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ABSTRACT

A historic return interval of 100 years for large fires in the U.S. southwestern deserts is being replaced by one where fires may reoccur as frequently as every 20 to 30 years. The shortened return interval, which translates to an increase in fires, has implications for management of Soil Corrective Action Units (CAUs) and Corrective Action Sites (CASs) for which the Department of Energy, National Nuclear Security Administration Nevada Field Office has responsibility. A series of studies was initiated at uncontaminated analog sites to better understand the possible impacts of erosion and transport by wind and water should contaminated soil sites burn. The first of these studies was undertaken at the Jacob Fire site approximately 12 kilometers (7.5 miles) north of Hiko, Nevada. A lightning-caused fire burned approximately 200 hectares during August 6-8, 2008. The site is representative of a transition between Mojave and Great Basin desert ecoregions on the Nevada National Security Site (NNSS), where the largest number of Soil CAUs/CASs are located. The area that burned at the Jacob Fire site was primarily a *Coleogyne ramosissima* (blackbrush) and *Ephedra nevadensis* (Mormon tea) community, also an abundant shrub assemblage in the similar transition zone on the NNSS. This report summarizes three years of measurements after the fire.

Seven measurement campaigns at the Jacob Fire site were completed. Measurements were made on burned ridge (upland) and drainage sites, and on burned and unburned sites beneath and between vegetation. A Portable In-Situ Wind Erosion Lab (PI-SWERL) was used to estimate emissions of suspended particles at different wind speeds. Context for these measurements was provided through a meteorological tower that was installed at the Jacob Fire site to obtain local, relevant environmental parameters. Filter samples, collected from the exhaust of the PI-SWERL during measurements, were analyzed for chemical composition. Runoff and water erosion were quantified through a series of rainfall/runoff simulation tests in which controlled amounts of water were delivered to the soil surface in a specified amount of time. Runoff data were collected from understory and interspace soils on burned ridge and drainage areas. Runoff volume and suspended sediment in the runoff were sampled; the particle size distribution of the sediment was determined by laboratory analysis. Several land surface and soil characteristics associated with runoff were integrated by the calculation of site-specific curve numbers. Several vegetation surveys were conducted to assess post-burn recovery. Data from plots in both burned and unburned areas included species identification, counts, and location. Characterization of fire-affected area included measures at both the landscape scale and at specific sites.

Although wind erosion measurements indicate that there are seasonal influences on almost all parameters measured, several trends were observed. PI-SWERL measurements indicated the potential for PM_{10} windblown dust emissions was higher on areas that were burned compared to areas that were not. Among the burned areas, understory soils in drainage areas were the most emissive, and interspace soils along burned ridges were least emissive. By 34 months after the burn (MAB), at the end of the study, emissions from all burned soil sites were virtually indistinguishable from unburned levels. Like the amount of emissions, the chemical signature of the fire (indicated by the EC-Soil ratio) was elevated immediately after the fire and approached pre-burn levels by 24 MAB. Thus, the potential for wind erosion at the Jacob Fire site, as measured by the amount and type of emissions, increased significantly after the fire and returned to unburned levels by 24 MAB.

The effect of fire on the potential for water erosion at the Jacob Fire site was more ambiguous. Runoff and sediment from ridge interspace soils and unburned interspace soils were similar throughout the study period. Seldom, if ever, did runoff and sediment occur in burned drainage area soils. For burned soils where runoff occurred at 1 MAB, the sediment size was finer than on unburned sites, but this effect disappeared by 3 MAB. For the three-year study under the conditions tested at the Jacob Fire site, the potential for water erosion appeared relatively unaffected by the fire.

Vegetation responses were documented for each year following the fire. By the end of the study, there was a substantial difference in plant densities and richness between drainage and ridge sites. Cheatgrass densities were higher in unburned plots, and cheatgrass was also more dominant in the community composition in unburned plots. Cheatgrass had increased in the burned area but so did other native species. Three years after the fire, the burned landscape continued to revegetate but had yet to approximate the condition of an unburned landscape. The results from the vegetation surveys support the wind erosion results, where the primary source of windborne particles originate from the understory, where lower plant diversity and densities were found. The soil appears to be more resilient and have a much shorter recovery time than the vegetation in this particular community.

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The authors gratefully acknowledge the input and assistance of the following: S. Campbell, W. Forsee, J. King, W. Meyer, G. Nikolich, and S. Zitzer. Site access was provided by the Bureau of Land Management and gratefully acknowledged. The thoughtful comments of P. Verburg and M. Berli improved the report and are greatly appreciated.

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LIST OF ACRONYMS

µm	micrometer
ac	acre
C	Celsius
CAS	Corrective Action Site
CAU	Corrective Action Unit
CCRFD	Clark County Regional Flood Control District
cm	centimeter
CN	Curve Number
DOE	U.S. Department of Energy
EC	elemental carbon
F	Fahrenheit
ft	ft
g	gram
ha	hectare
I_a	Initial abstraction
IMPROVE	Interagency Monitoring of PROtected Visual Environments
in	inch
kph	kilometers per hour
l	liter
m	meter
MAB	months after the burn
mg	milligram
ml	milliliter
mm	millimeter
mph	miles per hour
NAEG	Nevada Applied Ecology Group
NFO	Nevada Field Office
NNSA	National Nuclear Security Administration
NNSS	Nevada National Security Site
NOAA	National Oceanographic and Atmospheric Administration
NTTR	Nevada Test and Training Range
PI-SWERL	Portable In-Situ Wind ERosion Lab
PM	particulate matter
PM_{10}	particulate matter with diaµ less than 10 micrometers
RPM	revolutions per minute
SCS	Soil Conservation Service

SORD	Special Operation and Research Division
SSI	size selective inlets
TTR	Tonopah Test Range
U.S.	United States
USDA	U.S. Department of Agriculture
WRCC	Western Regional Climate Center

INTRODUCTION

Background

Fire is a natural but significant process in the desert ecosystem. Between 1998 and 2008, an average of nearly 850,000 hectares (ha; 2,100,396 acres, ac) per year burned in the Great Basin, including more than one million ha (2,471,054 ac) in 2007 alone (Chambers *et al*., 2008). A series of 11 fires burned approximately 300,000 ha (741,316 ac) in southern Nevada and Utah, including 206,000 ha (509,037 ac) in Nevada between June 22 and July 10, 2005. These Southern Nevada Complex Fires were lightning-caused, but followed a period of above average precipitation in the preceding fall and winter resulting in the accumulation of combustible fuel from a significant increase in invasive grasses and annuals (National Interagency Coordination Center, 2006). Though smaller in extent, the lightning-sparked Air Force Fire of early June 2005 burned approximately 8,000 ha (19,768 ac) of the Nevada Test and Training Range (NTTR) and 2,400 ha (5931 ac) on the west side of the Nevada National Security Site (NNSS) (Nevada Nuclear Security Administration, 2005). During the course of that fire, there was a concern that some contaminated soil sites on Buckboard Mesa and Dome Mountain on the NNSS could burn, which ultimately did not occur.

It is generally acknowledged that since the mid-1980s the Southwest United States (U.S.) fire regime is changing. An increase in occurrence and extent of fire has been associated with the spread of invasive annual grasses, most significantly cheatgrass (*Bromus tectorum*) and red brome (*Bromus rebens*). Both grasses were introduced to the region as part of Euro-American settlement and cattle grazing in the nineteenth century (Knapp, 1996). Although both species were found on a few disturbed sites on the NNSS as early as the 1960s, they spread across the region during a period of above-average precipitation in the 1980s (Hunter, 1991; Rickard and Beatley, 1965). Today, these grasses can rapidly invade disturbed areas and in many plant communities they have colonized interspaces between shrubs, increasing the total fuel load and allowing fires to move more easily between shrubs (Knapp, 1996). Besides increasing the chance of a fire occurring and the likelihood that it will burn larger areas than historic fires, Hansen and Ostler (2004) documented that invasive plants are quick to colonize areas that have burned on the NNSS, increasing the chance that subsequent fires will occur.

The changing fire regime has been correlated to an increase in average spring and summer temperatures, resulting in the loss of soil moisture earlier in the year and longer periods of dry plant surface biomass. Regionally, average spring and summer temperatures for the period 1987 to 2003 were 0.87 degrees Celsius (ºC) [1.5 degrees Fahrenheit, °F] higher than the period 1970 to 1986 (Westerling *et al*., 2006). Other factors contributing to the changing fire regime include drought conditions across large areas of the region since 1999 and accumulated fuel loads due to wildfire suppression (Savage and Swetnam, 1990; Neary *et al*., 1999; Westerling and Swetnam, 2003; Westerling *et al*., 2006). As a consequence, a historic return interval of 100 years for large fires in the U.S. southwestern deserts is being replaced by one where fires may reoccur as frequently as every 20 to 30 years (Brooks, 2006).

The effect of wildfires on the soil stability of sites for which the Department of Energy (DOE) has responsibility is not well known. The DOE's Nevada Applied Ecology Group (NAEG) examined a variety of disturbance mechanisms at sites where plutonium isotopes were among the most significant contaminants. The majority of these studies were carried out in the 1970s, before cheatgrass (*B. tectorum*) and red brome (*B. rubens*) spread throughout the NNSS, the Tonopah Test Range (TTR), and the NTTR. Fires were not considered because at that time they were comparatively infrequent and small in size (Bruce Church and Lynn Anspaugh, *personal communication*, 2007).

Implications

Fires and the changing fire regime have important implications for post-closure management and long-term stewardship of Soil Corrective Action Units (CAUs) and Corrective Action Sites (CASs), areas for which the DOE Nevada Nuclear Security Administration/Nevada Field Office (NNSA/NFO) has regulatory closure responsibility. For many Soil CAUs and CASs, where closure-in-place options are now being considered or implemented, there is an increasing chance they could experience a fire while the contaminated soils still pose a risk, especially considering the long half-life of some radionuclides such as plutonium-239 (Shafer *et al*., 2007; Shafer and Gomes, 2009). There is a need to better understand the possible impacts of fire-induced erosion and particle transport by wind and water, the response of vegetation post-fire, and the risks and perceived risks to site workers and public receptors in communities around the NNSS, TTR, and NTTR. Ultimately, there is a need to develop recommendations for stabilization and restoration of contaminated sites, should they burn.

Objective

A better understanding of the fire-related potential for erosion of contaminated soil is needed for management and stewardship of Soil CAUs/CASs. While it would be ideal to gather data at a radionuclide contaminated site, it is more practical to examine fires and their effects in “analog” environments similar to Soil CAUs and CASs. This report is the first in a series of studies initiated by DOE and carried out by the Desert Research Institute, which assess the effect of fire on wind and water erosion and on vegetation response to fire throughout the NNSS. This report addresses a three-year monitoring effort undertaken with the specific objective to assess the effects of fire on the potential for wind and water transport of particles and vegetation response post-fire in the transition region between the Great Basin and Mojave Desert regions of the NNSS. This region, the focus of this report, is where most of the CASs on the NNSS, specifically those within northern Yucca Flat, are found.

