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MONTANE VALLEY GRASSLAND PLANT COMMUNITIES ARE HIGHLY RESISTANT TO WILDFIRE

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BY

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THESIS

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ABSTRACT

Understanding the ecological role of fire in fire-adapted plant communities is of great importance for restoration and preservation; however, limited research has been conducted on the response of upper elevation, C_3 grassland plant communities to wildfire. This study investigates the effects of the Las Conchas wildfire of 2011 on plant community structure and function in the montane valley grasslands of the Valles Caldera National Preserve, Jemez Mountains, New Mexico, USA. Long term monitoring of nine burned and seven unburned grassland sites was used to measure vegetation composition and dynamics both spatially and temporally relative to fire. Results show that these fire-adapted plant communities are highly resilient to fire; fire had no significant effects on composition or structure beyond the normal range of variability. Instead, climatic drivers had the greatest influence on plant community dynamics over time. These results support land and fire management efforts to restore historic fire regimes which will help maintain grassland integrity and resilience in the face of a changing climate.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

INTRODUCTION

Ecological structure and functioning of montane valley grasslands are of critical biological and economic importance for these grasslands are a vital source of forage and habitat for wildlife, and they contribute to water quality and larger scale watershed function. Furthermore, montane grasslands are impacted by anthropogenic disturbances such as livestock grazing (VanAuken 2009) and development (York 2000) which has led to the threat and decline of these ecosystems in western North America. High elevation grasslands of the southwestern U.S. range in size but can occur in valleys, slopes, and ridges within subalpine conifer forests and possess properties unsuitable for tree growth (Brown 1994). In addition to factors such as temperature, moisture, and soil type, fire is a huge component in maintaining grasslands by resisting woody plant encroachment (Allen 1989; Coop 2007a; VanAuken 2009).

Fire history of the Southwestern United States shows fire to be an essential ecological component in maintaining the health and proper functioning of forests across the region by maintaining community structure, ecosystem functioning, and species diversity via fuel removal and the release of organically stored nutrients back into the environment (Dewar 2011). However, in the past 3,000 years there has been a decline in fire frequency due to anthropogenic induced fire suppression (Marlon et al. 2012) and as a result, many forest types in the southwestern region are undergoing changes in forest structure and fuel accumulation which then drives wildfires burning at increasing severities; the frequency of these more intense wildfires is also increasing due to climate change (Hurteau et al. 2014). Since montane grasslands are interspersed throughout mountain ranges within forest stands it is important to understand fire regimes on a landscape level in order to determine their impacts in more localized habitats. Research conducted on fire history of montane grasslands for the Valles Caldera National Preserve (VALL) shows that prior to 1900, fire occurred on average every 1.6 years somewhere on the Preserve's grasslands, with widespread fires occurring at decadal intervals and smaller low intensity fires, particularly in valley grasslands, occurring every 5.5 to 22.5 years (Dewar 2011). For the Preserve, fire has been the single most important factor in preventing forest encroachment and maintaining grassland communities, but as a result of anthropogenic induced fire suppression nearly 18% (between 1935 and 1996) of grassland area has been lost to tree encroachment (Coop 2007a). It is therefore of utmost importance to understand the role of fire in the ecosystem functioning of these grassland systems in order to restore and preserve their quality on large temporal and spatial scales.

Limited research has been conducted on the response of upper elevation C_3 grassland plant communities to wildfire. Most grassland studies of fire impacts focus on C_4 dominated rangelands at lower elevation with lower average annual precipitation. One study of post-fire succession in a subalpine meadow (Debenedetti & Parsons 1984) was conducted in a wet meadow with moisture levels and corresponding plant community composition that differed substantially from the drier grassland slopes of this study. Also lacking has been a long term study of the effects of wildfire with knowledge of pre-fire conditions and post-fire succession beyond the first 2 or 3 growing seasons following fire. Furthermore, studies on the fire ecology of the plant species present in this system have been studied from the herbaceous communities in forest understories rather than open grassland ecosystems.

This study investigates the effects of the Las Conchas wildfire of 2011 on plant community structure and function in the montane valley grasslands of the Valles Caldera National Preserve. Effective management, restoration, and preservation of montane grasslands is dependent on understanding historical fire regimes, impacts of burns and recovery, and the role climate plays in the process. Accordingly, the objectives of this study are to (1) determine how fire adapted grassland plant communities respond to wildfire by exploring how plant species composition, abundance, and distribution vary spatially and temporally; (2) determine what biotic and abiotic factors are most influential in stabilizing and/or driving changes in the years following a grassland burn; and (3) assess the ecological function of these grassland systems before and after fire with respect to forage and habitat quality and occurrence of non-native and invasive species. Implications for fire and land management practices based on the findings of this study are discussed.

METHODS

Site Description

The Valles Caldera National Preserve, located in the heart of the Jemez Mountains in north-central New Mexico, consists of high elevation forest and grassland systems that lie at the head of the Jemez River watershed which flows into the Rio Grande Valley. The 36,017 ha Preserve is centered on a volcanic caldera of a supervolcano that erupted 1.2 million years ago (Spell et al. 1993) and is comprised of large areas of low-lying mountain meadows and riparian vegetation with sloped mountain valley grasslands bordered by several forested lava domes ranging in elevation from 2,590 – 3,505m. The Valles Caldera is home to an estimated 10,522 ha of montane grassland dominated by C_3 grasses, particularly perennial bunch grasses such as *Festuca*, *Poa*, and *Muhlengbergia* which commonly occur in subalpine grasslands across the southwestern U.S. (Brown 1994), and represents some of the largest and highest quality occurrences with respect to ecological function and biodiversity within the Southern Rocky Mountains Ecoregion (Muldavin 2003). The grasslands of the Valles Caldera have formed on pre-historic lake beds that are between 70,000 and 500,000+ years old whose soils consist primarily of Mollisols that have developed from older quaternary alluvial fan (Qf) deposits from the surrounding domes (Muldavin 2003).

The Preserve has undergone extensive livestock grazing during the 19th and 20th centuries but currently supports only a relatively small (< 400 head) herd of cattle and a population of approximately 2,000 elk (*Cervus canadensis*) native to the region. Mean annual precipitation derived from the 2004-2015 water years (October-September) is 545 mm; 60% of this precipitation is produced by monsoonal rainfall between the months of

June-September and winter precipitation is primarily in the form of snowpack (Muldavin 2003).

Fire Description

On June 26, 2011 the Las Conchas Wildfire was ignited south of the Preserve by a downed power line at the height of extreme fire danger and spread to consume 63,131 ha of forest and grassland vegetation which at the time was the largest wildfire recorded in New Mexico history. A total of 12,141 ha of the burn occurred within Preserve boundaries, 25% of which was grassland. The burn intensity varied from forest to grassland from high severity stand-replacing in the forest to low intensity ground fire in the open grasslands. The grassland sites in this study burned over the course of five days from June 27- July 1 with a wind speed of 2 meters per second and an average relative humidity of 36%. Average air temperature for this time period was 18°C; minimum and maximum temperatures were within a 2°C range of the average. Data describing rate of spread and flame length were not available but anecdotal evidence identified two mechanisms of fire spread – head fire and backing fire, which occurred at different locations. Post-fire vegetation burn assessments by Preserve staff found exposed bare ground increased significantly ($P = 0.002$) from 0.9% in 2010 to 3.4% in 2011 after the fire and total herbaceous cover dropped significantly ($P = 0.02$) from 96% in 2010 to 93% in 2011.

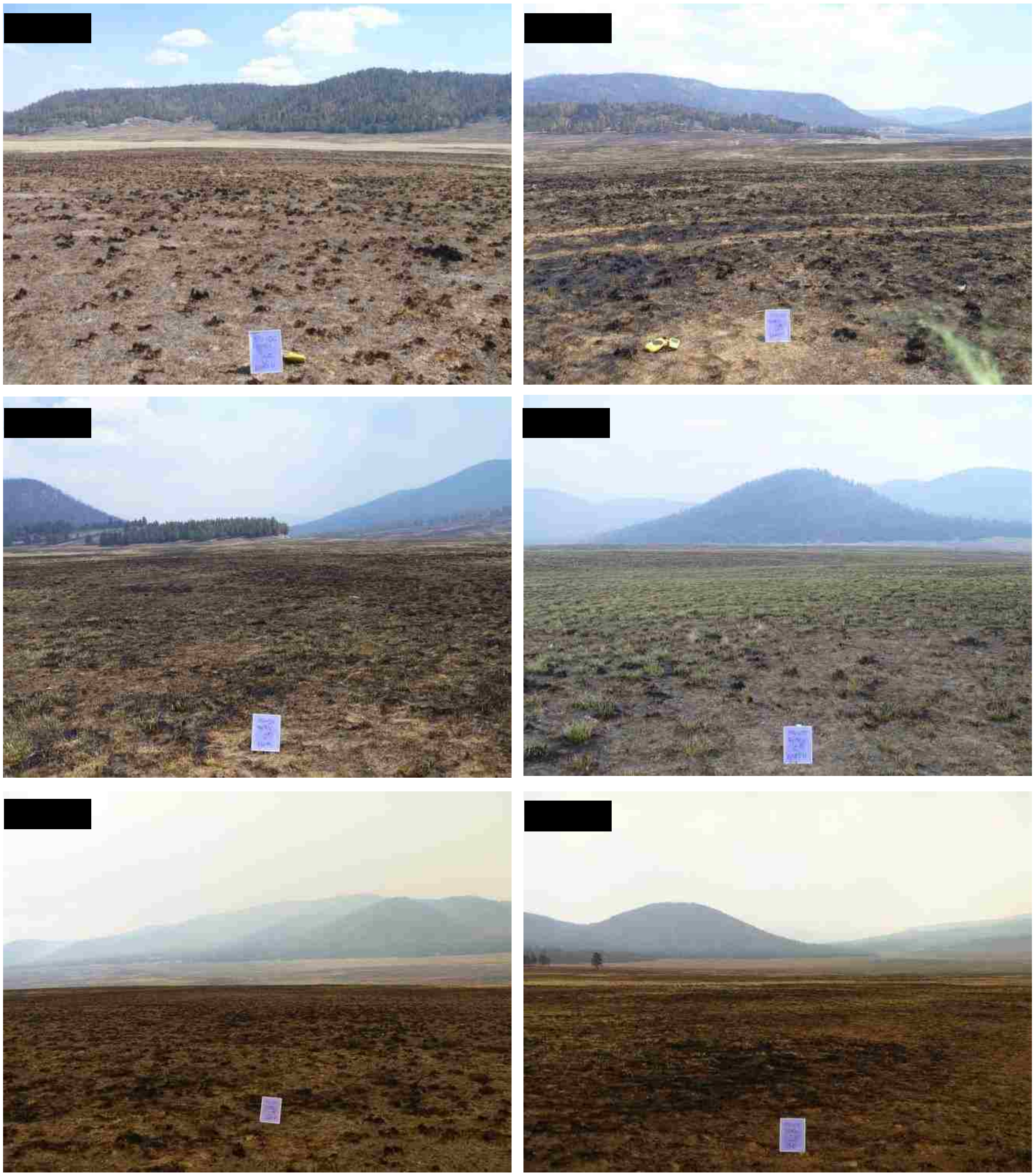


Figure 1. Post-burn photos of six different sampling sites taken in July 2011, less than one month after the June 26, 2011 Las Conchas fire.

Field Data Collection

With the federal acquisition of the Preserve in 2000, the Jornada Rangeland Research Program of New Mexico established a rangeland monitoring program consisting of 44 vegetation monitoring sites starting in 2001 (Barnes 2002). These sites were stratified by relatively homogenous repeating ecotypes, which were identified by soil type, land-form, floral community composition, management history, and future management potential. Three ecotypes were found to occur in patterns across the Preserve: Mountain Meadow, Mountain Valley, and Grazeable Woodland (Barnes 2002). For our analysis of fire effects on grassland communities, we used 16 monitoring sites from the Mountain Valley (MV) ecotype (Fig. 2).

