicle	x	x	pf	
<i>Scutellaria tuberosa</i> Benth.*	Lamiaceae	Danny's skullcap	x		pf	
<i>Sidalcea malviflora</i> (DC.) A. Gray ex Benth. *	Malvaceae	dwarf checker-bloom	x		pf	
<i>Silene hookeri</i> Nutt. *	Caryophyllaceae	Hooker's silene	x	x	pf	
<i>Sisymbrium altissimum</i> L.	Brassicaceae	tall tumble-mustard		x	af	x
<i>Sonchus asper</i> (L.) Hill	Asteraceae	spiny sowthistle		x	af	x

<i>Stellaria media</i> (L.) Vill.	Caryophyllaceae	common chickweed	x	x	af	x
<i>Stephanomeria virgata</i> Benth.	Asteraceae	rod wirelettuce	x	x	af	
<i>Taeniatherum caput-medusae</i> (L.) Nevski	Poaceae	medusahead	x		ag	x
<i>Thysanocarpus curvipes</i> Hook.	Brassicaceae	sand fringepod	x	x	af	
<i>Toxicodendron diversilobum</i> (Torr. and A. Gray) Greene	Anacardiaceae	Pacific poison oak	x	x	w	
<i>Torilis nodosa</i> (L.) Gaertn.	Apiaceae	knotted hedge-parsley	x	x	af	x
<i>Tonella tenella</i> (Benth.) A. Heller	Scrophulariaceae	lesser baby innocence	x	x	af	
<i>Tragopogon dubius</i> Scop.	Asteraceae	yellow salsify	x	x	af	x

<i>Triteleia hendersonii</i> Greene*	Liliaceae	Henderson's triteleia	x		pf	
<i>Verbascum thapsus</i> L.	Scrophulariaceae	common mullein	x	x	pf	x
<i>Verbena lasiostachys</i> Link*	Verbenaceae	western vervain	x		pf	
<i>Vulpia microstachys</i> (Nutt.) Munro	Poaceae	small fescue	x	x	ag	
<i>Vulpia myuros</i> (L.) C.C. Gmel.	Poaceae	rat-tail fescue	x	x	ag	x