BACKGROUND, FIRE AND SITE DESCRIPTIONS, AND MEASUREMENT CAMPAIGNS

The Mojave-Great Basin Desert Transition Region of the Nevada National Security Site

Ostler *et al.* (2000) and earlier workers (most prominently, Beatley, 1976) identified three ecological regions on the NNSS: the Mojave Desert in the southern portion of the NNSS, the Great Basin Desert in the northern portion, and the Mojave-Great Basin Transition between them (Figure 1). The transition region between Mojave and Great Basin represents 36.6 percent of the total land area of the NNSS (Ostler *et al*., 2000). The average annual temperature from a representative meteorological station (BJY) is 14°C (57°F) and

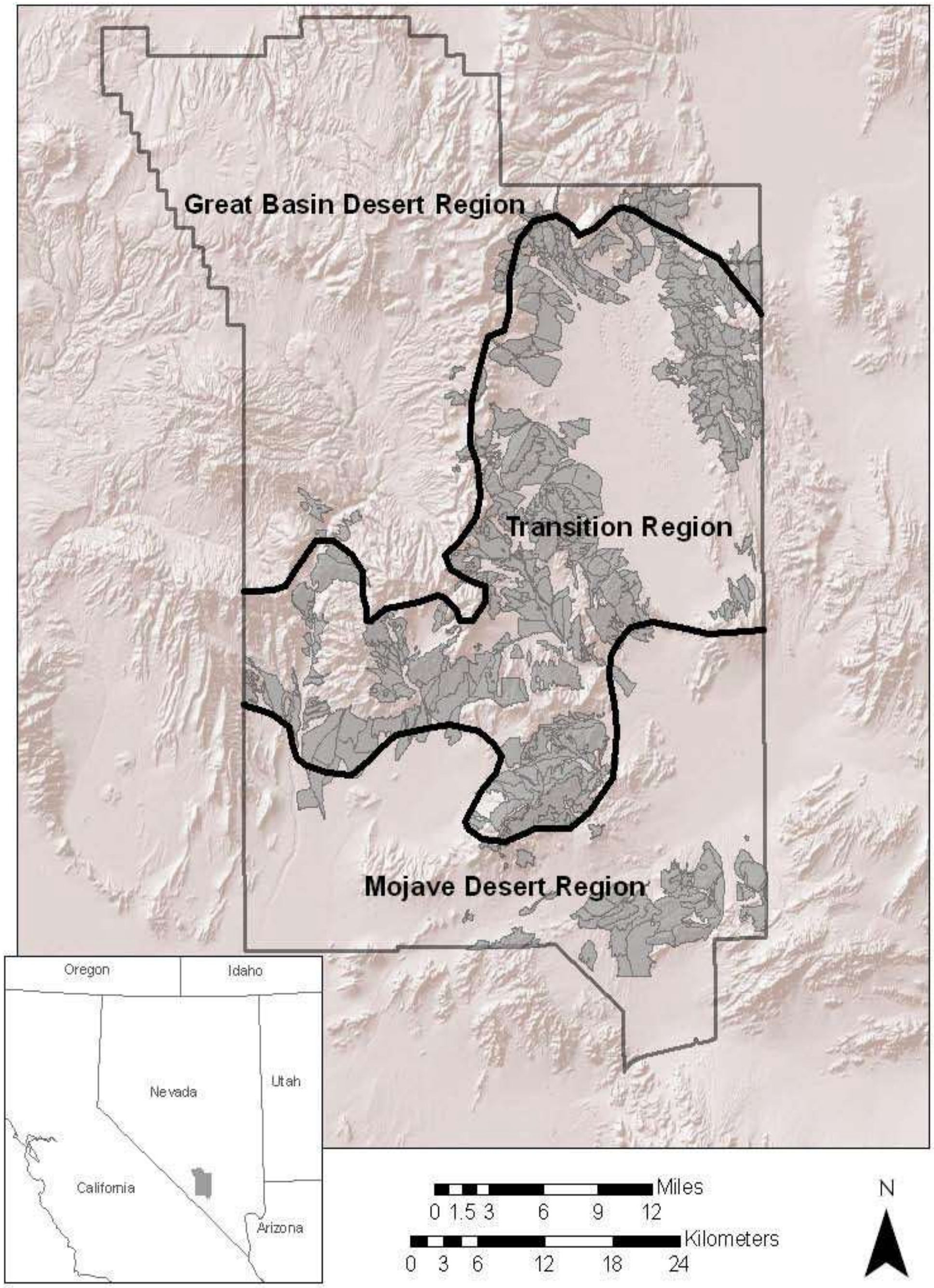


Figure 1. Ecoregions of the NNSS. Shaded areas indicate the distribution of the *Coleogyne ramosissima* Torr./*Ephedra nevadensis* (blackbrush/Mormon tea) shrub alliance. Although not confined exclusively to it, the shrubland is indicative of the Great Basin Desert-Mojave Desert Transition ecoregion on the NNSS (Hansen and Ostler, 2004).

ranges from a minimum average of 5°C (41°F) in December to a maximum average of 23°C (97°F) in July. Precipitation averages 16.0 centimeters (cm; 6.3 inches, in), 48 percent of which occurs during the cold-season months of December, January, February, and March (National Oceanic and Atmospheric Administration/Special Operations and Research Division [NOAA/SORD], 2007). Here, the dominant vegetation community is the blackbrush (*Coleogyne ramosissima* Torr.)/Mormon tea (*Ephedra nevadensis*) shrub alliance which in some areas consists of nearly pure stands of blackbrush. Blackbrush communities are one of the dominant vegetation community types in southern Nevada, occurring between 1220 meters (m; 3937 feet, ft) and 2000 m (6265 ft) in elevation above the creosote bush-bursage communities on the lower mountain slopes and valley floors, and below the pinyon-juniper-sagebrush communities that occur higher on the mountain slopes. Blackbrush is a native rosaceous perennial shrub which has a relatively slow growth rate limited by soil moisture associated with the shallow, gravelly soils on which it tends to associate (Brooks, 2005; Lei and Walker, 1995). Plant diversity in blackbrush communities is comparable to other major vegetation types although invasion by non-native species is increasingly reported.

Although blackbrush/Mormon tea occurs elsewhere in the southern part of the NNSS, it is predominant on mid- to upper piedmont slopes surrounding Yucca Flat as well as Mid Valley and Topopah Valley. The Yucca Flat shrublands were frequently disturbed by nuclear testing and include areas that are now part of Soils CASs of Northern Yucca Flat, portions of Plutonium Valley, as well as the west side of the basin. Although the blackbrush/Mormon tea community develops most frequently on Tertiary volcanic tuffs elsewhere in the southwest, it also forms on limestone-dominated substrates on the NNSS. On the NNSS, the blackbrush community occurs primarily in areas of undisturbed shallow soils on Quaternary alluvial and colluvial fans with moderate desert pavement development. Soils there are generally gravelly sandy loams to loams.

Over the past 30 years, blackbrush communities have become increasingly susceptible to wildfire due to drought and invasion by non-native annual grasses (Abella, 2009; Rew *et al*., 2010). Fire has long-term effects on these communities which experience little regeneration (Callison *et al*. 1985; Brooks and Matchett 2003; Abella *et al*., 2009). The lack of regeneration is often due in part to the rapid post-fire establishment of non-native annual grasses that are capable of perpetuating a grass/fire cycle which eliminates initial blackbrush regeneration that may have occurred after the first fire cycle (D'Antonio and Vitosek, 1992). Unlike other shrub species, blackbrush does not sprout from roots after fire and experiences slow recruitment rates (Humphrey, 1974; Brooks, 1995). High intensity wildfires have the ability to kill most of the native soil seed bank including blackbrush (Esque *et al*., 2010). Finally, changes to the soil chemistry and physical and hydrological properties due to fire may also play a role in the minimal post-fire re-establishment of blackbrush (Lei, 1999). Recovery times for blackbrush after fires vary, however it may take 50 to 100 years for the community to recover (Sugihara, 2006). This is significant because most wildland fires on the NNSS have occurred in the blackbrush zone (Hansen and Ostler, 2004). The severity of fires associated with blackbrush-dominated shrublands is classified as "replacement", where greater than 75 percent of crown fuel is removed (Medlyn, 2005). Because it is considered one of the most flammable native plant assemblages in the Mojave Desert, it is considered to be a hazardous fuel (Brooks, 2005).

The Jacob Fire and Site

The Jacob fire was caused by lightning and the Jacob fire study site is located approximately 16 kilometers north of Hiko, Nevada (37°42'17" North, 115°12'41" West) on land managed by the U.S. Bureau of Land Management (BLM). From August 6 to 8, 2008, approximately 186 ha (460 ac) were burned (Figure 2). No fire intensity data were available for the site. However, fires on mature stands and late seral patches of blackbrush which completely consume the vegetation are generally associated with moderate to high intensity fires (Brooks *et al*., 2007).

Among the reasons the Jacob fire was selected as an analog fire study site for the Mojave-Great Basin Transition ecoregion of the NNSS were:

- like the Mojave-Great Basin transition region of the NNSS, the area burned was dominated by blackbrush and Mormon tea.
- the elevation range of the fire, 1,200 to 1,500 m (approximately 3,900 to 4,900 ft), is similar to those at the margins of the Yucca Flat on the NNSS.
- the alluvial/colluvial material on which the fire occurred is dominated by clasts derived from volcanic tuffs (Taylor and Bartley, 2002).
- annual precipitation is similar. Annual precipitation at station BJY (1,241 m, 4,071 ft) at the north end of Yucca Flat on the NNSS averaged 16.0 centimeters (cm, 6.3 in) based on records from 1960 to 2006 (NOAA/SORD, 2007). Although there is no long-term record available for the Jacob fire site proper, the town of Hiko, Nevada (16 kilometers, 10 miles, south of the Jacob Fire site) averaged 17.0 cm (6.7 in) annually between 1999 and 2009 (Western Regional Climate Center, 2011), with a similar percentage (18 percent) occurring in the summer as at BJY on the NNSS (17 percent).