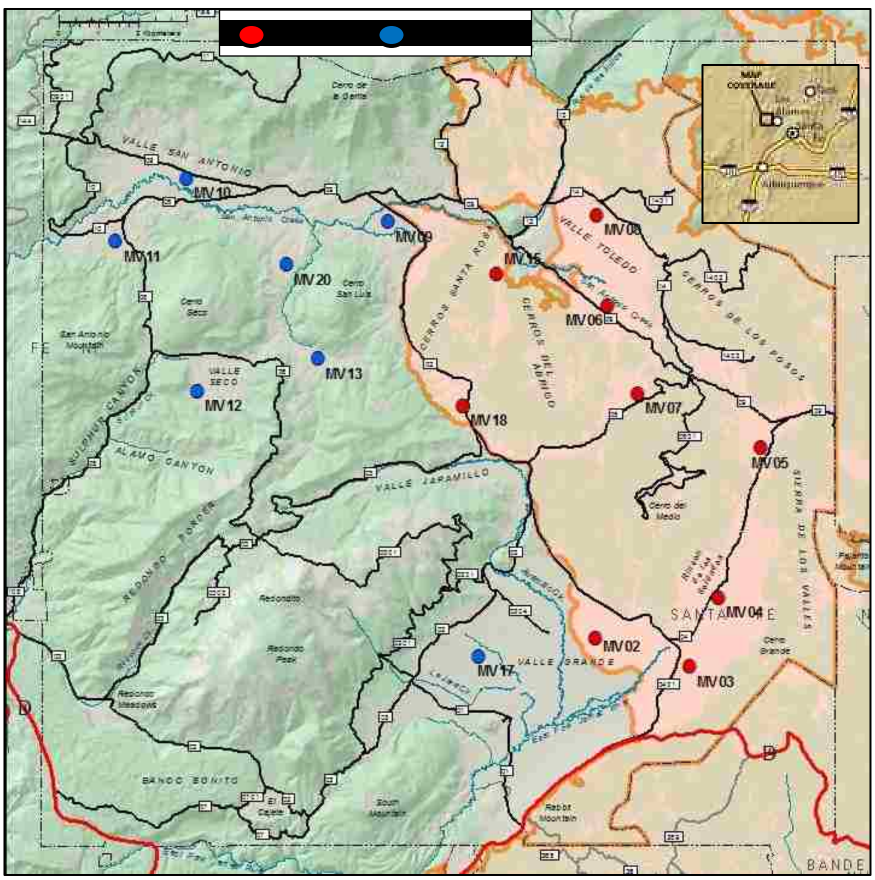


Figure 2. Distribution of Mountain Valley grassland study sites in the Valles Caldera National Preserve, New Mexico with Las Conchas Fire burn area. Note that burned sites are located on the eastern side in the burned zone.

The MV ecotype is located on the upper slope margins of the valleys, which consists of open grasslands dominated by native bunch grasses, particularly *Festuca arizonica* and *Danthonia parryi* (Barnes 2002); downslope, the grasslands grade to wet meadows and herbaceous wetlands and upslope to mixed-coniferous forests. The MV ecotype is composed of two different soil types: Tranquilar-Jarmillo complex (series 302) silt clay loam/loam to 33 and 91 cm depth, respectively; Cosey-Jarmillo association (series 304) silt loam to 38 cm (NCSS 1987). Of available MV sites, nine were burned in the Las Conchas Fire of 2011 and another seven unburned sites were selected to serve as controls for a total of 16 sites (Fig. 2).

This sampling protocol was established by the Jornada Rangeland Research Program of New Mexico as stated in Barnes (2002). Each monitoring site consisted of three, 100-meter transect lines radiating outward from a central location at zero, 120, and 240 degrees. The line-point intercept sampling method was used to identify and measure plant species present at each meter for a total of 100 points per line and 300 points per site. Measurements were taken with the use of a thin steel rod or dowel, 1.2 meters in length by 1 cm in diameter. The species of every live plant touching the rod, or intersecting the vertical line drawn by the rod from the top of the plant canopy down to the soil surface, were recorded to species level in order of appearance. If one species occurred more than once at a particular point, only its highest appearance was recorded. Canopy height measurements, estimated to the nearest centimeter, were recorded as the height of the point at which the tallest plant intersected the sampling rod. Where there was no live canopy, the height was recorded as 0 cm. Two photos were taken of each transect line with the measuring tape present on the ground for a total of at least six photographs per site per year. One photo was taken from the transect starting point facing the end point (denoted as photo point "A") and another taken from the terminal end of the transect looking back toward the center post (denoted as photo point "B"). Photo monitoring and data collection were conducted annually during the growing season (June-Sept.). Long term rangeland monitoring of these grassland sites was conducted from 2001 until present, although not all sites were sampled in all years (Table 1). Sampling responsibility was transferred from Will Barnes to Preserve staff in 2011; my participation in field sampling began in 2013 and continued through 2015.

Annual standing crop biomass was measured by collecting herbaceous vegetation at each sampling site using four replicate ¼ m^2 rings randomly placed inside ungulate grazing exclosures approximately 2 x 2 meters in size. All biomass material was dried at 60°C for at least 48 hours and weighed to the nearest gram; collections were made annually in the Fall.

Table 1. Site characteristics for all monitoring plots.

Site	Treatment	Total Years Grazed	Years grazed pre-fire	Years grazed post- fire	Years NOT Sampled	Soil Series*	Aspect (°)	Slope %	Elevation (m)	Dominant Grass**
MV02	BURN	6	6	0	1	302	211	2	2607	FESX1, MUMO, POPR
MV03	BURN	6	6	0	3	308	181	5	2611	FESX1, MUMO, POPR
MV04	BURN	7	3	4	1	302	153	4	2623	DAPA, POPR, FESX1
MV05	BURN	6	2	4	2	304	68	5	2734	DAPA, POPR, FESX1
MV06	BURN	5	5	0	1	302	45	3	2656	DAPA, FESX1
MV07	BURN	6	6	0	5	304	328	8	2692	DAPA, FESX1
MV08	BURN	5	5	0	1	304	198	8	2702	DAPA, FESX1
MV09	CONTROL	6	6	0	1	302	168	2	2585	FESX1, MUMO
MV10	CONTROL	6	6	0	2	302	245	4	2559	FESX1, MUMO
MV11	CONTROL	11	6	5	3	302	269	11	2574	FESX1, POPR
MV12	CONTROL	11	6	5	2	304	62	6	2685	DAPA, POPR, FESX1
MV13	CONTROL	11	6	5	3	304	27	4	2662	DAPA, POPR, FESX1
MV15	BURN	12	7	5	4	304	67	8	2650	DAPA, POPR, FESX1
MV17	CONTROL	13	8	5	2	302	123	3	2608	POPR
MV18	BURN	12	7	5	5	302	50	7	2670	DAPA, POPR, FESX1
MV20	CONTROL	11	6	5	4	304	36	5	2614	DAPA, POPR

* Soil Series taken from USDA NRCS Soil Survey of Sandoval County Area, New Mexico
** Dominant plant species by abundance. DAPA = *Danthonia parryi*; FESX1 = *Festuca arizonica* and *Festuca idahoensis* consolidated; MUMO = *Muhlenbergia montana*; POPR = *Poa pratensis*

Statistical Analysis

Assessments within and between two treatment groups (burn and control), before and after the 2011 fire, were used to determine spatial and temporal response of grassland communities to fire. This investigation took place over a 15 year study period, 10 years pre-fire and 5 years post-fire. Abundance data for all plant species encountered at each monitoring site for all years sampled was aggregated and transformed to the 4^{th} root in order to achieve the highest level of normality possible. We used non-metric multidimensional scaling (NMDS) with a Bray-Curtis similarity index to display spatial variability in species composition among sites and treatments, post-burn trajectories for each site, and species distribution of four dominant species. We then used permutational multivariate ANOVA (PERMANOVA) with a Bray-Curtis similarity index to make pairwise comparisons between 3 treatment types – *pre-fire, post-fire burned group, and post-fire control grou*p, for the following comparisons: pre-fire vs post-fire among burned sites, pre-fire vs post-fire among unburned sites, and post-fire burn vs post-fire unburned sites. We then used SIMPER to classify each monitoring site based on the top four most abundant species present across all sites. SIMPER was also used to identify which species contributed most to compositional differences within sites over time and between treatment groups. The species with the greatest contribution to compositional differences were determined to be those in the top 15% cumulative contribution range. All multivariate analyses were conducted using Primer 6 (PRIMER-E, Plymouth, UK).

Repeated Measures ANOVA was used to test differences in plant canopy height, species richness, species evenness, and grass and forb abundance between the burn and control groups for all years sampled using the treatment and year interaction term. We also used a randomized block design (RBD) for two treatment types – *before* (2001-2010) and *after* (2011-2015) fire in order to identify temporal changes in plant canopy height, species richness, species evenness, and grass and forb abundance. A randomized block design is often used instead of a completely randomized design in studies where there is extraneous variation among the experimental units that may influence the response. A significant amount of variation may be removed from the comparison of treatments by partitioning the experimental units into fairly homogeneous subgroups or blocks, in this case each site, with sampling years treated as replicates, was treated as a separate block. Lastly, resilience was measured using $ln(ASB_{201X}/aveASB_{2002\text{-}2010})$ (Tilman & Downing 1994) with significance testing between each post-fire year and the 10 year (pre-fire) annual standing biomass average.

In order to determine the influence of biotic and abiotic drivers on plant community structure, we used simple linear regression to correlate monsoon seasonal precipitation accumulation over the months of June and July prior to the sampling with plant canopy height, species richness, species evenness, and abundance of grass and forb functional types for all sites combined. For this correlation test all sites were included initially in order to determine if precipitation was influencing both treatments equally; if there had not been a significant correlation we would have partitioned out the sites based on treatment group and run the regression analysis for each treatment individually. Precipitation data were acquired from four different meteorological stations located throughout the Preserve. Each site was paired with the nearest weather station, rather than averaging precipitation data for all four weather stations, in order to reduce variability and capture the best representation possible of weather conditions at each site.

Furthermore, the influence of plant litter in terms of percentage ground cover was correlated to species evenness over time for all sites combined using Spearman ranked correlation.

Taxonomic Considerations

Some closely related species present at these monitoring sites have very similar morphology and as such pose challenges in identifying them correctly in the field. Accordingly, we found some inconsistency in the identification of certain species over the years sampled. In order to mitigate this we aggregated the abundance of like species and treated their presence at the Genus level. *Antennaria parvifolia*, *Antennaria rosea*, and *Antennaria rosulata* were combined due to overlapping leaf sizes and the absence of an inflorescence at time of sampling. *Festuca arizonica* and *Festuca idahoensis* were combined due to inconsistent identification throughout sampling years; at each site predominantly one of the two species was recorded for a given year but would then alternate in different sampling years for the same site. *Bromus anomalus* and *Bromus porteri* were combined also due to inconsistent differentiation as a result of overlapping sizes of various morphological characteristics. All *Carex* species, approximately 15 different species, were combined due to challenges in identifying them vegetatively because most were not in flower or seed set at the time of sampling.

RESULTS

We found a total of 157 different plant species with grasses accounting for 22% of total species richness and half of the total percent cover, while forbs account for 62% of species richness but only 27% of total canopy cover. PERMANOVA testing of site homogeneity consisted of 120 different site pairwise comparisons which revealed that species composition among all 16 sites was significantly different prior to the fire; therefore, these sites were not true replicates as was initially assumed. However, 10 site pairs out of the 120 pairwise comparisons (8%) were not significantly different following the fire and these homogenous pairs consisted of sites from both the control and burned treatment groups.

Immediate post-burn effects were tested by comparing community composition between years 2010 and 2011 using PERMANOVA. Significant differences occurred between pre-fire and the post-fire burn treatment group, but not for the unburned sites (Table 2, A). Based on SIMPER, the differences between pre- and post-burn were driven by a large decline in the abundance of *Agrostis scabra*, a small increase in *Danthonia parryi*, and a large decrease in both *Trifolium repen*s and *Poa fendleriana* (Table 2, A). Based on comparisons for years 2010 and 2012, however, these differences were no longer significant (primarily because *Agrostis scabra* had fully recovered). But there were significant differences between the pre-fire and post-fire unburned control group, indicating an effect of factors other than fire (Table 2, B). This dissimilarity was shown to be driven by most of the same species that contributed to the difference between pre-fire and post-fire burn; there was a relatively large decline in the average abundance of

Danthonia parryi, *Trifolium repen*s, and *Agrostis scabra* in addition to an average increase in *Bouteloua gracilis* for 2012.

Table 2. PERMANOVA and SIMPER results of pairwise comparisons for plant community assemblages in burned and control treatment groups, before and after fire for multiple temporal periods. Abundance values are based on data transformation to the 4th root. Comparisons with significance are highlighted.