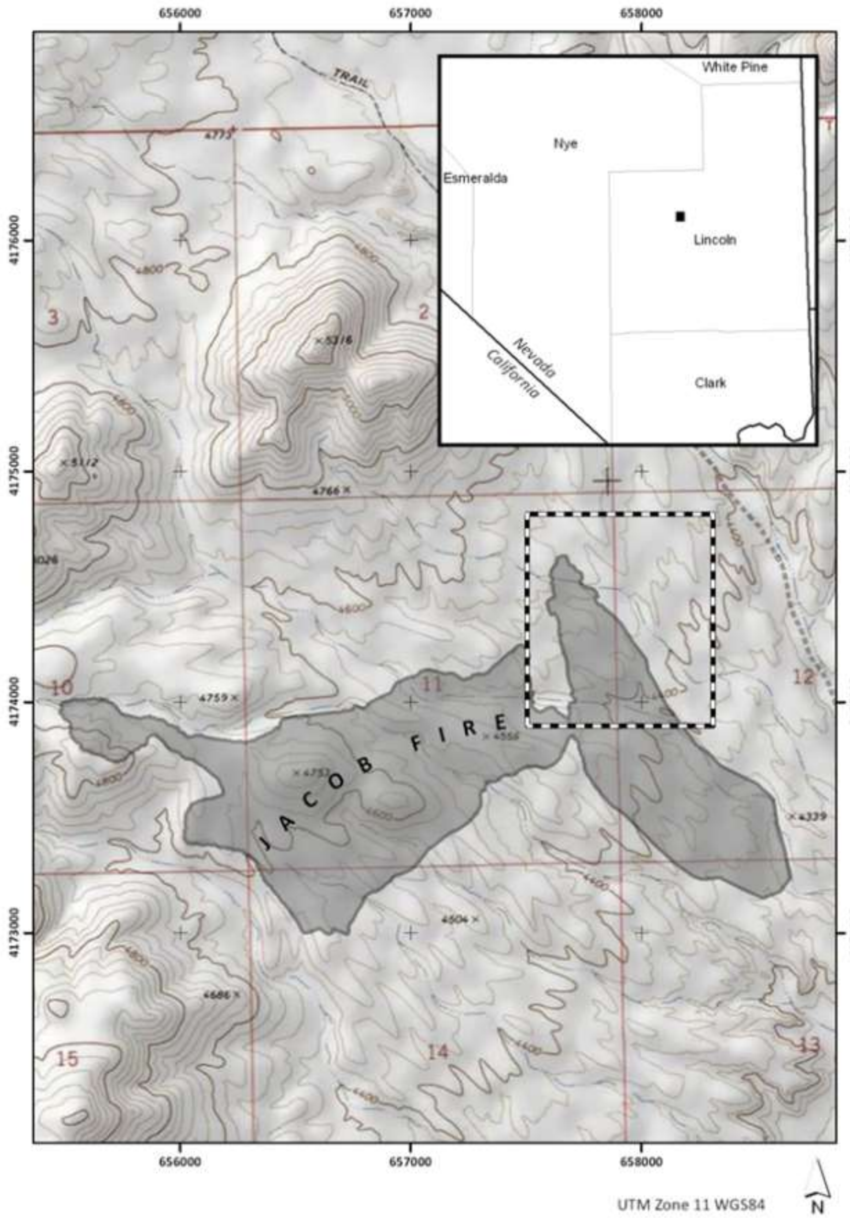


Figure 2. Study area location within Lincoln County, Nevada and 2008 Jacob Fire outline. Study plots were located within the area outlined by the dashed line.

Site Selection and Field Measurement

The effects of wildfire on soil vary both temporally and spatially. Therefore, after the August, 2008 wildfire, several field measurement campaigns were undertaken to monitor the effects of the wildfire over a three-year period from September, 2008 to June, 2011. These measurements began one month after the burn (1 MAB). Actual field measurement dates varied due to logistical and weather-related conditions, as indicated in Appendix A.

Topographic and vegetation-related soil features are important to assess source areas for wind and water erosion generally, and erosion due to fire, specifically (Shafer *et al*., 2007, Chief *et al*., 2011). At the Jacob Fire site, all vegetation was burned leaving only the occasional skeletons of larger shrubs. No vegetation canopies existed after the fire, although charred soil surfaces beneath burned plant canopies and uncharred surfaces were evident (Figure 3).

A general experimental area was established approximately one month after the fire. The area was selected such that burned and unburned areas were in close proximity to each other, were representative of local geomorphology and, for logistical reasons, were close to an access road. Sampling of the burn area was stratified by vegetation coverage (vegetated canopy understory or bare interspace) and landscape position (ridge or drainage). Measurements were made on unburned soil sites to provide a baseline (control) from which fire effects could be assessed. However, due to the need for proximity to roadway access, the unburned control area was representative of the burned ridge area only. This resulted in a matrix of six soil sampling sites: burned ridge understory sites; burned ridge interspace sites; burned drainage understory sites; burned drainage interspace sites; unburned ridge understory sites; and, unburned ridge interspace sites.

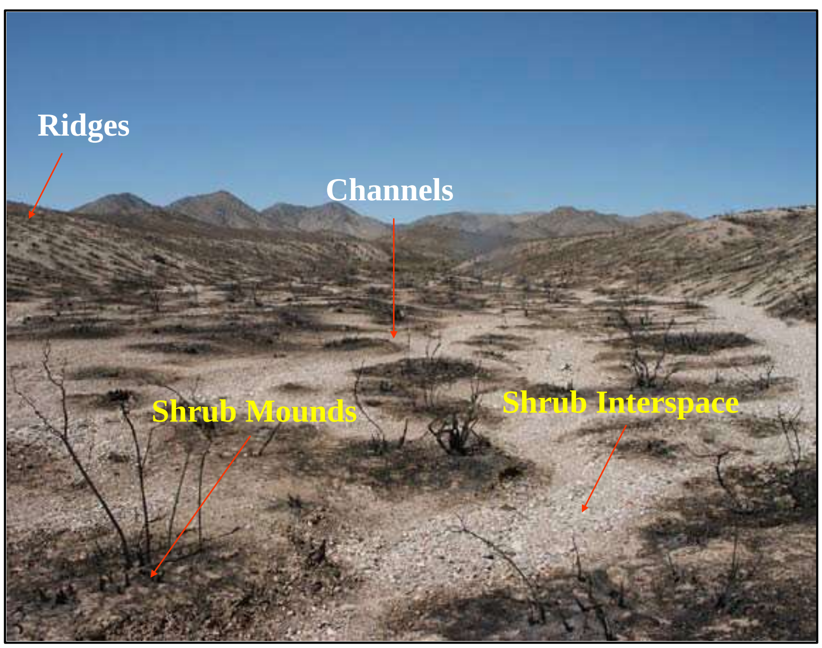


Figure 3. The Jacob Fire site.

Wind erosion measurements were collected on burned understory and burned interspace areas on both ridge and drainage areas, and on unburned ridge sites. A portable wind tunnel was used to estimate suspended particle emissions at different wind speeds. Context for these measurements was provided through data from a meteorological tower installed at the Jacob Fire site to obtain local wind speed, wind direction, and other environmental parameters necessary for assessing environmental airborne particulate matter throughout the duration of the study.

Runoff and water erosion were quantified through a series of rainfall/runoff simulation tests, where controlled amounts of water were applied to the soil surfaces with a device that simulated rainfall. Generated runoff and sediment were collected and analyzed. Soil moisture was measured, and soil texture was determined from the sampled sites.

Vegetation surveys included species identification and counts, and locations of sampled sites. Six vegetation surveys in 50 cm x 50 cm (19.6 in x 19.6 in; 0.25 $m^{2,}$ 2.7 ft^2) plots were conducted throughout the period 2009 - 2011. Plots were located in each of the six sample categories identified above. Individual species and the number of individual plants by species were recorded in each plot.

WIND EROSION POTENTIAL AND CHEMISTRY OF A BURN SITE: MEASUREMENTS AND RESULTS

Measurements

Meteorological Instruments

A meteorological station was installed at the Jacob Fire site in October, 2008 (3 MAB) to monitor wind and other environmental parameters. The station consisted of a three-meter (m) post, mounted on a tripod, and anchored to the ground for stability. The station was instrumented to measure ambient air temperature and relative humidity, wind speed and direction, precipitation, soil temperature at 1.0 cm (0.4 in), and soil volumetric water content near the surface (0 – 10 cm, 0 – 4 in). All instruments were scanned once every five seconds and data were processed and stored on a datalogger at ten minute intervals. The datalogger had sufficient on-board memory to accommodate three months of continuous operation and the entire station was powered by a solar panel and a battery to provide power overnight.

Wind Erosion Measurements

A Portable In-Situ Wind ERosion Lab (PI-SWERL) was used to measure the potential for dust emission and wind erosion from the soil surface (Etyemezian *et al*., 2007). Specifically, the PI-SWERL was used to estimate abundance and composition of airborne particulate matter (PM) with aerodynamic diameters less than 10 micrometers (µm, PM_{10}). Five types of soil sites were sampled at the Jacob Fire site: burned ridge understory soils, burned ridge interspace soils, burned drainage understory soils, burned drainage interspace soils, and unburned soils. (Measurements were collected only on unburned interspace soils, as unburned plant understory sites precluded instrumentation.) There were four replicates on each burned soil type, and eight replicates on the unburned soils. The chemical composition of the emissions collected on exhaust filters was also analyzed. Due to concerns about of the emissions collected on exhaust filters was also analyzed. Due to concerns about minimum loading, these filter samples were aggregated into burned ridge, burned drainage, and control soil samples.

For all measurements, a hybrid ramp/step measurement cycle was used. The cycle consisted of:

1. a 60 second clean air flush,
2. sharp acceleration to 500 revolutions per minute (RPM),
3. a 60 second linear ramp to 2,000 RPM,
4. maintain 2,000 RPM for 60 seconds,
5. 60 second ramp to 3,000 RPM,
6. maintain 3,000 RPM for 90 seconds,
7. 60 second ramp to 4,000 RPM,
8. maintain 4,000 RPM for 90 seconds, and
9. turn off motor and clean air flush for 60 seconds.

Each value of RPM corresponds to a friction velocity, u_*, that is a measure of the amount of wind shear applied to a soil surface. Friction velocity can be related to surface wind speed (measured at a height above ground level) and the surface roughness with the Prandtl Equation:

$$\frac{u_{ref}}{u_*} = \frac{1}{\kappa} \cdot \ln\left(\frac{z_{ref}}{z_0}\right) \tag{1}$$

where u_{ref} is the wind speed measured at a reference height z_{ref} (usually 10 m, 32 ft), κ is the von Karman constant equal to 0.41, and z_0 is the aerodynamic roughness height. The roughness height determines how the wind speed translates into shear stress at the soil surface. Values of z_0 vary depending on the physical roughness of a surface. Typical values are: 0.2 millimeters (mm; 0.008 in) for smooth ice, 30 mm (1.2 in) for grasslands, 500 mm (19.7 in) for urban areas, and 5 mm (0.2 in) for flat desert terrain.

Cumulative emissions (grams per square meter, g/m^2) were calculated at several different points during the PI-SWERL test cycle (Figure 4). This parameter answers the questions "How much PM_{10} is available for emissions when the soil is exposed to specific wind speeds?"

Cumulative emissions were calculated at the end of the ramp to 2,000 RPM, at the end of the 2,000 RPM step, at the end of the ramp to 3,000 RPM, at the end of the 3,000 RPM step, at the end of the ramp to 4,000 RPM, and at the end of the 4,000 RPM step. Values of cumulative emissions at each of these points were averaged over replicate measurements for the same types of location. For example, for all PI-SWERL measurements completed on burned drainage soils, the cumulative emissions at the end of the ramp to 2,000 RPM were averaged together. For reference, assuming that the roughness height for a

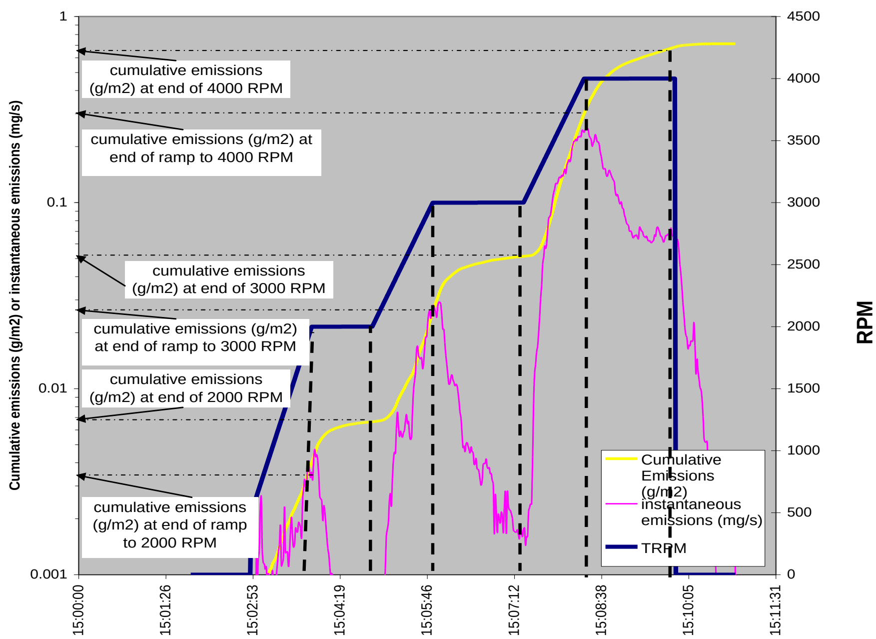


Figure 4. Cumulative emissions (g/m^2) calculated at different points during the PI-SWERL measurement cycle.

desert surface is 5 mm, 2,000 RPM translates roughly to a sustained wind speed of approximately 27 kilometers per hour (kph; 17 miles per hour, mph), 3,000 RPM translated to 39 kph (24 mph), and 4,000 RPM translated to 48 kph (30 mph).

<u>Filter Samples and Analysis</u>

Filter samples were collected at the exhaust port of the PI-SWERL chamber. The filter sampling apparatus included size selective inlets (SSI) to collect only particles in the PM_{10} size range, a flow control valve to ensure that flow rates (five liters per minute) were appropriate for the correct operation of the SSI, a filter holder, and suction source to maintain flow through the filters. Particles were sampled onto two types of filters (Teflon® and quartz-fiber) in each case. Teflon® filters were subjected to gravimetric analysis (for particle mass) and X-ray fluorescence spectroscopy to quantify the elemental composition. A portion of the quartz-fiber filter was analyzed for elemental carbon, organic carbon and carbonate using a thermal/optical carbon analyzer that is based on the preferential oxidation of organic carbon and elemental carbon (EC) compounds at different temperatures. Major water-soluble anions (chloride, nitrate and sulfate) and cations (sodium, magnesium, ammonium, calcium and potassium) were analyzed with an ion chromatography system.

In order to ensure adequate filter loading, one filter set was used for each of the three types of soil surfaces measured (unburned, burned ridge, burned drainage). Therefore, each set of filters represented a composite of multiple (eight) PI-SWERL measurements. A sample of filter data used for analysis to 13 MAB is included in Appendices B and C.

Chemical characteristics of PM_{10} samples are reported in terms of emission profiles or normalized abundances, where the mass of a compound or element on the filter sample is divided by the total mass of PM_{10} particles. This type of normalized concentration is more appropriate for examining differences in chemical composition between samples than the absolute mass of a constituent.

Results

Meteorology

Meteorological data were summarized by monthly average, minimum, and maximum 10-minute values. Data were lost for most of December, 2008 and May, 2009 as well as the first part of June, 2009. Nevertheless, temperature and relative humidity at the Jacob site (Figure 5 and Figure 6) followed expected trends for the region, with temperatures peaking June through August (maximum values around 40 °C, 104 °F) and exhibited sub-zero minima during the months of November through April. November through February had the highest average relative humidity, with March and April relative humidities also slightly elevated compared to the rest of the year. Soil temperature temporal trends (Figure 7) followed ambient temperature trends. Soil water content (Figure 8), in contrast, generally followed ambient temperature trends, but peak monthly soil water content and, to a lesser extent, average soil water content were clearly influenced by the amount of precipitation. For example, elevated values of peak soil water content in February and July 2009 were clearly a result of precipitation during those months.

The average and maximum 10-minute wind speeds on a monthly basis are shown in Figure 9 and a wind rose diagram illustrating the frequency and magnitude of winds from specific directions is shown in Figure 10. March to May, were the windiest months with 10-minute maximum winds near 56 kph (35 mph). In June, 2009 and February, 2011, 10-minute peak winds exceeded 72 kph (45 mph), indicating that short-duration high winds are possible during seasons other than the spring. As discussed in a later section, substantial dust emissions, especially from burned areas, can occur at sustained wind speeds of as little as 39 kph (24 mph) when the soil surface is dry enough.

Surface winds at the Jacob site over the measurement period were predominantly north-northeast and southwest, with an infrequent, but notable northwestern component. Wind speeds were rarely higher than 32 kph (20 mph) and were light to moderate the majority of the time.

PI-SWERL Measurements

Figure 11 through Figure 18 show the cumulative PM_{10} emissions as measured by the PI-SWERL at 1, 3, 6, 13, 21, 24, 34 and 36 MAB at the Jacob fire site. These data show several significant trends. First, in almost all cases, emissions from the unburned control sites were the lowest among all of the surfaces tested. Second, emissions from burned drainage understory soils were generally higher than emissions from burned understory soils on ridges. Third, emissions from drainage interspace soils were also generally higher than emissions from ridge interspace soils. Fourth, both for drainage and ridge locations, emissions from burned understory soils were higher, sometimes by more than an order of

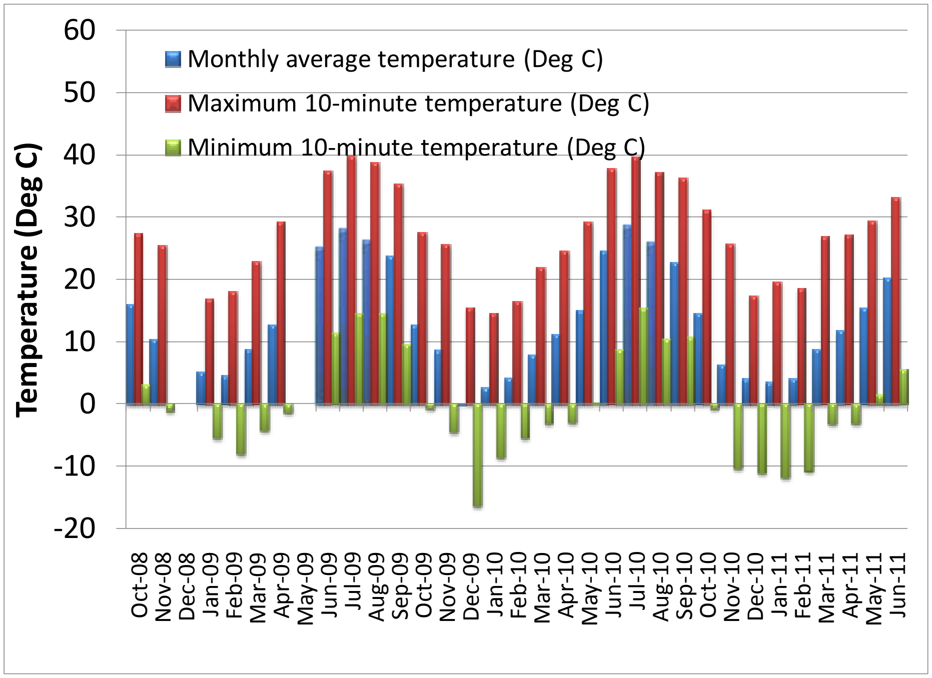


Figure 5. Monthly average air temperature (Deg C) at Jacob site

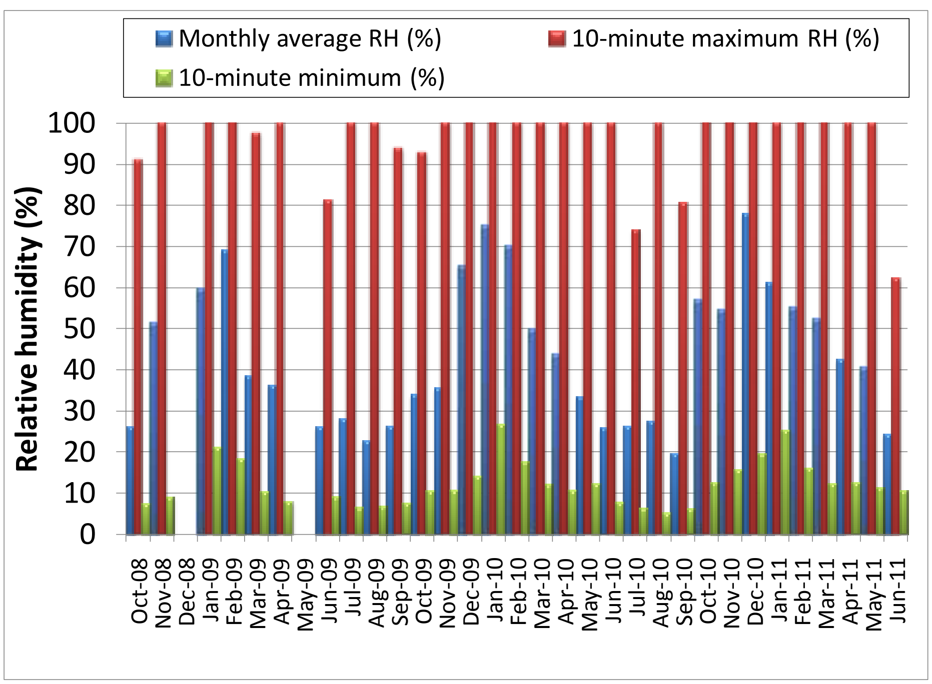


Figure 6. Monthly relative humidity (RH) at Jacob site

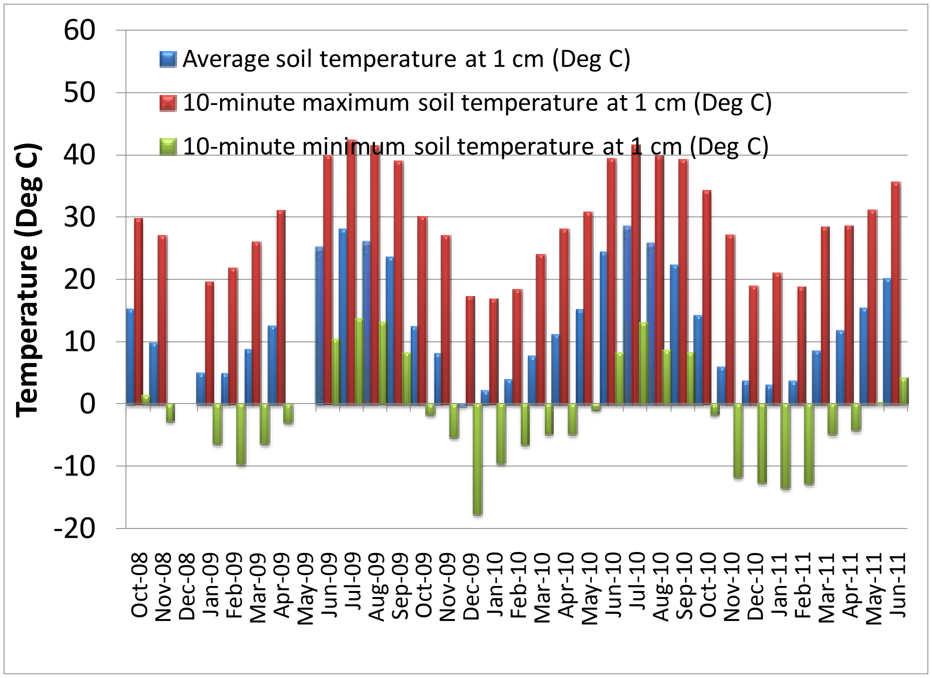


Figure 7. Temperature of soil in top 1 cm at Jacob site.