Pairwise Comparisons: Community	PERMANOVA				PERDISP*		
A: Pre(2010)-Post(2011)	***n***	***t***	***p***	**Permutations**	***t***	***p***	**Permutations**
Pre-Fire , Post-Fire.BURN	16, 9	1.7	0.002	999	0	0.9	999
Pre-Fire, Post-Fire.CONTROL	16, 7	1.3	0.1	996	0	0.7	999
Post-Fire.BURN, Post-Fire.CONTROL	9, 7	1.2	0.1	963	0	0.8	999

SIMPER: Pre-Fire , Post-Fire.BURN	**Pre-Fire**	**Post-Fire.BURN**				
Species	**Av.Abund**	**Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Danthonia parryi	2.36	2.59	1.47	1.09	4.23	4.23
Agrostis scabra	1.2	0	1.12	1.57	3.23	7.47
Trifolium repens	1.4	0.45	1.08	1.51	3.12	10.58
Poa fendleriana	1.57	0.85	1.08	1.33	3.11	13.7

Pairwise Comparisons: Community	PERMANOVA				PERDISP*		
B: Pre(2010)-Post(2012)	***n***	***t***	***p***	**Permutations**	***t***	***p***	**Permutations**
Pre-Fire , Post-Fire.BURN	16, 7	1.3	0.1	997	0.4	0.7	999
Pre-Fire, Post-Fire.CONTROL	16, 7	1.4	0.02	996	1.1	0.3	999
Post-Fire.BURN, Post-Fire.CONTROL	7, 7	0.7	0.9	744	1.3	0.2	999

SIMPER: Pre-Fire, Post-Fire.CONTROL	**Pre-Fire**	**Post-Fire.CONTROL**				
Species	**Av.Abund**	**Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Danthonia parryi	2.36	1.83	1.62	1.27	4.59	4.59
Trifolium repens	1.4	0.29	1.17	1.54	3.32	7.91
Bouteloua gracilis	0.44	1.14	1.06	1.17	3.02	10.93
Agrostis scabra	1.2	0.34	1.02	1.41	2.89	13.82

Pairwise Comparisons: Community C: Pre(2001-2010)-Post(2011-2015)	PERMANOVA				PERDISP*		
	n	*t*	*p*	Permutations	*t*	*p*	Permutations
Pre-Fire , Post-Fire.BURN	132, 40	2.5	0.001	998	0	0.8	999
Pre-Fire, Post-Fire.CONTROL	132, 28	1.8	0.005	997	1	0.2	999
Post-Fire.BURN, Post-Fire.CONTROL	40, 28	1.6	0.01	999	2	0.009	999

SIMPER: Pre-Fire , Post-Fire.BURN	**Pre-Fire**	**Post-Fire.BURN**				
Species	**Av.Abund**	**Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Danthonia parryi	2.18	2.65	1.54	1.12	4.23	4.23
Elymus elymoidies	1.15	1.37	1.02	1.37	2.81	7.04
Poa fendleriana	1.43	0.92	0.99	1.28	2.72	9.76
moss species	1.25	0.8	0.97	1.28	2.66	12.42
Taraxacum officinale	1.27	1.55	0.93	1.22	2.57	14.99

SIMPER: Pre-Fire, Post-Fire.CONTROL	**Pre-Fire**	**Post-Fire.CONTROL**				
Species	**Av.Abund**	**Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Danthonia parryi	2.18	1.9	1.59	1.21	4.31	4.31
Bouteloua gracilis	0.47	1.01	1	1	2.7	7.01
Poa fendleriana	1.43	1.08	0.96	1.23	2.59	9.6
moss species	1.25	0.68	0.95	1.26	2.58	12.19
Elymus elymoidies	1.15	1.37	0.85	1.31	2.31	14.49

SIMPER: Post-Fire.BURN, Post-Fire.CONTROL	**Post-Fire.BURN**	**Post-Fire.CONTROL**				
Species	**Av.Abund**	**Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
Danthonia parryi	2.65	1.9	1.61	1.22	4.35	4.35
Elymus elymoidies	1.37	1.37	0.97	1.4	2.61	6.96
Bouteloua gracilis	0.03	1.01	0.9	0.91	2.44	9.41
Poa fendleriana	0.92	1.08	0.89	1.17	2.4	11.81
Symphyotricum ascendens	0.9	0.84	0.84	1.25	2.28	14.09

*PERDISP is a permutational dispersion test, i.e. test of variability or spatial spread between groups.

The long term study of these grasslands allowed for the measure of fire impacts on a broader, multi-year temporal scale. When all 16 sampling sites for the entire study period were grouped according to treatment, all pairwise comparisons were found to be significantly different (Table 2, C). The significance between pre-fire conditions to not only the post-fire burn group but also to the unburned control group is of importance. SIMPER showed the greatest contributors to the difference between the pre-fire and post-fire burn group to be *Danthonia parryi*, *Elymus elymoides*, and *Taraxacum officinale* which all increased in abundance on average along with average declines in *Poa fendleriana* and moss species. The significance between pre-fire and the post-fire control group was explained by an average decrease in *Danthonia parryi*, *Poa fendleriana* and moss species as well as an average increase in *Bouteloua gracilis* and *Elymus elemoides.*

Overall, differences within sites pre and post-fire were much smaller than differences among sites but there was no consistent direction or degree of dispersion in NMDS multi-variate space after fire within the two treatment groups (Fig. 3).

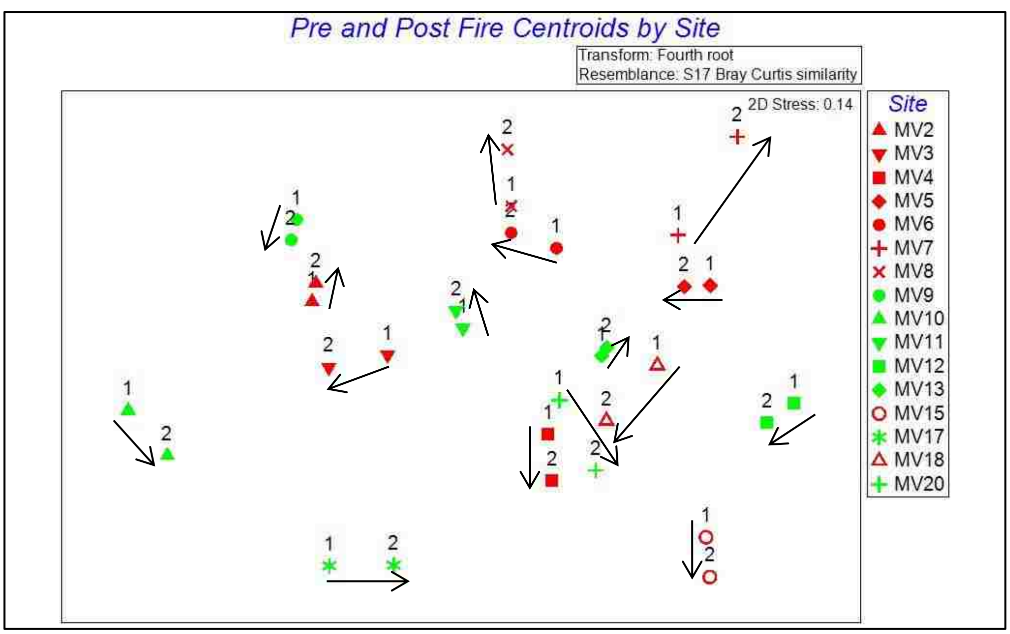


Figure 3. NMDS in 2-D space for the cluster centroids of all 16 sites split into pre and post fire time periods, arrows illustrate direction of temporal trajectory. Red symbols = Burn; Green symbols = Control; 1 = Pre-Fire; 2 = Post-Fire

Trends within burned and unburned sites are not apparent and both treatment groups behaved similarly following the Las Conchas Fire. Also, despite the fact that there is no clear clustering of the control and burn group either before or after fire, there is a significant difference between the two treatments after fire (Table 2, C).

Sites were dominated by six combinations of four abundant grass taxa: *Danthonia parry, Festuca spp., Muhlenbergia montana, and Poa pratensis* (Table 1). We found these species to be present at every site and also to be the most abundant overall; however, their abundance and distribution varied widely across sites and is visible from Figure 4. This variation in abundance is what drives the significant differences observed between all sampling sites where essentially each site consists of the same suite of species but with frequencies that fluctuate enough to create a different community configuration. In Figure 3 there seems to be some clustering of sites by species dominance where sites on the left portion of the figure (MV 2, 3, 9, 10, 17) all lack a presence of *Danthonia parryi* but have the highest levels of *Muhlenbergia montanta* and *Festuca* species. Sites on the upper right-hand side (MV6, 7, and 8), which have the lowest frequency of *Poa pratensis* and the largest of *Danthonia parryi*, are consistent in their direction of movement post-fire with a slight increase in abundance of *Poa pratensis;* these sites are also geographically close to each other. Sites on the lower right-hand quadrant (MV4, 5, 12, 13, 15, 18, 20) are all dominated by *Danthonia parryi*, *Poa pratensis*, and *Festuca* species with relative abundances of each being similar across sites; these sites have all been grazed by cattle in the 5 years following fire (Table 1).


Pre and Post Fire Centroids by Site
DAPA Relative Abundance
Transform: Fourth root
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.14
DAPA
30
120
210
300
Pre and Post Fire Centroids by Site
MUMO Relative Abundance
Transform: Fourth root
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.14
MUMO
9
36
63
90
Pre and Post Fire Centroids by Site
FESX1 Relative Abundance
Transform: Fourth root
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.14
FESX1
20
80
140
200
Pre and Post Fire Centroids by Site
POPR Relative Abundance
Transform: Fourth root
Resemblance: S17 Bray Curtis similarity
2D Stress: 0.14
POPR
20
80
140
200

[redacted]

The same three treatment comparisons were made using only the top four dominant species across all years and showed no significant difference between pre-fire conditions and the burn group post-fire but did show significant differences between pre-fire conditions and the post-fire control group and also between the post-fire control and post-fire burn group (Table 3). There were no significant differences between temporal (pre & post-fire) or treatment (control & burn) groups for the *Festuca* or *Muhlenbergia* species. The abundance of *Poa pratensis* differed significantly between the post-fire control group and pre-fire conditions and also between the burn and control groups post-fire (Fig. 5). Only *Festuca* differed significantly (P = 0.01) between 2010 and 2011 among the burned sites, and nearly so between 2010 and 2012 (P = 0.07). Average *Festuca* percent cover went from 28% in 2010 to 14%, 17%, and 32% in 2011, 2012, and 2015, respectively; the 2015 average was not significantly different from the 2010 pre-fire average (Fig. 5).

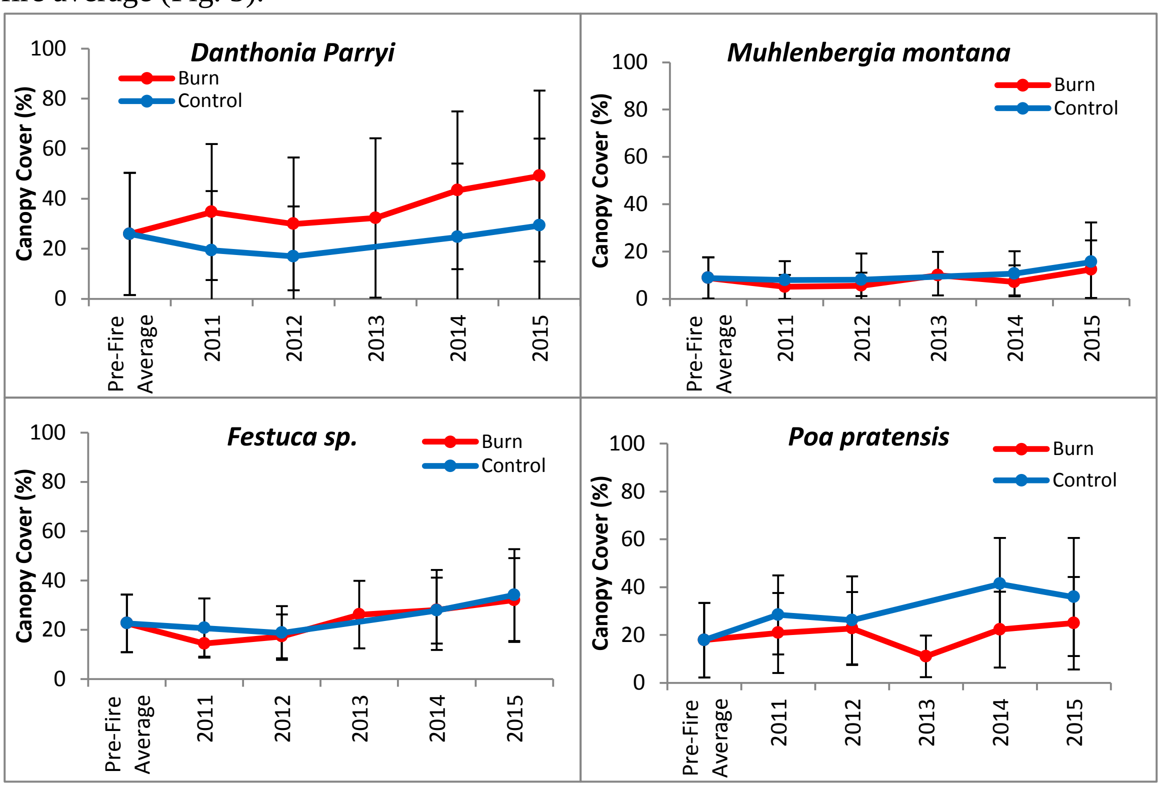


Figure 5. Temporal trends of species canopy cover between control and burn treatment groups for the top four dominant species.

Table 3. PERMANOVA and SIMPER results of pairwise comparisons for dominant species assemblages in burned and control treatment groups, before and after fire for entire study period. Abundance values are based on data transformation to the 4th root. Comparisons with significance are highlighted.

Pairwise Comparisons: Dominant Species	**PERMANOVA**			**PERDISP***		
Pre(2001-2010)-Post(2011-2015)	***t***	***p***	**Permutations**	***t***	***p***	**Permutations**
Pre-Fire , Post-Fire.BURN	1.2	0.26	999	1.6	0.21	999
Pre-Fire, Post-Fire.CONTROL	2	0.03	998	0.5	0.69	999
Post-Fire.BURN, Post-Fire.CONTROL	1.9	0.05	998	1.6	0.2	999

SIMPER	**Pre-Fire**	**Post-Fire.BURN**				
Species	**Av.Abund**	**Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
DAPA	2.18	2.65	8.53	1.12	47.07	47.07
POPR	2.49	2.61	3.88	1.3	21.41	68.48
MUMO	2.04	2.03	3.31	1.25	18.28	86.76
FESX1	2.79	2.81	2.4	1.35	13.24	100

SIMPER	**Pre-Fire**	**Post-Fire.CONTROL**				
Species	**Av.Abund**	**Av.Abund**	**Av.Diss**	**Diss/SD**	**Contrib%**	**Cum.%**
DAPA	2.18	1.9	8.96	1.23	44.72	44.72
POPR	2.49	3.03	4.26	1.28	21.27	65.99
MUMO	2.04	2.04	4.07	1.27	20.3	86.29
FESX1	2.79	2.83	2.75	1.38	13.71	100

SIMPER	**Post-Fire.BURN**	**Post-Fire.CONTROL**				
Species	**Av.Abund**	**Av.Abund**	Av.Diss	Diss/SD	Contrib%	Cum.%
DAPA	2.65	1.9	9.08	1.23	46.92	46.92
POPR	2.61	3.03	3.79	1.23	19.57	66.48
MUMO	2.03	2.04	3.71	1.33	19.15	85.63
FESX1	2.81	2.83	2.78	1.43	14.37	100

*PERDISP is a permutational dispersion test, i.e. test of variability or spatial spread between groups.