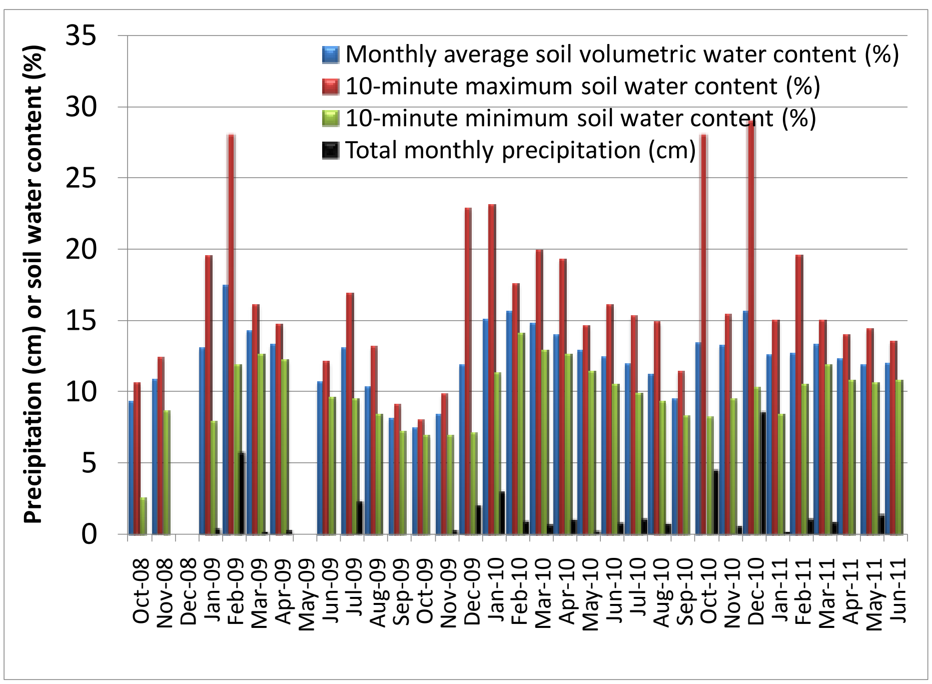


Figure 8. Monthly soil volumetric water content (0 – 10 cm) and total monthly precipitation (centimeters, cm) at Jacob site.

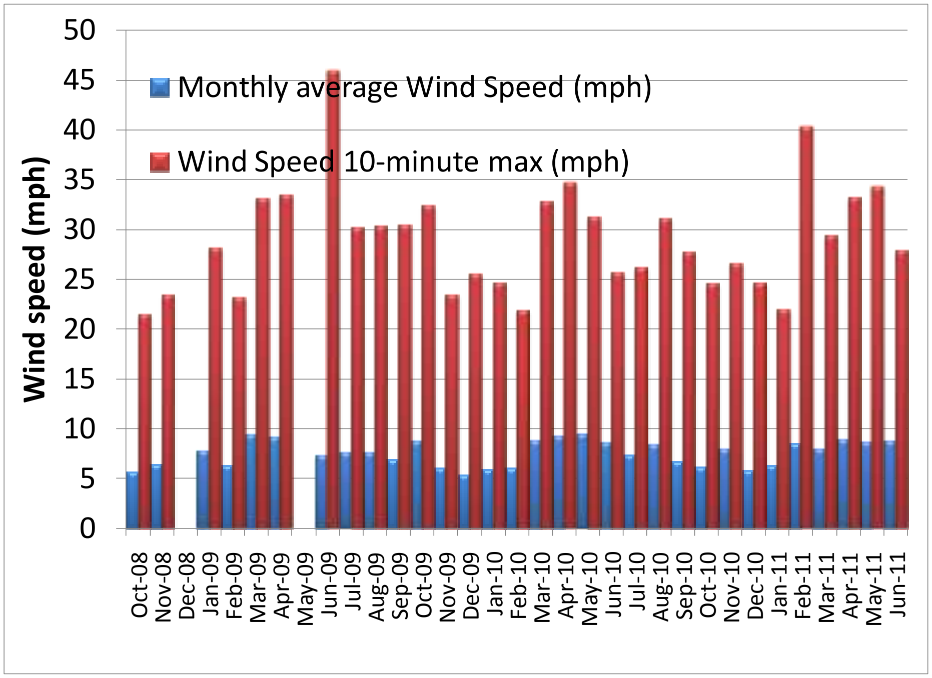


Figure 9. Monthly average and maximum wind speeds at the Jacob site.

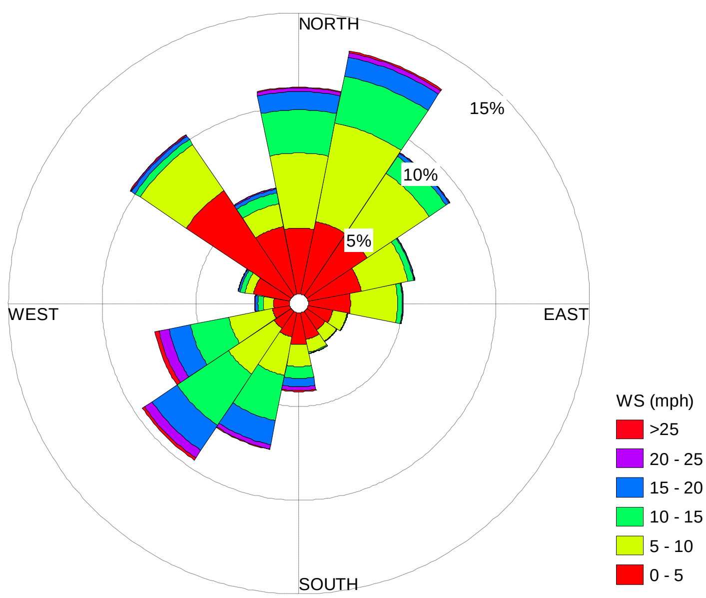


Figure 10. Wind rose from the Jacob fire site (October, 2008 to June, 2011).

magnitude, than emissions from unburned interspace soils. The tests conducted at 3 MAB (Figure 12) differed slightly from these general trends in that emissions from burned ridge soils were comparable to those from the control and emissions from ridge understory soils were comparable to those from interspace soils. This was attributed to the relatively high soil moisture content, as observed by field personnel, during the 3 MAB sampling effort as compared to other sampling times. Drainage sites did not appear to be as affected by the increased soil moisture as ridge sites likely due to the coarser soils there.

Moist soil at 3 MAB notwithstanding, the relationship between PM_{10} windblown emissions and the type of surface sampled is quite clear. Areas that were subjected to fire have higher emissions than the unburned surfaces. These observations provide a clear indication that the fire initially caused the soil surface over the entire burned area to be more susceptible to wind erosion. However, emissions from burned plant understories were especially elevated.

Figure 19 shows the PM_{10} emissions as measured by the PI-SWERL through the end of the 3,000 RPM step. Figure 20 shows the same information, but with emissions measured during a specific period normalized to the emissions from the control areas during that same period. Although the relationship between PI-SWERL RPM and wind speed is dependent on a number of parameters, for reference, it was estimated that at the Jacob Fire site, 3,000 RPM was approximately equivalent to 39 kph (24 mph) winds. Results are grouped by location (drainage versus ridge) and by understory or interspace soils.

By definition, in Figure 20 all of the unburned control measurements are equal to unity because the PM_{10} emissions for different areas during different sampling times were normalized (divided) by the unburned control measurements for that sampling time. The PM_{10} emissions in drainage understory soils were initially (at 1 MAB) highest among all the different soils sites tested. However, emissions from those surfaces decreased significantly by 21 MAB and were nearly the same as emissions from unburned control soils by 24 MAB. There was significant variability among measurements completed in the interspaces in drainage locations and it was difficult to identify with much confidence a trend over time for such locations. Measurements on burned ridges, both on the interspace and understory soils, suggest that PM_{10} emissions rapidly attenuated to unburned control values starting at 3 MAB. Although the 24 MAB measurements were slightly higher than at 21 MAB for these two soils, this observation was likely a result of measurement variability rather than a real physical occurrence. This was supported by a return to lower levels at 36 MAB. Overall, at the end of the three-year study, all of the surfaces at the Jacob Fire site appeared to have returned to nearly unburned PM_{10} emission levels.

<u>Filter Samples and Analysis</u>

Filter-based chemical measurements of PM_{10} emitted during PI-SWERL tests were used to examine how quickly soil surfaces returned to unburned chemical compositions. Thus, the ratio of elemental carbon (EC) to crustal (soil) (EC:Soil) PM_{10} was examined. Elemental or black carbon is primarily a product of combustion processes such as burning

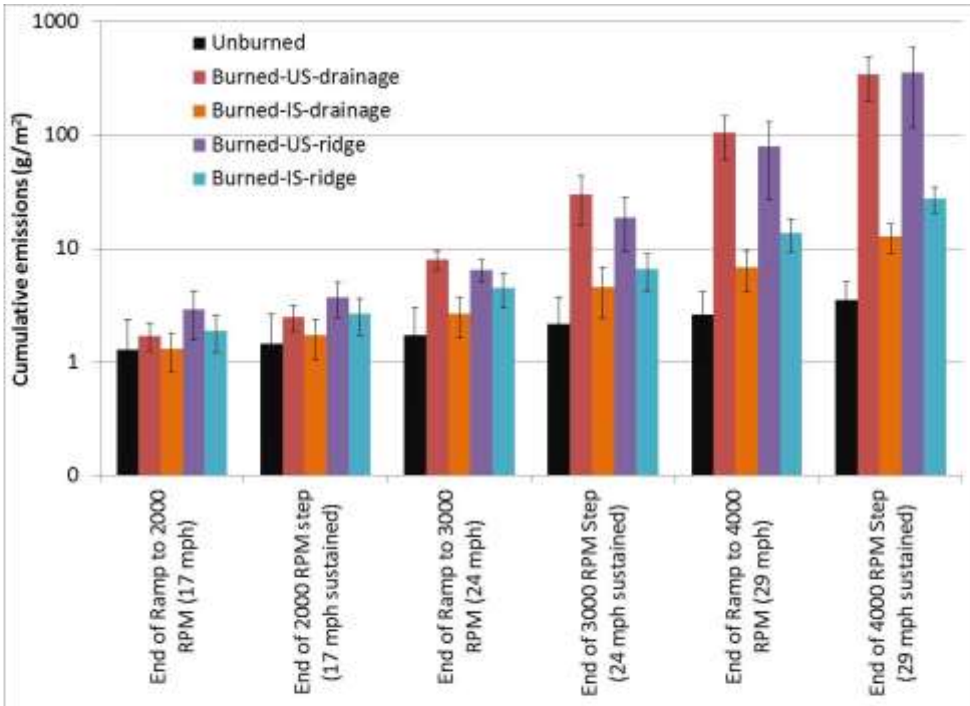


Figure 11. PI-SWERL results at 1 MAB.