In addition to treatment comparisons, pre and post-fire comparisons were made over the entire 15 year study period for each site individually and revealed that not all sites were significantly different between the two time periods. Eight of the 16 sites, were significantly different post-fire and these consisted of five burn and three control sites. The other half was not significantly different post-fire and consisted of four burn and four control sites. In these temporal comparisons the only detectable change in composition from fire was the loss of *Symphyotrichum ascendens* and *Deschampsia caespitosa* at two sites (MV7 and MV18, respectively) following 2010 which have not reappeared since the fire. These two species are relatively rare with average cover values less than 2%; *Symphyotrichum ascendens* was present at all other sites but *Deschampsia* was not. Rare species (in terms of frequency) such as mosses, *Agrostis scabra*, *Poa fendleriana*, *Vicia americana*, and *Symphotricum* contributed most to the differences seen post-fire for each site individually but in general the species with the greatest contribution to dissimilarity, including the few named here, varied across sites and was not consistent within treatment group. Of the species that contributed to the top 15% cumulative abundance, some increased (*Vicia americana*) while others decreased (*Agrostis scabra*, *Poa fendleriana*, *Symphotricum ascendens*) after 2011 but, again, some of these declined in the control sites as well so a clear effect of fire was not detectable.

To further support grassland resilience to fire we found that deviation, among the burned sites, from the pre-fire biomass average was -0.8 the first year after fire in 2011, followed by 0.2, -0.1, 0.4, and 0.4 in 2012, 2013, 2014, and 2015, respectively. Annual standing biomass immediately after the 2011 fire was significantly lower than the pre-fire average (P = 0.0001), and fully recovered to pre-fire levels by 2012. Standing biomass for each post-fire year, from 2012-2014, did not differ from the 10 year pre-fire average, however, biomass levels for 2015 were significantly (P = 0.002) higher than the pre-fire average (Fig. 6). This temporal trend tracks average precipitation accumulation for the months of June and July as well as for the full water year (Fig. 7).

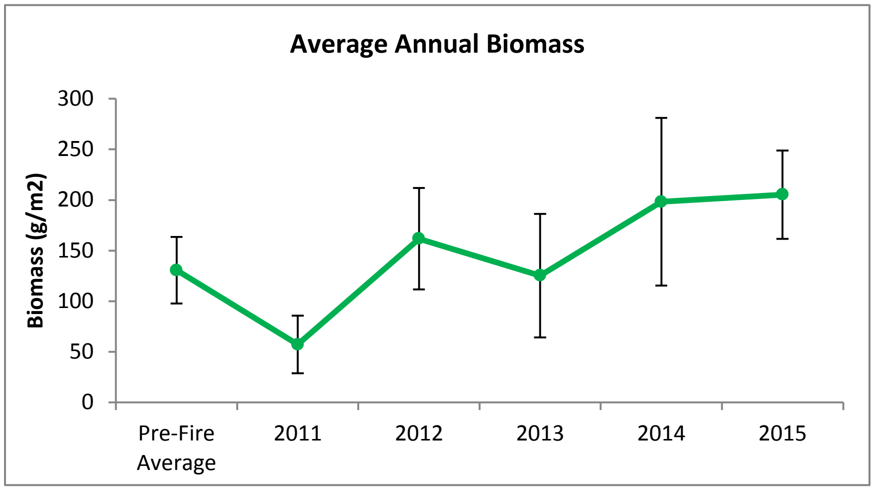


Figure 6. Average annual standing crop biomass in g/m^2 for pre-fire time period and all post-fire years.

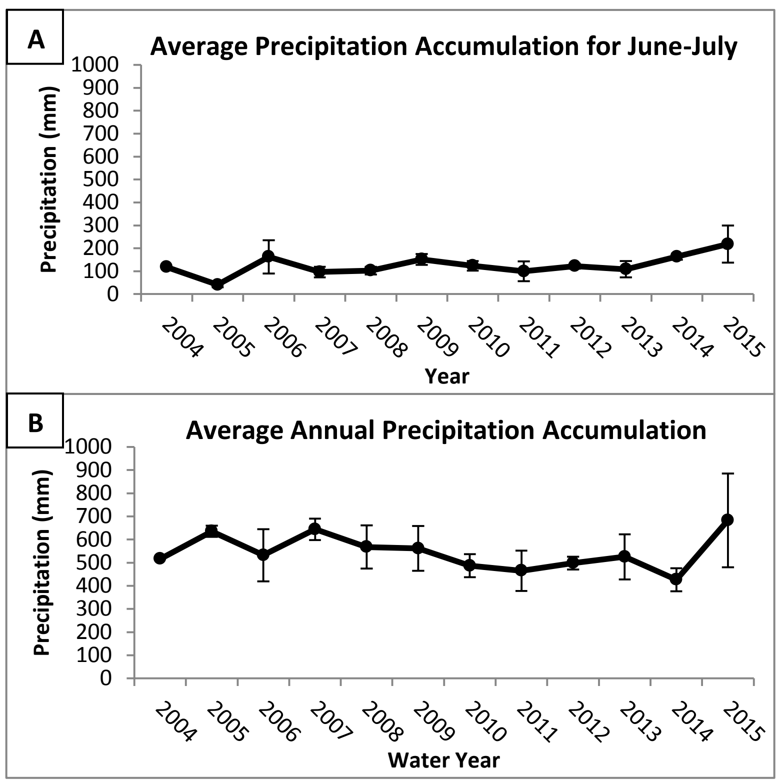


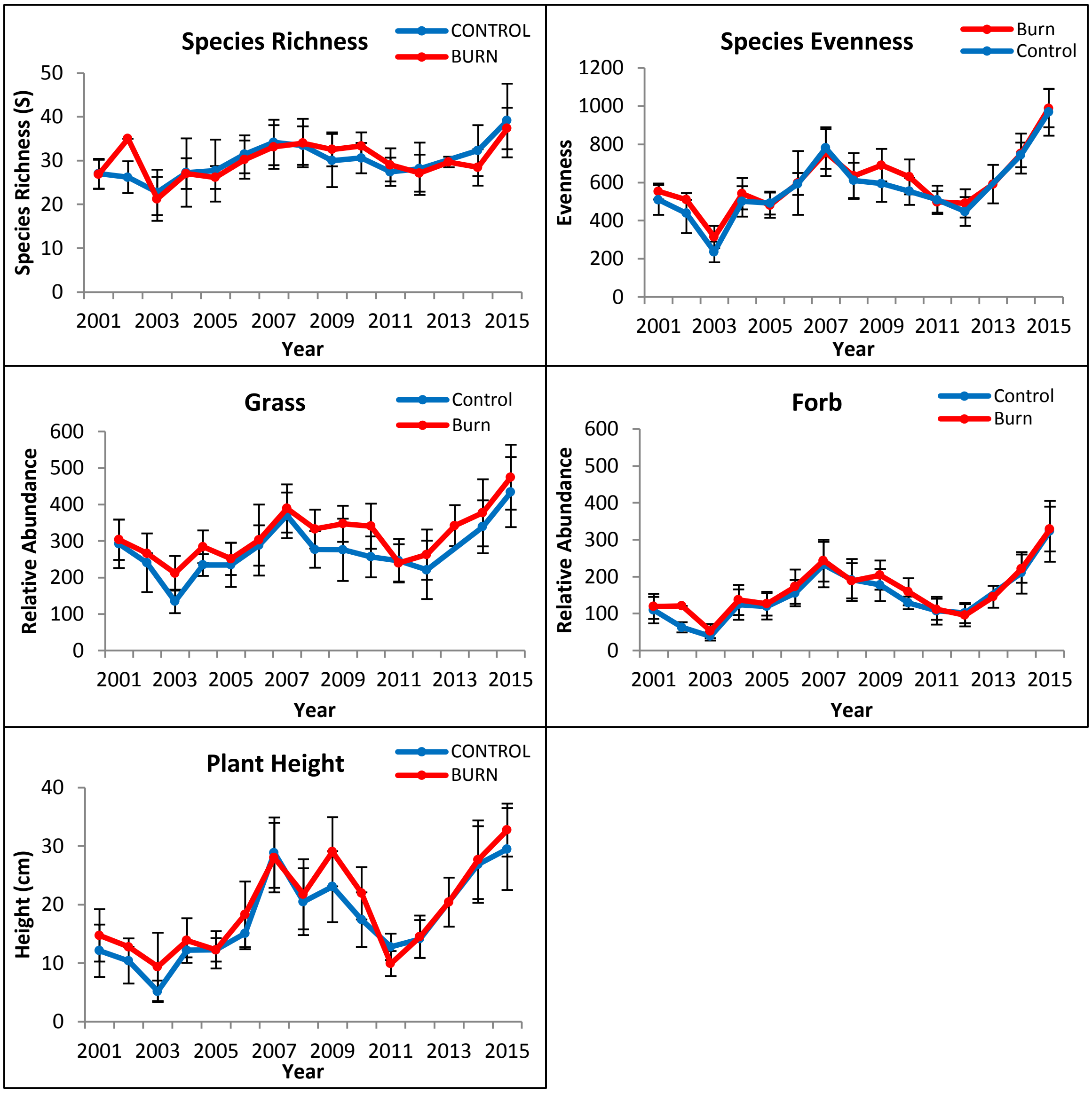
Figure 7 (a) Precipitation accumulation by water year (Oct-Sept) averaged over 4 weather stations. (b) Precipitation accumulation for June and July averaged over 4 weather stations.

Based on Repeated Measures ANOVA, no significant differences occurred between the burn and control treatment groups in any year for plant canopy height, species richness, species evenness, grass and forb abundance (Table 4, Fig. 8). Temporal trends in community assemblage, however, were still visible despite high interannual variability in precipitation. The Randomized Block Design (RBD) tests found that all variables other than species richness were significantly higher after fire even for the control sites (Table 4). A visual assessment of the temporal trends for each variable shows that the burned and control sites are responding similarly following the 2011 fire (Fig. 8).

Table 4. Results of Repeated Measures ANOVA and Randomized Block Design for five plant community variables; data shown in Fig. 8 below.

Repeated Measures ANOVA Treatment:Year	**Df**	**Sum Sq**	**F value**	**P-value**
Plant Height	1	17	0.301	**0.59**
Richness	1	18	0.611	**0.43**
Evenness	1	10376	0.435	**0.51**
Grass	1	133	0.018	**0.89**
Forb	1	4025	0.819	**0.37**

Random Block Design Before/After Fire	**Df**	**Sum Sq**	**F value**	**P-value**
Plant Height	1	331.5	4.7563	**0.03 ***
Richness	1	70.9	3.0605	**0.08**
Evenness	1	402486	14.6929	**<0.05 *****
Grass	1	51215	7.8948	**<0.05 ****
Forb	1	50697	9.4613	**<0.05 ****