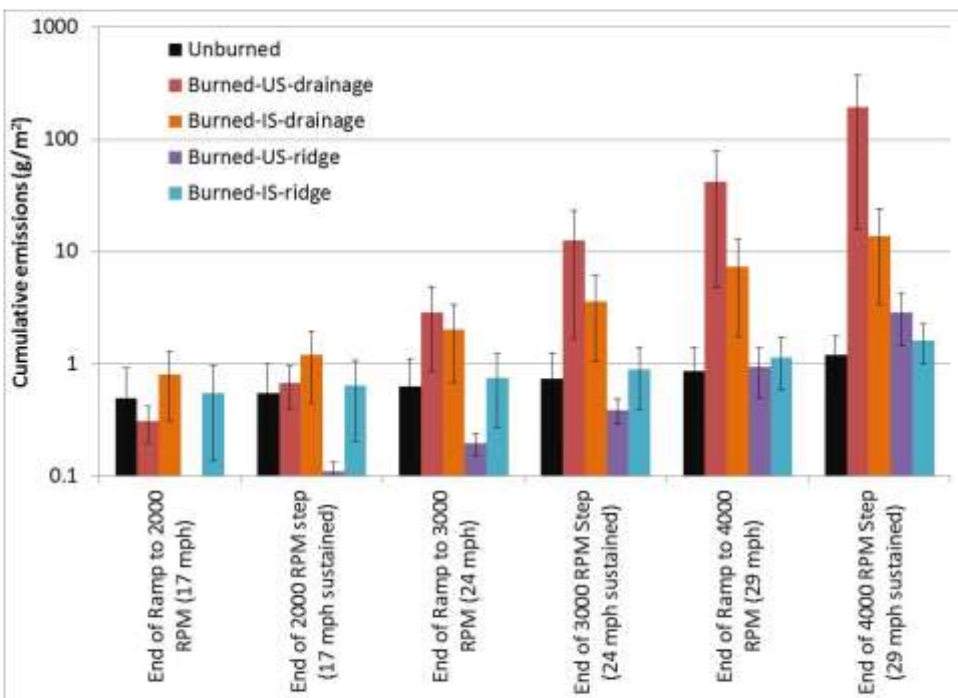


Figure 12. PI-SWERL results at 3 MAB.

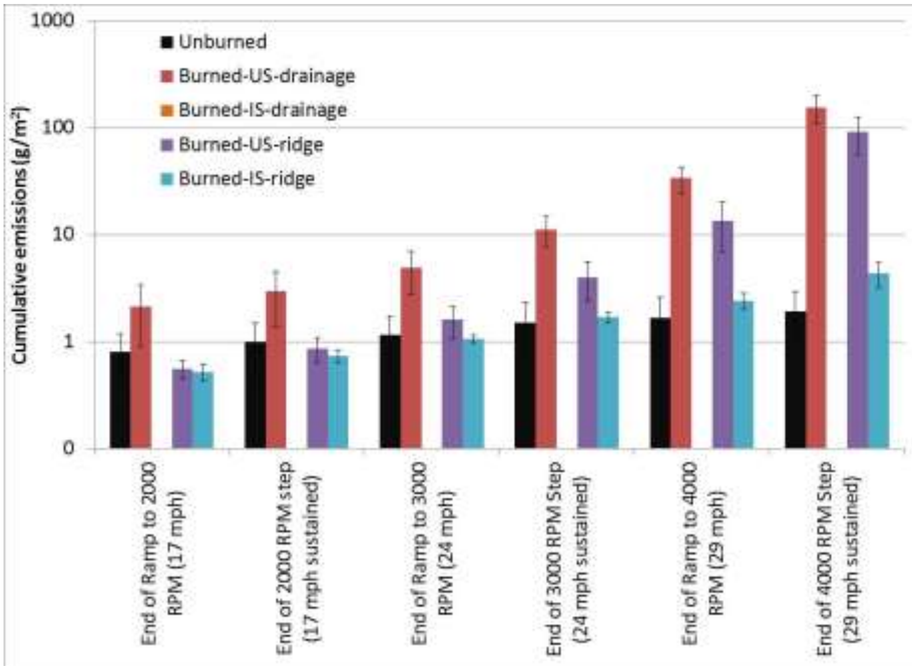


Figure 13. PI-SWERL results at 6 MAB. For measurements in the wash, only burned plant mound areas were sampled at 6 MAB.

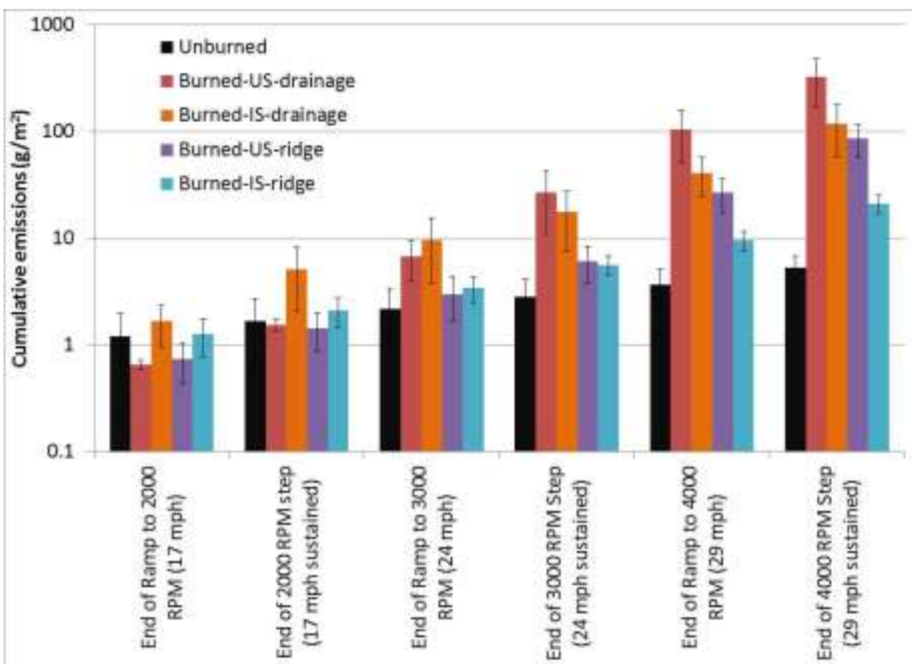


Figure 14. PI-SWERL results at 13 MAB.

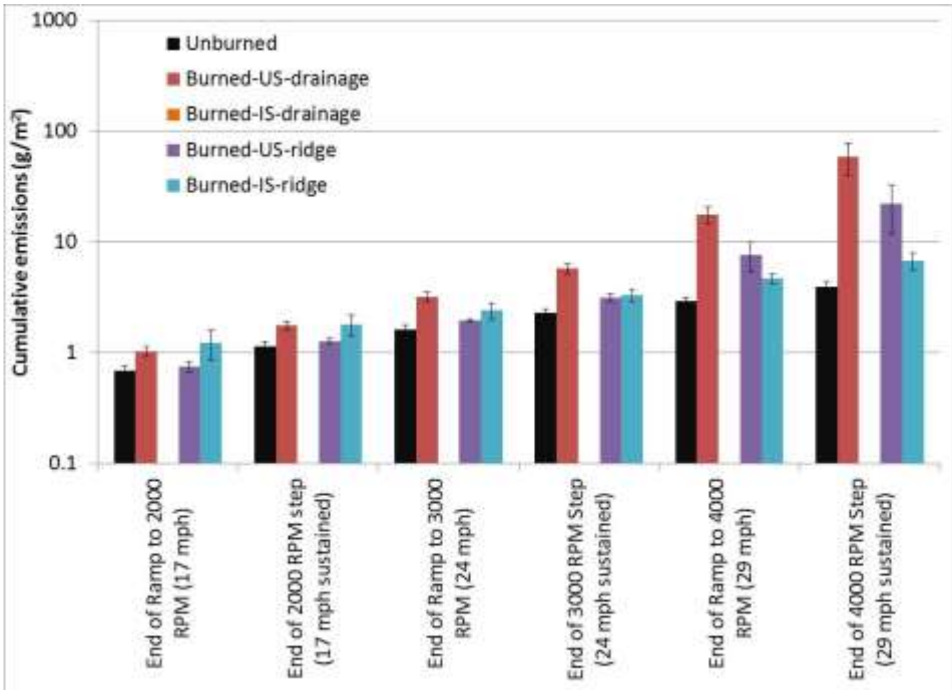


Figure 15. PI-SWERL results at 21 MAB. Burned interspace drainage soils were not sampled because grasses were too thick to allow placement of PI-SWERL.

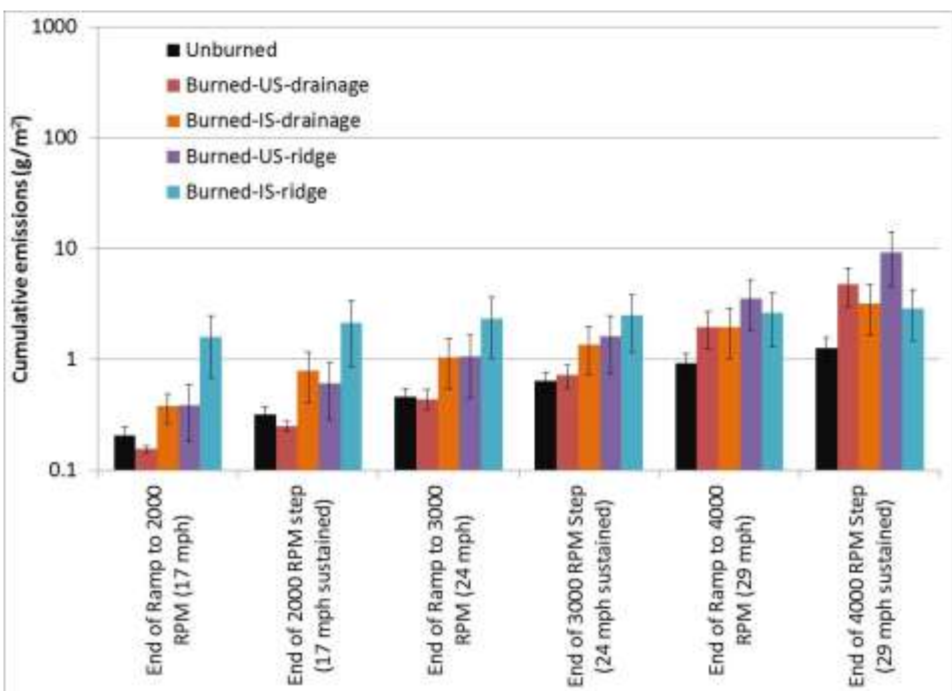


Figure 16. PI-SWERL results at 24 MAB.