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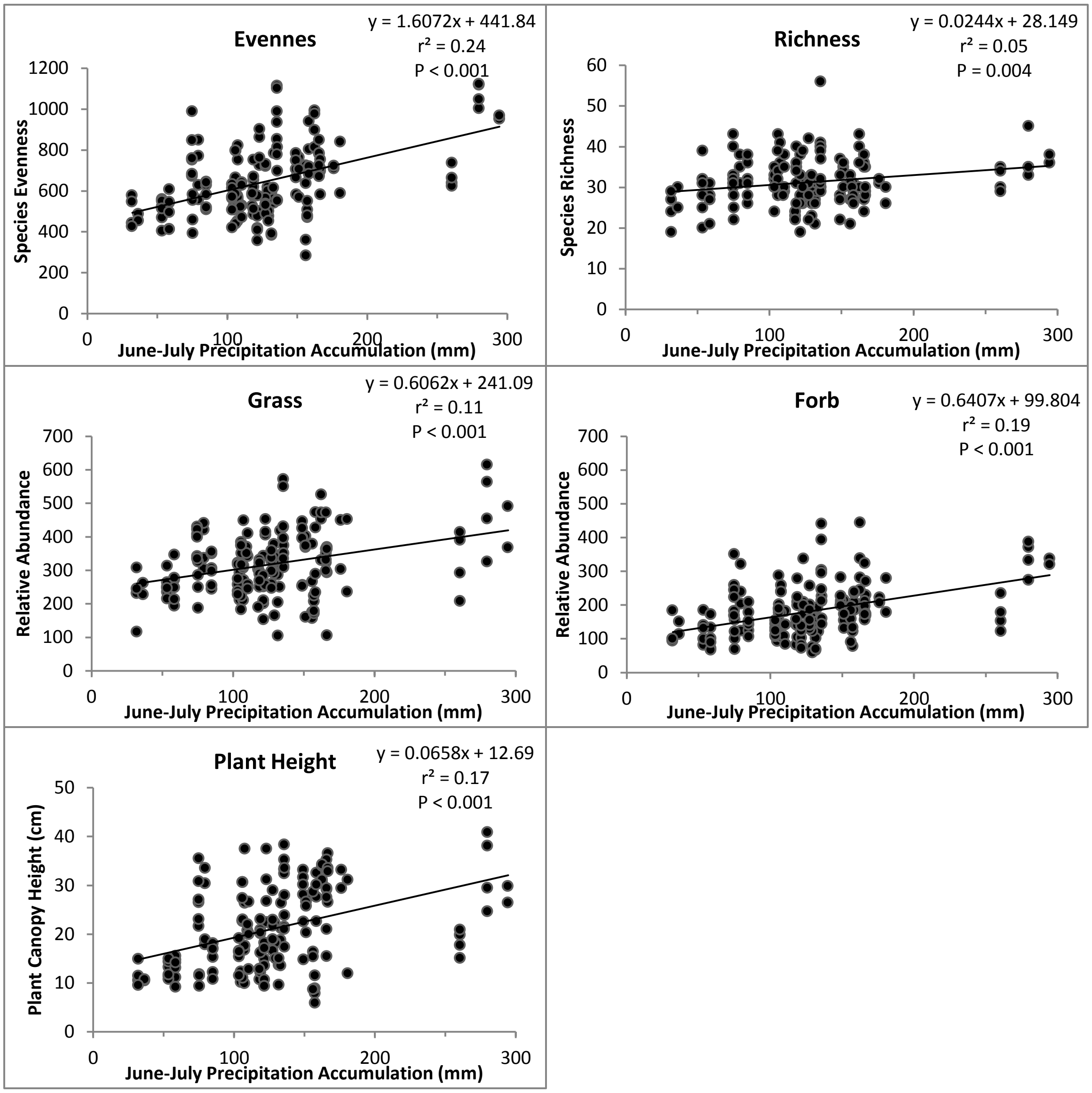
All five community structure variables were positively correlated with summer monsoonal precipitation, but not winter or annual totals (Fig. 9). This suggests that temporal patterns in plant community structure are strongly driven by climate overshadowing any effect of treatment for all years even after the fire. Litter, as a biotic influence on community trends, was negatively correlated with species evenness (r^2 = -0.67, P < 0.001).



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DISCUSSION

Fire is a natural disturbance that can be used as a management tool for maintaining healthy ecosystems, particularly in those that have evolved with fire (Wright & Bailey 1982). The potential negative impacts of fire in grassland systems include the loss of herbaceous cover which leads to water runoff and erosion consequently impacting water quality and watershed function (Dahm et al. 2015). Fire may increase opportunities for invasion or colonization of undesired species (Hunter 2006), and change plant community composition in a way that impacts habitat type and quality leading to altered trophic structure and dynamics (Ford & McPherson 1996). Yet, the Las Conchas wildfire had few impacts on the montane valley grasslands of the Valles Caldera National Preserve. There were no differences in plant canopy height, species richness, species evenness, and grass and forb abundance between the burn and control groups for any year including the year immediately after the fire. Pre-fire biomass conditions were reached by the second growing season after fire and no new exotic species were detected; existing exotic species such as *Poa pratensis*, which are considered naturalized, followed the same post-fire temporal recovery trend in abundance as native species. These results suggest that this montane grassland system is highly resilient following fire.

Unlike most fire prone and fire adapted systems of lower and drier elevations in the Southwest, the montane grasslands of the Valles Caldera National Preserve are dominated by C_3 cool season rather than C_4 warm season grasses. Studies have found that fire favors C_4 grasses (Tix & Charvat 2005) and that large expansions of C_4 grasses in the late Miocene were driven by frequent fire (Scheiter et al. 2012); therefore, it is generally thought that C_4 species are better adapted to fire than C_3 species. Consequently, most studies on fire impacts to grasslands are conducted in C_4 dominated grasslands, therefore, grassland response to fire in a C_3 dominated system has not been widely investigated. C_3 dominance in the Valles Caldera may be due to the relatively high levels of precipitation received coupled with lower temperatures of the valley bottoms (mean minimum weekly temperature in valley bottoms is -1.3° C, June-August) caused by cool-air drainage from the surrounding mountain slopes. This causes the phenomenon of the inverted treeline (Coop 2007b) which creates a cooler, moister environment relative to the surrounding forests. We suspect that these conditions allow this unique environment to support C_3 grassland expansions, over $C_{4,}$ which are also fire adapted.

These grasslands follow a common trend observed in other grassland systems where a small number of "dominant" species account for the majority of herbaceous cover while the majority of species are relatively rare (Collins and Glenn 1991). The top four dominant species *Danthonia parry, Festuca spp.*, *Muhlenbergia montana, and Poa pratensis* were virtually unaffected by the fire. *Danthonia parryi* was not evenly distributed between the control and burn treatment groups and occurred more frequently in the burn sites even before the fire thus accounting for its contribution to significance based on SIMPER analyses. Again, this is likely an effect of its naturally inconsistent distribution rather than an effect from fire (Fig. 8). Therefore, these four dominant species were not found to be responsible for post-fire significance at the community level; instead, rarer species were more highly influenced and accounted for significance.

The *Festuca arizonica/ F. idahoensis* group drop in abundance from 2010 to 2011 may be due to a slower growth rate relative to the other dominant species rather than species mortality, especially for *F. arizonica*. Studies have found that *F. arizonica* survives most fires (Servis & Boucher 1999) and may even become more abundant after fire (Harris & Covington 1983; Haisley 1984; Sackett et al. 1996), but recovery is slower in more severe burns (Vose & White 1987). Here, average percent cover for Arizona fescue had fully recovered by 2012. Similarily, *F. idahoensis* can survive low severity fires but is harmed by severe burning and can be severely damaged by fire in all seasons (Wright et al. 1979; Boyer & Dell 1980; Cattelino 1980; Smith & Busby 1981). Depending on the ratio of Idaho to Arizona fescue (which cannot be determined here) it may be possible that the greater sensitivity to fire by Idaho fescue may account for the significant differences observed between 2010 and 2011.

Muhlenbergia montana has been shown to decrease in density the first few years after fire (Gaines et al. 1958; Oswald & Covington 1984) but may also increase in density beyond pre-fire levels after recovery (Andariese 1982). Harris & Covington (1983) found Mountain muhly increased to pre-fire biomass within 10 months after a prescribed burn in central Arizona.

Several studies have shown that *Poa pratensis* is most affected by the timing of fire where late spring fires are the most damaging because Kentucky bluegrass is a cool season grass (Daubenmire 1968; Risser et el. 1981). Because the Las Conchas fire occurred in late June it seems likely that the timing of the fire was outside of its most vulnerable stage.

The Fire Effects Information System (FEIS) did not return any results for *Danthonia parryi* but the individuals present in the Valles Caldera are robust and likely re-sprouted shortly after the fire. A study on post-fire recovery of the closely related *Danthonia spicata*, a perennial bunch grass also present in these grasslands, found that adult survivorship increased with increasing plant size but changes in population size were driven mainly by genetic and environmental changes (Scheiner 1988). SIMPER showed *Danthonia parryi* to be the greatest contributor to significance between all pairwise comparisons at the community level with average abundance increasing in the post-fire burn group and decreasing in the post-fire control group. The uneven distribution of *Danthonia parryi* between the two treatments accounts for these results rather than an effect from fire.

The only detectable change in composition from fire was the loss of *Symphyotrichum ascendens* and *Deschampsia caespitosa* following the burn. Since the relative abundance of *Symphyotrichum* is low at MV7 it may still be present in the community but simply has not been documented since 2011; no studies have been found on the fire ecology of this species. *Deschampsia caespitosa* typically prefers moist habitats such as wet meadows. This species can tolerate all but the most severe fires (DeBenedetti & Parsons 1984). However, *Deschampsia* decreases with excessive cattle grazing which may explain its relatively low abundance and subsequent inability to recover after fire under continued grazing pressure (Mueggler & Stewart 1980).

All plant community analyses showed the control group following the same trends as the burn group after fire indicating that fluctuations in community structure were not driven by fire nor were they due to simple post-fire succession but rather driven by other abiotic and biotic factors. The species with the greatest contributions to significance between pre-fire and both post-fire treatment groups, for the full study period, were similar suggesting that factors driving temporal change were consistent between the two treatments therefore eliminating fire alone as a significant influence on community change. Significant positive correlations between precipitation and plant community characteristics showed that climatic variables had a greater influence on post-fire behavior in both treatment types than fire. Species evenness showed a significant negative correlation with ground cover of dead plant material across all sites. Litter accumulation can suppress plant growth; therefore, removal of dead plant material can increase germination by making resources such as sunlight more readily available (Xiong & Nilsson 1999). Litter removal from fire resulted in an increase in species evenness at the community level which seems to be due to the increased colonization or proliferation of rarer species given that the abundance of our four most dominant species was not significantly altered. However, control sites that did not undergo burning also exhibited this same response which is likely due to litter reduction from cattle grazing given that five of the seven control sites were grazed in the five years following the 2011 fire. As a result, litter removal whether by fire or grazing coupled with precipitation trends resulted in a somewhat post-fire homogenization across these grassland sites.

Fire alone did not have any significant effects, adverse or otherwise, beyond the normal range of variability for these grassland systems. Similarly, a synthesis (Scheintaub 2009) on fire effects in the semi-arid grasslands of the North American Great Plains (C_4 dominated grasslands receiving less than 600 mm mean annual precipitation) found a neutral to negative response in aboveground net primary productivity after fire regardless of season of fire, grazing history, and mean or actual precipitation. Further, variability in ANPP, which was similar to that of plant community composition, cover, and diversity, was not attributed to any particular site or fire characteristic. Vermeire et al. (2014) also identified fire resilience in semiarid rangelands in the northern mixed prairie of Montana (C_3 dominated; mean annual precipitation 339 mm) by observing that interannual weather-induced shifts in biomass were greater than those from fire, grazing, or the interaction of fire and grazing. However, results from an experimental fire on an ecotonal zone of Chihuahuan Desert scrubland and short-grass steppe in the Sevilleta National Wildife Refuge of New Mexico showed plant species demography was highly impacted by fire. Of five abundant grass species, three were significantly negatively impacted by fire in ways that were not detected in this study (Parmenter 2008). The dominant plant species of the Valles Caldera showed no significant decline in abundance beyond the first year after fire; and on the community level species evenness increased indicating greater resilience to fire than desert grasslands.

The long term establishment of perennial species, particularly cool-season bunch grasses, and their exposure to grazing, fire, and extreme low temperatures has resulted in a system that is highly variable across both space and time. The high level of difference among sites irrespective of fire may be attributed to variations in grazing pressure along with varying climatic and environmental conditions that together create microhabitats within each site and essentially create the spatial mosaic of valley flora which exists throughout the volcanic crater of the Valles Caldera. The specific drivers and the degree to which they influence each unique assemblage are complex and present an opportunity for further study. Although the significant differences between all sites may reduce sample size it also presents the opportunity to investigate the effect of fire among different versions of a montane grassland plant community for a more robust sense of fire impacts to the overall ecosystem. A significant impact to these plant communities from wildfire burning was not detected outside of this range of variability therefore demonstrating a relatively high level of resistance to fire.

Studies on recent forest encroachment into the grasslands of the Valles Caldera found fire to be a large component in the maintenance of grasslands with 20th century fire suppression leading to the greatest loss of grassland acreage since 1935 (Coop 2007a); furthermore, Coop encouraged the use of prescribed fire as a management tool to restore and maintain grasslands of the Valles Caldera. The implications of the results in this study support land and fire management efforts to restore historic fire return intervals by providing evidence that any risk of adverse or unwanted effects on plant community composition and function for high elevation fire-adapted grasslands is low to none. Burning promotes diversity and thus stability via removal of litter accumulation. Furthermore, maintaining fire frequency or fire return interval is crucial for maintaining ecosystem function since these grasslands are adapted to a particular fire regime (Keeley et al. 2011). It is unclear, however, if the potential for more frequent, high intensity fire under climate change will alter these low intensity, fire-adapted ecosystems in terms of community composition and ecological function.

In conclusion, restoring historic fire regimes will help maintain the health and resilience of this landscape as well as reduce loss of grassland to shrub and tree encroachment driven by fire suppression (Allen 1989; Coop 2007a). In addition, natural systems that are extremely resistant to environmental extremes should be preserved and protected from anthropogenic development activities, as they will be able to naturally resist climate change better than other less resistant/resilient ecosystems.