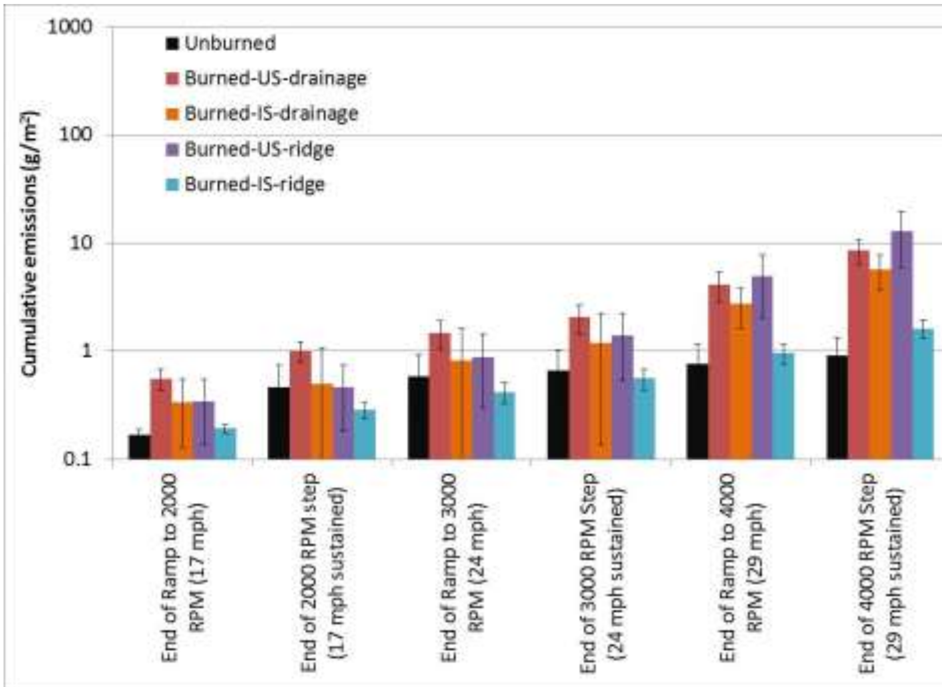


Figure 17. PI-SWERL results at 34 MAB.

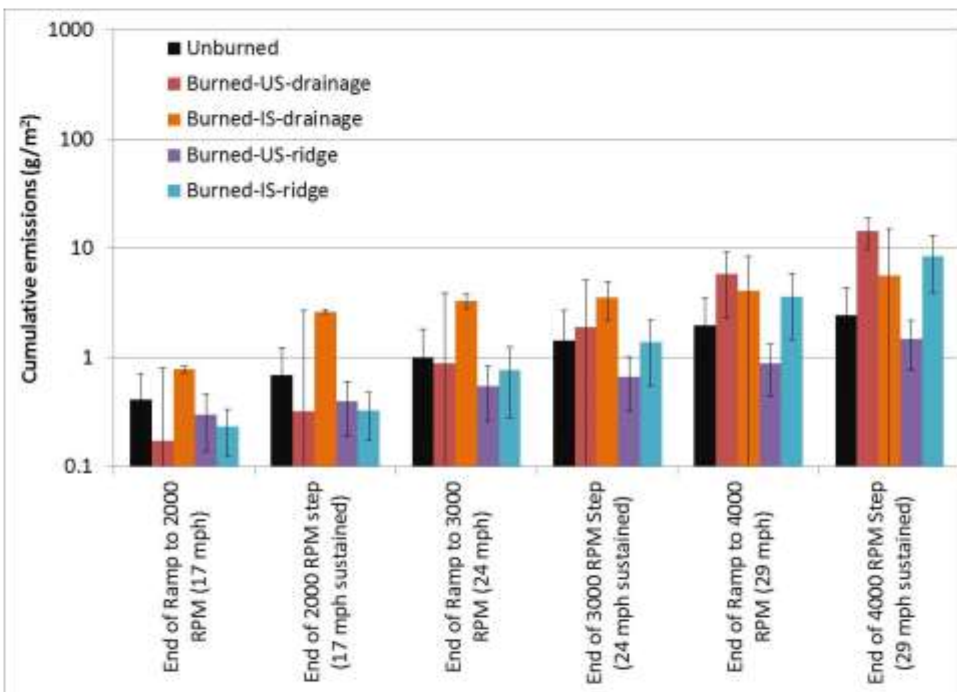


Figure 18. PI-SWERL results at 36 MAB.

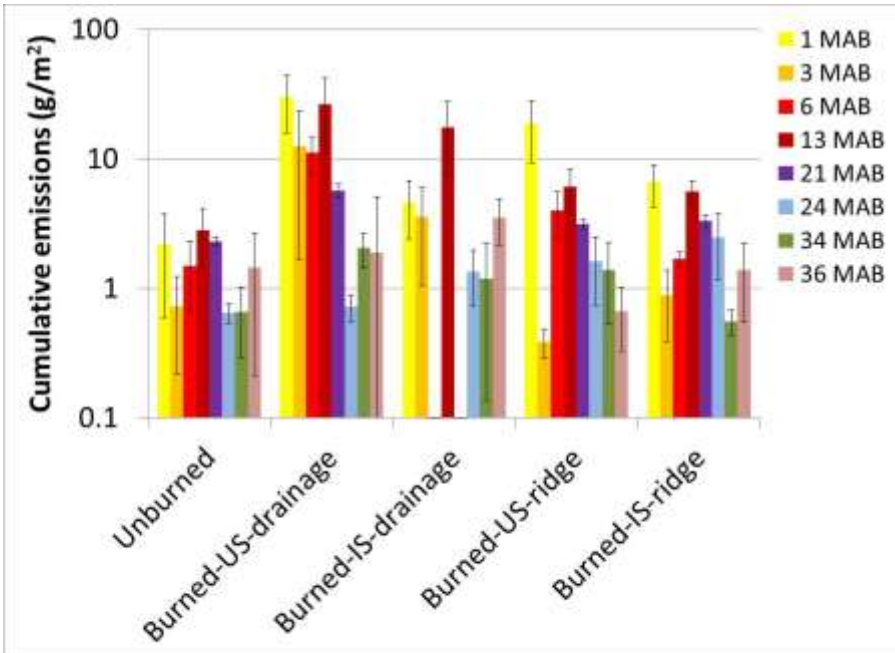


Figure 19. PM_{10} emissions at the end of the 3,000 RPM Step (39 kph sustained) at 1, 3, 6, 13, 21, 24, 34 and 36 MAB.

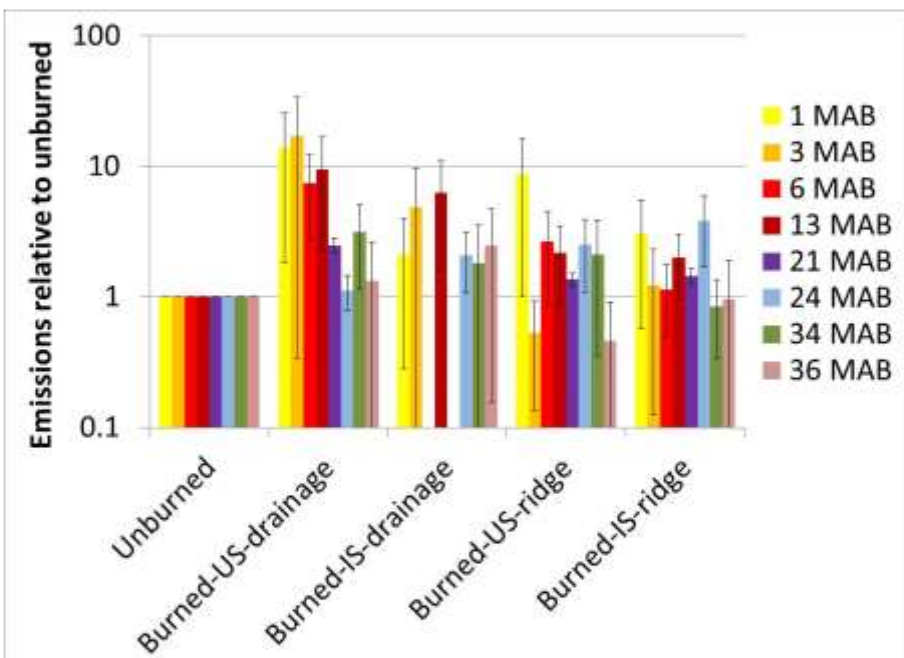


Figure 20. Normalized PM_{10} emissions at the end of the 3,000 RPM Step (39 kph sustained) at 1, 3, 6, 13, 21, 24, 34 and 36 MAB. Emissions at each period in time were normalized to emissions measured at the control locations during the same period.

of fossil fuels or biofuels. It is estimated using thermal optical reflectance performed on quartz- fiber filters. The amount of soil material is inferred through the assumption that silicon (Si), aluminum (Al), iron (Fe), calcium (Ca), and titanium (Ti) exist in their metal oxide form. Following the IMPROVE equation (Interagency Monitoring of PROtected Visual Environments) (Malm *et al*., 1994), the mass of PM_{10} associated with soil is estimated from:

$$[\text{Soil}] = 2.20 * [\text{Al}] + 2.49 * [\text{Si}] + 1.63 * [\text{Ca}] + 2.42 * [\text{Fe}] + 1.94 * [\text{Ti}] \quad (2)$$

where the concentrations of metals are determined from x-ray fluorescence spectroscopy of Teflon® filters.

The ratios of EC:Soil are shown in Figure 21. There was substantial scatter in the data. However, it appeared that immediately after the fire (1 MAB), the EC:Soil ratio was elevated for burned ridges and burned drainage soils as compared to the unburned soils indicating elevated levels of burned organic material (*e.g.*, vegetation, litter, soil surface organic matter). Starting at 13 MAB, EC-Soil ratios from the burned areas began to approach the EC:Soil ratios on the unburned areas and by 24 MAB, the EC:Soil ratio was not distinguishable between burned and control soils. This provides further evidence that most of the burned surfaces at the Jacob fire site had started to return to unburned conditions within two years after the fire.