APPENDIX A: Plant Species Abundance Data

Family	USDA Scientific Name	Treatment	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
	Achnatherum	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.0
Poaceae	lettermanii (Vasey)	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
	Achillea millefolium L.	Control	17.4	4.2	2.5	10.3	15.4	11.9	28.6	26.3	18.4	12.9	10.9	10.3		31.6	49.7
Asteraceae	var. occidentalis DC.	Burn	25.3	19.0	3.0	17.5	19.4	25.3	40.2	30.3	28.7	18.2	9.6	10.1	16.8	32.4	49.9
	Agoseris aurantiaca	Control	0.2	0.2	0.0	0.3	0.1	0.3	0.0	0.1	0.4	0.3	0.1	0.1		1.1	0.9
Asteraceae	(Hook.) Greene var.	Burn	0.4	0.0	0.0	0.1	0.3	0.1	0.4	0.3	0.6	0.7	0.1	0.0	0.5	1.3	1.3
	Agoseris glauca (Pursh)	Control	0.8	0.4	0.3	0.3	0.0	0.6	0.4	0.3	0.4	0.4	0.1	0.4		0.9	1.1
Asteraceae	Raf. var. glauca	Burn	0.0	2.0	0.3	0.1	0.0	0.1	0.3	0.0	0.4	0.7	0.9	0.1	0.0	0.6	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.9		0.0	0.6
Asteraceae	Agoseris species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.3
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.0
Poaceae	Agrostis species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	7.2	0.0	0.0	0.3	0.1	0.0	2.3	3.4	6.1	5.4	1.1	0.7		1.1	5.3
Poaceae	Agrostis scabra Willd.	Burn	11.0	0.0	0.0	0.4	0.1	3.1	9.7	14.3	22.1	6.8	0.0	1.4	0.8	3.3	5.0
		Control	0.0	0.0	0.3	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.0
Liliaceae	Allium cernuum Roth	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
	Allium geyeri S. Watson	Control	0.0	0.0	0.3	0.3	0.3	0.3	0.1	1.0	0.1	0.1	0.0	0.1		0.7	1.3
Liliaceae	var. geyeri	Burn	0.0	0.0	0.0	0.1	0.0	0.2	0.2	0.5	0.3	0.3	0.2	0.0	0.0	0.6	1.3
	Androsace	Control	0.0	0.0	0.0	0.5	0.0	3.4	5.0	0.9	1.0	0.0	0.3	0.0		0.0	1.4
Primulaceae	septentrionalis L.	Burn	0.0	1.0	0.0	0.1	0.0	0.3	0.8	0.0	0.0	0.0	0.0	0.0	0.3	0.0	1.3
		Control	13.8	7.2	4.5	8.5	17.7	23.0	35.6	29.1	40.4	22.3	19.3	16.3		39.4	49.1
Asteraceae	Antennaria species	Burn	13.6	6.0	8.3	16.0	18.0	22.2	40.2	33.0	33.4	22.1	13.2	10.9	13.5	22.7	36.3
	Artemisia carruthii	Control	0.8	0.2	0.0	0.8	1.6	1.4	2.0	0.7	0.6	1.4	1.3	0.1		0.6	2.6
Asteraceae	Wood ex Carruth	Burn	0.1	0.0	0.0	0.6	0.0	0.6	0.0	0.3	0.6	0.1	0.0	0.1	0.2	0.1	0.9
	Artemisia dracunculus	Control	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Asteraceae	L.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Arabis drummondii	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Brassicaceae	Gray	Burn	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
	Arenaria fendleri Gray	Control	7.8	12.2	4.8	7.8	11.9	17.3	18.9	16.6	9.7	13.3	8.4	7.9		12.7	27.6
Caryophyllaceae	var. fendleri	Burn	3.3	5.0	15.3	15.4	16.7	17.1	17.2	16.3	15.6	18.1	13.0	11.7	9.7	14.7	25.6
		Control	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.9	0.1
Asteraceae	Artemisia frigida Willd.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Arabis hirsuta (L.) Scop.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Brassicaceae	var. pycnocarpa (M.	Burn	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Artemisia ludoviciana	Control	0.0	0.2	0.0	0.3	0.1	0.0	0.4	0.7	0.0	0.1	0.3	0.7		1.0	1.3

APPENDIX A: Plant Species Abundance Data

Family	USDA Scientific Name	Treatment	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
Asteraceae	Nutt. ssp. ludoviciana	Burn	1.1	0.0	0.0	0.6	0.9	1.2	1.3	1.5	2.1	0.8	0.3	0.4	1.2	1.3	2.2
	Artemisia ludoviciana	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4		0.6	0.0
Asteraceae	Nutt. ssp. mexicana	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.3	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	6.7
Asteraceae	Asteraceae species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
	Besseya plantaginea (E.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Scrophulariaceae	James) Rydb.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.2	0.0
	Blepharoneuron	Control	73.2	52.8	7.5	38.8	16.6	16.1	23.4	17.0	19.9	17.4	29.9	24.7		38.0	52.3
Poaceae	tricholepis (Torr.) Nash	Burn	73.5	45.0	28.5	32.9	24.1	15.3	25.7	31.9	30.2	26.8	29.4	38.1	56.8	42.3	63.7
	Bouteloua gracilis	Control	6.4	5.4	10.5	16.8	15.7	15.3	13.1	8.3	11.0	9.0	13.7	10.9		12.1	10.7
Poaceae	(Willd. ex Kunth) Lag.	Burn	0.1	0.0	0.3	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0		0.0	0.0
Brassicaceae	Brassicaceae species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	Bromus ciliatus L.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	6.1	0.0	0.0	0.0
	Bromus inermis Leyss.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	ssp. inermis	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	Bromus species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
	Bromus anomalus Rupr.	Control	2.2	2.0	0.3	0.5	0.7	0.3	1.4	0.7	0.9	0.7	0.7	1.1		0.6	2.6
Poaceae	ex Fourn. And Bromus	Burn	15.0	0.0	0.3	2.5	2.9	4.7	8.4	1.6	5.4	9.7	4.9	1.4	16.2	11.9	20.0
	Calamagrostis	Control	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	canadensis (Michx.)	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Calochortus gunnisonii	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.4	0.0	0.0	0.0		0.0	0.9
Liliaceae	S. Watson var.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.1	0.0	0.0	0.0	0.0	0.1
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.0
Scrophulariaceae	Castilleja lineata Greene	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Campanula parryi Gray	Control	0.4	0.2	0.3	0.0	0.9	0.7	0.0	0.7	0.4	0.4	1.3	0.0		0.7	0.4
Campanulaceae	var. parryi	Burn	0.5	3.0	0.0	0.0	0.4	2.7	0.1	1.0	1.0	1.8	1.6	0.3	0.0	0.1	0.2
	Campanula rotundifolia	Control	6.2	2.6	2.3	4.8	5.6	10.0	18.7	10.3	5.4	9.3	4.3	6.6		16.6	33.3
Campanulaceae	L.	Burn	3.6	1.0	3.3	3.9	3.3	6.9	11.2	6.4	12.1	6.4	3.6	7.6	8.8	14.3	22.3
	Calamagrostis stricta	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	(Timm) Koel. ssp.	Burn	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.8	0.1	0.0	0.0	0.0	0.0
		Control	20.6	34.8	16.5	40.5	38.6	49.6	48.9	46.7	30.4	44.6	42.6	30.9		39.9	45.1
Cyperaceae	Carex species	Burn	10.0	23.0	18.3	32.6	32.9	36.1	31.7	36.9	33.8	26.7	41.3	39.3	47.0	46.0	43.2

APPENDIX A: Plant Species Abundance Data

Family	USDA Scientific Name	Treatment	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
Cyanobacteria, green		Control	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0		0.0	0.0
algae, lichen, moss,	Cryptobiotic Crust	Burn	0.0	0.0	0.0	2.1	0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
		Control	3.4	0.6	0.0	3.0	4.4	5.4	10.6	9.3	20.1	7.6	7.7	3.4		1.6	2.9
Caryophyllaceae	Cerastium arvense L.	Burn	0.4	1.0	0.0	0.8	0.7	2.0	3.9	3.8	9.3	5.2	1.0	0.9	0.5	0.6	1.8
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Chenopodiaceae	Chenopodium atrovirens	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Chamaesyce	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.0
Euphorbiaceae	serpyllifolia (Pers.)	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Cirsium arvense (L.)	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Asteraceae	Scop.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
	Conyza canadensis (L.)	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Asteraceae	Cronquist var.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Crepis runcinata (James)	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0		0.0	0.0
Asteraceae	Torr. & Gray ssp.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Dasiphora floribunda	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Rosaceae	(Pursh) Kartesz, comb.	Burn	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.1	0.0	0.0	0.0	0.7	0.0	0.0	1.1
		Control	43.8	38.6	14.0	15.8	42.7	51.1	58.3	54.6	62.4	56.9	58.1	50.9		74.0	87.9
Poaceae	Danthonia parryi Scribn.	Burn	83.9	118.0	66.3	91.4	75.4	101.9	112.3	126.5	123.2	120.2	104.0	89.7	96.8	130.0	147.2
	Danthonia spicata (L.)	Control	0.8	0.4	0.0	2.5	3.7	1.6	2.9	3.0	3.0	3.6	1.6	0.0		1.1	0.0
Poaceae	Beauv. ex Roemer &	Burn	0.0	7.0	0.0	2.3	2.6	3.3	3.6	1.8	3.2	1.6	0.7	0.0	0.0	0.0	0.0
	Deschampsia caespitosa	Control	1.8	0.0	1.0	2.0	1.4	0.7	2.0	2.1	1.6	0.9	1.4	0.7		0.1	0.0
Poaceae	(L.) Beauv.	Burn	1.5	3.0	0.3	0.0	0.0	0.1	1.1	3.0	0.6	1.7	0.0	0.3	0.0	0.0	0.2
	Descurainia incana	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Brassicaceae	(Bernh. ex Fisch. &	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Draba aurea Vahl ex	Control	0.0	0.0	0.0	0.0	0.1	0.0	0.3	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Brassicaceae	Hornem.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Brassicaceae	Draba species	Burn	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Draba helleriana Greene	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Brassicaceae	var. helleriana	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Caryophyllaceae	Drymaria species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Elymus alaskanus	Control	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	(Scribn. & Merr.) A.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Elymus elymoides	Control	6.4	5.8	0.5	3.8	7.7	9.6	9.1	3.1	3.3	8.1	9.0	5.7		7.3	17.6

APPENDIX A: Plant Species Abundance Data

Family	**USDA Scientific Name**	**Treatment**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**
Poaceae	(Raf.) Swezey ssp.	Burn	4.8	0.0	6.5	9.5	9.3	13.4	15.6	4.3	7.0	6.7	6.4	14.6	23.3	15.4	25.9
	xElyhordeum macounii	Control	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	(Vasey) Barkworth &	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
	Elymus trachycaulus	Control	1.6	2.0	0.8	1.0	3.0	5.6	7.6	6.3	2.4	2.3	4.6	3.9		2.9	7.9
Poaceae	(Link) Gould ex	Burn	0.9	0.0	3.5	4.9	3.0	4.0	5.8	2.5	3.8	3.4	1.0	3.7	5.3	4.2	8.8
	Elymus trachycaulus	Control	1.0	3.4	0.0	1.3	1.7	2.0	7.0	2.6	1.7	4.3	0.6	1.6		14.7	12.6
Poaceae	(Link) Gould ex	Burn	0.6	5.0	0.8	1.1	1.3	4.2	5.9	2.9	1.0	1.1	1.3	0.7	0.5	2.9	4.0
	Elymus trachycaulus	Control	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0		0.0	0.0
Poaceae	(Link) Gould ex	Burn	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	1.3	0.4		0.0	0.0
Poaceae	Elymus species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	3.7	0.0	2.3	0.0	0.0
	Epilobium ciliatum Raf.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Onagraceae	ssp. ciliatum	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Erigeron divergens Torr.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6		0.0	0.0
Asteraceae	& Gray	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	3.6	4.0	3.0	21.0	6.4	11.1	14.7	24.0	17.9	19.6	11.6	8.9		4.7	15.3
Asteraceae	Erigeron flagellaris Gray	Burn	1.1	3.0	6.3	8.9	5.7	12.6	15.9	21.0	9.4	11.3	6.2	5.3	5.3	13.3	19.8
	Erigeron formosissimus	Control	28.6	4.0	3.3	19.3	19.3	21.6	24.9	27.0	35.3	19.0	21.4	18.3		43.1	41.9
Asteraceae	Greene var.	Burn	39.6	1.0	2.0	28.4	13.9	23.4	34.0	21.1	33.1	26.2	22.3	14.7	26.7	42.8	47.2
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.0
Asteraceae	Erigeron species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.2	0.0	0.0
	Erigeron subtrinervis	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Asteraceae	Rydb. ex Porter &	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Festuca arizonica Vasey	Control	81.8	57.6	44.3	46.3	53.9	55.1	93.4	72.0	81.4	65.0	62.1	56.1		83.4	102.3
Poaceae	and Festuca idahoensis	Burn	68.4	53.0	43.0	51.9	51.4	52.2	90.6	72.0	86.8	84.4	43.2	52.0	78.5	84.0	96.2
		Control	5.2	0.4	2.0	2.0	1.6	1.0	1.7	2.0	1.6	1.1	2.3	1.7		3.1	3.4
Poaceae	Festuca thurberi Vasey	Burn	2.9	0.0	0.0	2.4	0.0	1.7	2.6	2.5	2.6	2.3	0.7	1.1	0.0	1.6	1.2
	Fragaria virginiana	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Rosaceae	Duchesne	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2
		Control	0.0	0.0	0.0	0.0	0.0	0.1	0.3	0.3	0.0	0.0	0.0	0.0		0.0	0.1
Fungi	Fungi species	Burn	0.5	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.4
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0		1.1	2.0
Rubiaceae	Galium boreale L.	Burn	0.0	0.0	0.0	0.0	0.0	0.1	0.4	0.1	0.1	0.4	0.3	0.0	0.0	0.7	1.4
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.1	0.0	0.0	0.0		0.0	0.0
Gentianaceae	Gentiana affinis Griseb.	Burn	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.9	0.2	0.0	0.0	0.1	0.0	0.2	0.4

APPENDIX A: Plant Species Abundance Data

Family	USDA Scientific Name	Treatment	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
	Gentianella amarella	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.3	0.0	0.0		0.0	2.3
Gentianaceae	(L.) Boerner ssp. acuta	Burn	0.0	0.0	0.0	0.0	0.0	0.1	0.3	0.9	0.2	0.4	0.0	0.0	0.0	0.0	1.1
	Geranium caespitosum	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0		0.0	0.0
Geraniaceae	James	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.0		0.0	0.0
Gentianaceae	Gentiana parryi Engelm.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Geum triflorum Pursh	Control	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Rosaceae	var. ciliatum (Pursh)	Burn	1.5	0.0	0.0	0.0	0.0	0.6	0.8	0.6	0.8	0.2	0.3	0.1	0.2	0.6	0.9
	Gnaphalium exilifolium	Control	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Asteraceae	A. Nels.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Heterotheca villosa	Control	0.2	0.0	0.0	0.0	0.6	0.4	1.3	0.4	0.0	0.7	0.4	0.7		1.7	2.1
Asteraceae	(Pursh) Shinners	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.2	0.1	0.3	0.3	0.3	0.4	0.3
	Hieracium fendleri Sch.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1		0.0	0.1
Asteraceae	Bip. var. fendleri	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.2	0.6
	Hordeum	Control	0.4	0.0	0.3	0.5	0.6	0.9	0.9	0.0	0.6	0.1	0.0	1.0		0.1	0.9
Poaceae	brachyantherum Nevski	Burn	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Hymenopappus	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Asteraceae	newberryi (Gray) I.M.	Burn	0.0	0.0	0.0	0.1	0.0	0.0	0.2	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.1
		Control	0.2	0.2	0.0	0.3	0.1	0.1	0.6	0.3	1.0	0.6	0.4	0.3		0.7	0.9
Iridaceae	Iris missouriensis Nutt.	Burn	1.1	0.0	1.3	0.8	0.4	0.6	1.0	2.0	2.2	0.9	1.1	1.0	1.2	0.8	1.4
	Juncus balticus Willd.	Control	13.2	12.0	12.5	28.8	22.9	14.4	26.7	14.9	13.7	18.3	12.9	10.4		17.7	16.4
Juncaceae	var. montanus Engelm.	Burn	14.1	8.0	6.8	17.9	18.1	12.1	18.6	12.0	11.8	14.1	4.4	9.4	5.8	15.8	17.4
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Juncaceae	Juncus longistylis Torr.	Burn	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Juncus nevadensis S.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.4
Juncaceae	Watson	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Juncus saximontanus A.	Control	0.4	0.0	1.0	0.8	0.6	0.4	0.3	0.7	0.7	0.4	0.0	0.0		0.0	0.0
Juncaceae	Nels.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0		0.0	0.0
Juncaceae	Juncus tenuis Willd.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
	Koelaria macrantha	Control	34.2	18.8	8.3	31.0	29.1	40.6	63.6	34.3	25.4	26.4	15.4	14.4		56.4	59.7
Poaceae	(Ledeb.) J.A. Schultes	Burn	14.9	18.0	27.0	51.8	46.4	50.0	60.9	34.1	28.9	34.4	19.0	22.0	17.8	53.1	48.1
	Lathyrus lanszwertii	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3		0.0	0.7
Fabaceae	Kellogg var. leucanthus	Burn	0.0	0.0	0.0	0.3	0.3	0.2	0.1	0.1	0.3	0.6	0.2	0.6	0.7	0.4	1.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1		0.0	0.0

APPENDIX A: Plant Species Abundance Data

Family	USDA Scientific Name	Treatment	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
Brassicaceae	Lepidium species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Lepidium virginicum L.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0		0.6	0.1
Brassicaceae	var. pubescens (Greene)	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1
		Control	2.8	3.8	1.0	0.5	0.1	2.0	0.4	0.1	0.0	0.6	0.0	0.0		0.3	2.4
Lichen	Lichen species	Burn	2.0	3.0	4.3	1.5	0.4	3.0	0.6	1.8	1.2	0.3	0.8	0.7	2.0	0.2	3.3
	Lolium arundinaceum	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	(Schreb.) S.J.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.0		0.0	0.0
Asteraceae	Madia glomerata Hook.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.3	0.1	0.0	0.0		0.0	0.1
Lamiaceae	Mentha arvensis L.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mertensia lanceolata	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Boraginaceae	(Pursh) DC.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2
		Control	18.6	21.4	5.3	6.8	1.6	16.1	15.1	1.4	2.7	0.7	3.1	1.0		1.9	3.6
	unknown moss species	Burn	29.9	35.0	8.8	15.6	8.0	19.2	13.3	7.4	4.3	1.2	5.7	2.6	2.3	2.8	6.6
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	Muhlenbergia filiformis	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0
	Muhlenbergia montana	Control	23.4	33.0	32.3	49.0	34.0	46.9	34.3	31.4	31.7	24.4	23.7	24.1		31.7	46.6
Poaceae	(Nutt.) A.S. Hitchc.	Burn	12.5	3.0	24.0	21.9	15.9	27.2	22.9	19.8	19.2	20.1	15.2	16.6	29.8	21.2	37.0
	Muhlenbergia	Control	2.8	7.8	3.5	12.8	4.1	10.6	8.3	7.4	6.1	3.3	3.3	1.7		3.7	5.4
Poaceae	richardsonis (Trin.)	Burn	4.0	8.0	0.8	2.1	2.1	5.1	2.9	4.6	3.0	4.1	3.0	0.6	0.0	2.9	4.3
	Muhlenbergia wrightii	Control	0.0	4.4	1.3	4.5	1.1	4.4	3.3	4.3	3.4	7.3	6.9	5.4		6.6	9.4
Poaceae	Vasey ex Coult.	Burn	2.6	0.0	2.3	2.0	2.3	2.2	2.6	1.1	1.1	1.3	3.1	2.7	6.5	2.6	3.3
		Control	1.6	0.0	0.5	0.0	1.1	0.0	5.6	6.3	0.3	0.4	0.0	0.0		0.3	4.7
Scrophulariaceae	Orthocarpus luteus Nutt.	Burn	0.6	0.0	0.0	0.0	0.7	0.0	2.9	1.4	1.9	1.2	0.0	0.1	0.0	0.6	1.8
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Oxalidaceae	Oxalis violacea L.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Packera neomexicana	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0		0.0	0.0
Asteraceae	(A. Gray) W.A. Weber	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.6	0.4	0.1	0.0	0.3	0.0	1.0
	Pascopyrum smithii	Control	0.0	3.0	2.0	4.3	4.9	10.3	5.3	7.7	0.6	1.1	1.6	1.0		0.0	0.0
Poaceae	(Rydb.) A. Love	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Penstemon rydbergii A.	Control	0.0	0.0	0.3	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Scrophulariaceae	Nelson var. rydbergii	Burn	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.3	0.2	0.2	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.3	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.1		0.1	0.3
Poaceae	Phleum pratense L.	Burn	0.3	0.0	0.0	0.0	0.1	0.0	0.0	1.1	0.1	0.2	0.2	0.1	0.0	0.0	0.3

APPENDIX A: Plant Species Abundance Data

Family	**USDA Scientific Name**	**Treatment**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**
	Pinus ponderosa P. & C.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Pinaceae	Lawson var. scopulorum	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Plantaginaceae	Plantago major L.	Burn	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Potentilla anserina (L.)	Control	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.3	5.7	0.3		0.0	0.0
Rosaceae	Rydb.	Burn	0.0	1.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0
		Control	0.6	0.0	0.0	1.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.3
Polygonaceae	Polygonum aviculare L.	Burn	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1	3.9	0.1		0.0	17.3
Poaceae	Poa compressa L.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	2.1	0.0	0.0	0.0	9.8
	Polygonum douglasii	Control	0.0	0.0	0.0	0.0	0.1	0.0	0.3	0.0	0.0	0.1	0.0	0.0		0.0	0.1
Polygonaceae	Greene ssp. douglasii	Burn	0.0	0.0	0.3	0.0	0.0	0.0	0.2	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0
	Poa fendleriana (Steud.)	Control	0.0	5.0	6.0	1.5	11.6	15.3	32.3	16.7	13.0	19.3	8.4	14.9		0.0	7.7
Poaceae	Vasey	Burn	7.0	6.0	8.5	6.6	14.1	14.2	18.0	10.0	8.9	15.1	3.4	11.7	2.7	0.0	9.7
	Potentilla hippiana	Control	19.8	18.4	12.5	40.0	26.3	34.4	44.9	17.0	13.0	10.4	7.4	14.6		35.4	43.1
Rosaceae	Lehm. var. hippiana	Burn	23.6	41.0	11.3	34.9	35.9	41.9	49.6	25.5	23.6	15.9	15.4	16.0	34.3	50.2	56.8
	Potentilla norvegica L.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Rosaceae	ssp. monspeliensis (L.)	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	1.3
Poaceae	Poa occidentalis Vasey	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	Poa palustris L.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Potentilla pensylvanica	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Rosaceae	L. var. pensylvanica	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Poa pratensis L. ssp.	Control	36.0	58.4	25.3	58.3	67.3	57.4	78.4	67.1	82.4	92.6	85.3	78.4		124.0	107.4
Poaceae	pratensis	Burn	48.4	42.0	9.0	40.9	35.9	35.4	44.2	38.0	57.7	61.2	62.6	68.1	33.0	66.7	74.8
	Potentilla pulcherrima	Control	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Rosaceae	Lehm.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
	Prunella vulgaris L. ssp.	Control	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Lamiaceae	lanceolata (W. Bart.)	Burn	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
	Pseudocymopterus	Control	0.0	0.0	0.0	0.0	0.0	0.1	0.4	0.1	0.1	0.0	0.0	0.4		0.7	0.6
Apiaceae	montanus (Gray) Coult.	Burn	0.4	0.0	0.0	0.6	0.0	0.1	0.0	0.4	0.7	0.1	0.2	0.1	0.7	0.2	0.3
	Ranunculus	Control	2.0	2.0	0.3	0.5	2.0	0.7	3.0	2.1	2.0	2.0	1.0	0.4		2.1	4.9
Ranunculaceae	cardiophyllus Hook.	Burn	0.6	4.0	0.3	2.0	2.7	2.1	4.9	2.9	5.0	4.0	3.4	1.0	3.3	3.6	10.0
	Ranunculus cymbalaria	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0

APPENDIX A: Plant Species Abundance Data

Family	USDA Scientific Name	Treatment	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
Ranunculaceae	Pursh	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.7	0.1	0.0	0.0	0.0	0.1
	Ranunculus inamoenus	Control	0.0	0.0	0.0	0.5	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Ranunculaceae	Greene var. inamoenus	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Ranunculus macounii	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Ranunculaceae	Britt.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Ranunculaceae	Ranunculus species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
	Rorippa nasturium-	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0		0.0	0.0
Brassicaceae	aquaticum (L.) Hayek	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Rorippa palustris (L.)	Control	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Brassicaceae	Bess. ssp. fernaldiana	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Rorippa sphaerocarpa	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.1		0.0	0.6
Brassicaceae	(Gray) Britt.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.3	0.0	0.0	0.0		0.0	0.1
Polygonaceae	Rumex acetosella L.	Burn	0.5	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.0
	Rumex salicifolius	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.0
Polygonaceae	Weinm. var. mexicanus	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Saxifraga rhomboidea	Control	1.0	0.0	0.0	0.3	0.4	1.1	1.3	0.1	0.0	0.0	0.0	0.4		0.6	1.4
Saxifragaceae	Greene	Burn	1.3	0.0	0.3	1.9	1.6	1.8	2.2	0.9	0.6	0.1	0.0	0.4	1.7	2.8	1.9
		Control	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Asteraceae	Senecio species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Silene drummondii	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.3
Caryophyllaceae	Hook. var. drummondii	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.3
Caryophyllaceae	Silene species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2
	Sisyrinchium montanum	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.9	0.1	0.1		0.0	0.4
Iridaceae	Greene	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.2	1.0	0.0	0.0	0.2	0.0	0.7
	Silene scouleri Hook.	Control	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Caryophyllaceae	ssp. pringlei (S. Wats.)	Burn	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.1	0.1	0.0	0.1	0.0	0.1	0.1
	Solidago missouriensis	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.1	0.0		0.3	0.0
Asteraceae	Nutt. var. missouriensis	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.5	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.7	0.0
Asteraceae	Solidago species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
	Solidago simplex Kunth	Control	0.2	0.2	0.0	0.0	0.6	1.7	0.6	1.1	0.0	0.0	0.0	0.1		0.0	0.9
Asteraceae	ssp. simplex var.	Burn	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.4	0.0	1.1	0.0	0.1	0.0	0.2	1.8

APPENDIX A: Plant Species Abundance Data

Family	**USDA Scientific Name**	**Treatment**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Asteraceae	Solidago velutina DC.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0
	Stellaria longifolia	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Caryophyllaceae	Muhl. ex Willd.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
	Symphyotrichum	Control	0.4	4.4	2.8	5.3	4.6	6.1	9.3	8.3	7.0	4.4	2.6	6.0		5.7	3.7
Asteraceae	ascendens (Lindl.)	Burn	0.0	17.0	1.3	2.4	4.1	5.1	9.1	7.3	6.2	5.3	2.7	7.6	4.8	1.3	12.3
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.4
Asteraceae	Symphotrichum species	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
	Taraxacum officinale	Control	11.6	5.0	0.0	3.0	4.9	6.9	5.4	7.0	4.1	4.0	6.6	3.1		7.0	9.7
Asteraceae	G.H. Weber ex Wiggers	Burn	11.9	10.0	2.8	8.3	7.3	14.4	14.0	10.9	13.1	12.0	27.6	9.1	13.0	18.4	21.7
	Thalictrum fendleri	Control	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Ranunculaceae	Engelm. ex Gray	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Thinopyrum	Control	0.0	0.0	0.3	0.3	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
Poaceae	intermedium (Host)	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Tragopogon dubius	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1
Asteraceae	Scop.	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
	Trifolium longipes Nutt.	Control	0.0	0.0	0.0	0.5	0.0	1.3	0.9	2.0	0.4	0.3	0.1	0.1		0.0	4.1
Fabaceae	ssp. reflexum (A. Nels.)	Burn	0.0	2.0	0.0	0.0	0.0	0.1	0.3	0.5	0.2	0.0	0.6	0.1	0.0	0.0	0.0
		Control	2.6	0.0	0.8	1.3	0.0	0.3	0.6	3.1	5.1	5.4	0.1	0.3		2.0	5.3
Fabaceae	Trifolium repens L.	Burn	12.3	1.0	0.0	0.1	0.0	0.1	1.0	3.9	16.0	13.6	1.4	1.4	0.2	0.0	5.0
		Control	0.0	0.8	0.0	0.0	0.1	0.9	0.1	0.1	0.0	0.0	0.0	0.1		1.3	1.4
	Unknown Forb	Burn	0.4	6.0	0.0	0.0	0.4	0.2	0.2	0.1	0.1	0.0	0.6	0.0	0.0	2.8	1.2
		Control	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		1.6	0.1
	Unknown Grass	Burn	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.3	0.1	4.0	2.0	0.1
	Veronica peregrina L.	Control	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0		0.0	0.3
Scrophulariaceae	ssp. xalapensis (Kunth)	Burn	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0
		Control	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.3	0.0
Violaceae	Viola adunca Sm.	Burn	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.9	0.0	0.0	0.2	0.0	0.0	0.1	0.0
	Vicia americana Muhl.	Control	1.6	0.8	0.0	0.0	0.7	2.4	4.6	5.0	2.3	2.9	3.9	2.4		3.1	11.7
Fabaceae	ex Willd. subspecies	Burn	1.0	8.0	0.0	1.6	1.4	4.6	4.9	7.9	13.3	14.0	13.1	5.9	8.5	10.4	22.2

REFERENCES

Allen, C.D. 1989 *Changes in the landscape of the Jemez Mountains, New Mexico.* PhD thesis, University of California Berkeley, Berkeley, CA.

Andariese, S. W. 1982. *Time-response graphs for understory production following fall prescribed burning in Arizona ponderosa pine on basalt soils.* thesis, Northern Arizona University, Flagstaff, AZ.

Barnes, W. S. 2003. *2002 Rangeland Monitoring Report.* Valles Caldera National Preserve, Jemez Springs, New Mexico [Purchase Order No. VCT0207]

Boyer, D. E., Dell, J. D. 1980. *Fire effects on Pacific Northwest forest soils.* Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, Watershed Management and Aviation and Fire Management. 59 p.

Brown, D. E. 1994. *Biotic Communities : Southwestern United States and Northwestern Mexico.*Lake City: University of Utah Press.

Cattelino, P. J. 1980. *A reference base for vegetative response and species reproductive strategies*. Final Report. Supplement No. 10 to Master Memorandum between Intermountain Forest and Range Experiment Station and Gradient Modeling, Inc. Missoula, MT: Gradient Modeling, Inc. 30 p.

Clarke, K.R. & Gorley, R.N. 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E, Plymouth, UK

Collins, S. L., and Glenn, S. M. 1991. Importance of spatial and temporal dynamics in species regional abundance and distribution. *Ecology* 72: 654-664.

Coop, J. D., and Givnish, T. J. 2007. Spatial and temporal patterns of recent forest encroachment in montane grasslands of the Valles Caldera, New Mexico, USA. *Journal of Biogeography* 34: 914-927

Coop, J. D., and Givnish, T. J. 2007. Gradient analysis of reversed treelines and grasslands of the Valles Caldera, New Mexico. *Journal of Vegetation Science* 18: 43-54.

Dahm, C. N., Candelaria-Ley, R. I., Reale, C. S., Reale, J. K., & Van Horn, D. J. 2015. Extreme water quality degradation following a catastrophic forest fire. *Freshwater Biology* doi: 10.1111/fwb.12548

Daubenmire, R. 1968. *Ecology of fire in grasslands*. In: Cragg, J. B., ed. Advances in ecological research: Vol. 5. New York: Academic Press: 209-266.

Debenedetti, S. H., and Parsons, D. J. 1984. Postfire succession in a sierran subalpine meadow. *American Midland Naturalist* 111: 118-125.

Dewar, J.J. 2011. *Fire history of montane grasslands and ecotones of the Valles Caldera, New Mexico, USA*, thesis, University of Arizona

Ford, P. L., McPherson, G. R. 1996. *Ecology of fire in shortgrass prairie of the southern Great Plains.* In: Finch, D. M., Editor. Ecosystem disturbance and wildlife conservation in western grasslands - A symposium proceedings. September 22-26, 1994; Albuquerque, NM. General Technical Report RM-GTR-285. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. p. 20-39.

Gaines, E. M., Kallander, H. R., Wagner, J. A. 1958. Controlled burning in southwestern ponderosa pine: results from the Blue Mountain plots, Fort Apache Indian Reservation. *Journal of Forestry*. 56: 323-327.

Haisley, J. R. 1984. *The effects of prescribed burning on four aspen-bunchgrass communities in northern Arizona*. Northern Arizona University, Flagstaff, AZ.

Harris, G.R., Covington, W.W. 1983. The effect of a prescribed fire on nutrient concentration and standing crop of understory vegetation in ponderosa pine. *Canadian Journal of Forest Research*. 13: 501-507.

Hunter, M.E., Omi, P. N., Martinson, E.J., Chong, G.W. 2006. Establishment of non-native plant species after wildfires: effects of fuel treatments, abiotic and biotic factors, and post-fire grass seeding treatments. *International Journal of Wildland Fire* 15: 271–281.

Hurteau, M. D., Bradford, J. B., Fule, P. Z., Taylor, A. H., and Martin, K. L. 2014. Climate change, fire management, and ecological services in the southwestern US. *Forest Ecology and Management* 327: 280-289.

Keeley, J. E., Pausas, J. G., Rundel, P. W., Bond, W. J., and Bradstock, R. A. 2011. Fire as an evolutionary pressure shaping plant traits. *Trends in Plant Science* 16: 406-411.

Marlon, J.R., Bartlein, P.J., Gavin, D.G., Long, C.J., Anderson, R.S., Briles, C.E., Brown, K.J., Colombaroli, D., Hallett, D.J., Power, M.J., Scharf, E.A., Walsh, M.K. 2012. Long-term perspective on wildfires in the western USA. *Proceedings of the National Academy of Sciences* 109: E535–E543.

Mueggler, W. F. & Stewart, W. L. 1980. *Grassland and shrubland habitat types of western Montana.* U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. [Gen. Tech. Rep. INT-66], Ogden, UT

Muldavin, E. 2003. *A Vegetation Survey and Preliminary Ecological Assessment of Valles Caldera National Preserve, New Mexico.* New Mexico Natural Heritage

Program, University of New Mexico, Albuquerque, New Mexico [Cooperative Agreement No. 01CRAG0014]

National Cooperative Soil Survey, Natural Resources Conservation Service, United States Department of Agriculture. 1987. *Soil Survey of Sandoval County Area, New Mexico.*

Oswald, B. P. Covington, W. W. 1984. Effect of a prescribed fire on herbage production in southwestern ponderosa pine on sedimentary soils. *Forest Science* 30(1): 22-25.

Parmenter, R. 2008. Long-Term Effects of a Summer Fire on Desert Grassland Plant Demographics in New Mexico. *Rangeland Ecology and Management* 61: 156-168

Risser, P. G., Birney, E. C., Blocker, H. D.; [and others]. 1981. *The true prairie ecosystem.* US/IBP Synthesis Series 16. Hutchinson Ross Publishing Company, Stroudsburg, PA: 557 p.

Sackett, S.S., Haase, S. M., Harrington, M. G. 1996. *Lessons learned from fire use for restoring southwestern ponderosa pine ecosystems.* In: Covington, Wallace; Wagner, Pamela K., technical coordinators. Conference on adaptive ecosystem restoration and management: restoration of Cordilleran conifer landscapes of North America: Proceedings; 1996 June 6-8; Flagstaff, AZ. Gen. Tech. Rep. RM-GTR-278. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 54-61.

Santa Fe National Forest and Valles Caldera National Preserve. 2010. *Southwest Jemez Mountains Collaborative Forest Landscape Restoration Proposal.* Jemez Springs, New Mexico

Scheiner, S.M. 1988. Population Dynamics of an Herbaceous Perennial Danthonia spicata during Secondary Forest Succession. *American Midland Naturalist* 119: 268-281.

Scheintaub, M. R., Derner, J. D., Kelly, E. F., and Knapp, A. K. 2009. Response of the shortgrass steppe plant community to fire. *Journal of Arid Environments* 73: 1136-1143.

Scheiter, S. Higgins, S.I., Osborne, C. P., Bradshaw, C., Lunt, D., Ripley, B.S., Taylor, L.L., Beerling, D. J. 2012. Fire and fire-adapted vegetation promoted C4 expansion in the late Miocene. *New Phytologist* 195: 653-666.

Servis, S., Boucher, P.F. 1999. *Restoring fire to southwestern ecosystems: is it worth it?* In: Gonzalez-Caban, Armando; Omi, Philip N., tech. coords. Proceedings of the symposium on fire economics, planning, and policy: bottom lines; 1999 April 5-9; San Diego, CA. Gen. Tech. Rep. PSW-GTR-173. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 247-253. [37051]

Smith, M. A., Busby, F. 1981. *Prescribed burning: effective control of sagebrush in Wyoming*. RJ-165. Laramie, WY: University of Wyoming, Agricultural Experiment Station. 12 p.

Spell, T.L., Kyle, P.R., Thirlwall, M.F., and Campbell, A., 1993, Isotopic and geochemical constraints on the origin and evolution of postcollapse rhyolites in the Valles Caldera, New Mexico. *Journal of Geophysical Research* 98: 723-739.

Tilman, D., Downing, J. A. 1994. Biodiversity and stability in grasslands. *Nature* 367: 363-365.

Tix, D., Charvat, I. 2005. Aboveground biomass removal by burning and raking increases diversity in a reconstructed prairie. *Restoration Ecology* 13: 228–238.

Van Auken, O.W. 2009. Causes and consequences of woody plant encroachment into western North American grasslands. *Journal of Environmental Management* 90: 2931-2942.

Vermeire, L. T., Crowder, J. L., and Wester, D. B. 2014. Semiarid Rangeland Is Resilient to Summer Fire and Postfire Grazing Utilization. *Rangeland Ecology and Management* 67: 52-60.

Vose, J. M., White, A. S. 1987. Processes of understory seedling recruitment 1 year after prescribed fire in an Arizona ponderosa pine community. *Canadian Journal of Botany* 65: 2280-2290.

Wright, H. A., Neuenschwander, L. F., Britton, C. M. 1979. *The role and use of fire in sagebrush-grass and pinyon-juniper plant communities: A state-of-the-art review*. Gen. Tech. Rep. INT-58. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 48 p.

Wright, H. A. Bailey, A. W. 1980. *Fire ecology and prescribed burning in the Great Plains: a research review.* General Technical Report INT-GTR-77. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 60 p.

Xiong, S., Nilsson, C. 1999. The effects of plant litter on vegetation: a meta-analysis. *Journal of Ecology* 87: 984-994

York, A.M., Shrestha, M., Boone, C.G., Zhang, S., Harrington Jr., J.A., Prebyl, T.J., Swann, A., Agar, M., Antolin , M.F., Nolen, B., Wright , J.B. & Skaggs , R. 2011.

Land fragmentation under rapid urbanization: a cross-site analysis of Southwestern cities. *Urban Ecosystems* 14: 429-455.