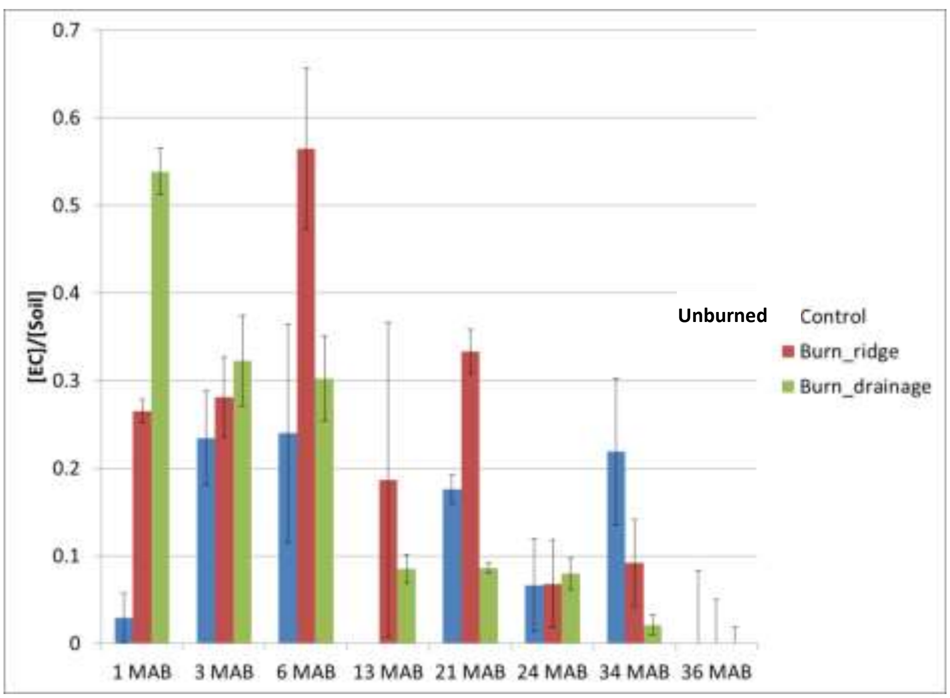


Figure 21. Ratio of elemental carbon (EC) to soil constituents (Soil) from PM_{10} filter samples collected during different field measurements. EC was not available for control at 13 MAB due to breach in filter integrity.

Conclusions

Based on three years of wind erodibility measurements and simultaneous chemical characterization of PM_{10} emitted from both burned and unburned areas, the Jacob Fire site had started to return to unburned levels of wind erodibility about 24 months after the fire. Similarly, it appeared that the signature of the fire, as parameterized in the EC:Soil ratio started to fade one year after the fire and was essentially absent two years after the fire.

POTENTIAL FOR WATER EROSION FOLLOWING A WILDFIRE

Measurements

Methodology and Equipment

Watershed runoff and sediment load can increase substantially after a fire. In a review of fire-related literature, Berli *et al*. (2008) found that within the Southwest U.S, a large increase in watershed runoff could be on the order of 14 times unburned levels, and sediment yield could increase to 416 times unburned levels. Generally, it is recognized that these elevated levels are associated with the first intense post-fire rainfall, and tend to return to unburned levels within two to three years after that. But, the specific characteristics of post-fire runoff and sediment are not well-documented for sites specific to the NNSS. Therefore, data were gathered from field measurements at the Jacob Fire site, analogous to the Transition ecoregion of the central NNSS. The specific objective was to identify characteristics of runoff and sediment for three years following a fire.

Standard methods to predict runoff and sedimentation are not applicable to many arid and semi-arid watersheds due to the lack of data. Watersheds that do have runoff data usually have short periods of record and many periods of no flow. Associated sediment data are even more scarce. Efforts to obtain the data are logistically difficult if not impossible—fires, themselves, are unpredictable, as is the location, timing, and intensity of rainfall that leads to runoff. However, small-sized rainfall simulators can be used to provide realistic, site-specific rainfall, runoff, and sediment data from which to begin to understand watershed response. Runoff simulators are designed to determine the runoff and infiltration properties of field soils under specified rainfall inputs and non-ponded conditions.

A parameter important to some watershed modeling methods is the initial abstraction (I_a) and infiltration loss of precipitation. Initial abstraction is the amount of precipitation that initially infiltrates into the soil prior to the occurrence of any runoff. Most rainfall-runoff models do not directly account for I_a and infiltration losses, and rely on precipitation loss components that are considered to be sub-basin or overland flow area averages. The Soil Conservation Service (SCS) (now Natural Resource Conservation Service) Curve Number (CN) approach (USDA-SCS, 1986) is commonly used to account for precipitation losses and is recommended by the Clark County (Nevada) Regional Flood Control District (1999). This method for estimating rainfall excess was developed from studies of natural rainfall and runoff from small (less than 10 acres) agricultural watersheds in the mid-west and southeast United States (USDA-SCS, 1986), where the method has been shown to produce reasonable results. A drawback to the SCS CN approach is the assumption that one I_a value is appropriate for precipitation events for all return periods; however, this may not be correct.

The SCS method relates the drainage characteristics of soil groups to a CN (USDA-SCS, 1986). This relation is based on soil group classification, vegetation cover, land use type (urban, agricultural, or desert), and antecedent moisture conditions, based on the amount of rainfall in the prior five to 30 days. Curve numbers range from 100, which represents a completely impervious surface with decreasing CNs for more permeable surfaces. For example, a typical CN for asphalt pavement is 98, whereas a CN for a golf course may be 61 or less. In a typical application, local regulatory agencies or SCS guidance is required for CN selection. However, if runoff volume, precipitation volume, and I_a are measured or determined from measured data, a CN can be directly calculated.

The total depth of runoff from a storm is related to a CN by the following equations:

$$Q = \left\{\frac{(P-I_a)}{[(P-I_a)+S]}\right\} \quad (3)$$

where Q is runoff volume; P is precipitation volume; I_a is initial abstraction; and S is the potential maximum retention of precipitation in the soil after runoff has begun (as runoff occurs, some infiltrated precipitation is retained in the soil). S is related to I_a as:

$$S = \frac{I_a}{0.2} \quad (4)$$

The CN is then calculated as follows:

$$CN = \frac{1000}{(S+10)} \quad (5)$$

The portable rainfall simulator (Figure 22) used for this study consisted of a flat Plexiglas reservoir (61 cm x 61 cm, 24 in x 24 in) for water, with hypodermic needles on the underside (Mutchler and Moldenhauer, 1963; Munn and Huntington, 1976). Water drops were produced on the needles by providing a constant gravity head, wetting a 3,612 cm^2 (1422 in^2) area directly beneath the rainfall simulation. Rainfall simulator measurements were conducted at rates of approximately 3.56 cm/hr (1.40 in/hr), simulating the maximum intensity of a 1-hour, 100-year storm, as per the NOAA Atlas 14 (National Weather Service/NOAA, 2004) for the area. Rainfall simulations lasted one hour.

To sample total runoff and sediment, a small trough was entrenched just downslope of the footprint of the rainfall simulator. A piece of PVC pipe, sliced lengthwise and capped at both ends, was placed in the trough with a 90-degree-bend aluminum flashing installed over the trough lip to collect the flow. If flow occurred during the simulation, water collected in the trough was sampled for 30 seconds every five minutes, and the volume recorded. Runoff sampling was stopped at the end of the one-hour simulation. The runoff samples were sent to a laboratory to determine suspended sediment concentration and particle size distribution.

Figure 22. Photograph of portable rainfall simulator.

The rainfall simulator was calibrated before and after each rainfall simulation experiment to ensure that the application rate was approximately at the target rate and to evaluate any drift from the initial rate. This was accomplished by placing a Plexiglas plate under the simulator and collecting the output at three one-minute increments before each test. The depth of the collected output was measured in a graduated cylinder to determine the rainfall rate. Without flow interruption, the plate was quickly removed and the experiment was started. At the end of the experiment, again without flow interruption, the Plexiglas plate was inserted back under the simulator, and as before, the output was captured for at least one one-minute increment. These initial and post-calibration rates were averaged and used as the application intensity in calculations to determine infiltration properties.

After the initial calibration measurements, the experimental precipitation event was started on the test surface, during which four time readings were taken: (1) when initial ponding occurred anywhere on the surface test plot, (2) when initial runoff occurred anywhere on the surface test plot, (3) when runoff occurred in each quadrant of the surface test plot, and (4) when initial runoff reached the collection trough. A “pre-rainfall” (dry) soil sample was collected adjacent to each test plot before the test, and a soil sample was collected from the center of each test plot after one-hour of rainfall simulation. The gravimetric (by weight) soil moisture content, bulk density, and porosity was determined from these samples (Appendices D and E). Soil texture was determined from a particle size analysis of the post-simulator bulk density samples.

A total of six rainfall simulation tests were performed during each field visit to the Jacob Fire site. Rainfall simulator data was gathered at two burned ridge sites, at two burned drainage sites, and at two unburned sites. At each site, rainfall simulation tests were performed on an understory soil and interspace soil. There were no replicates. Each rainfall simulator location was marked after the test to ensure subsequent test locations were within the same sample site, but on surfaces undisturbed by previous tests. Field measurements occurred at 1, 3, 7, 15, 22, 25, and 34 MAB.

Results

Soil Texture

The soil surface textures at the Jacob Fire site at 34 MAB are presented in Figure 23. Two groups of soil textures are apparent: coarse soils, with high sand and low clay contents; and finer soils, with lower sand and higher clay contents. The coarser soils are associated with drainage areas (both understory and interspace) while the finer soils appear on the unburned sites (both understory and interspace). Burned ridge understory soils were coarser than burned ridge interspace soils. At all locations, the clay content of interspace soils were higher than clay contents at understory soils—understory soils were more coarse than at interspace soils. These vegetation-associated surface texture results are consistent with similar results at the NNSS (Caldwell *et al*., 2008).

Runoff Analysis

The total runoff volumes recorded for each site throughout the 34-month period appear in Figure 24. The spatial and temporal runoff responses from the rainfall simulations were highly variable. The runoff from burned ridge interspace and unburned interspace sites appear similar. Little runoff occurred on the burned drainage soils where soil appeared coarser, but small amounts of runoff were associated with understory soils in the drainage areas where finer sediment may have accumulated. No temporal or seasonal runoff trends appear in the runoff data. Entire runoff hydrographs are included in Appendix F.

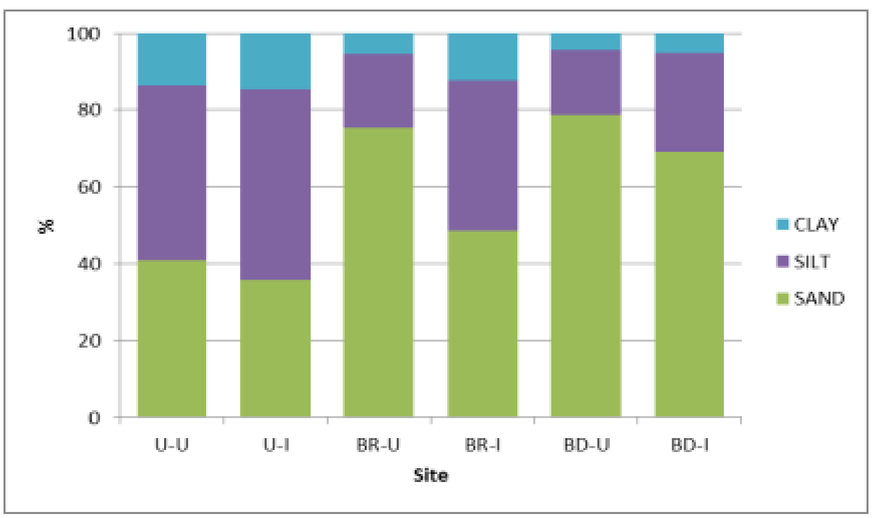


Figure 23. Surface soil textures for runoff plots at the Jacob Fire site. (B = burned, U = unburned, R = ridge, D = drainage, U = plant understory, I = interspace).

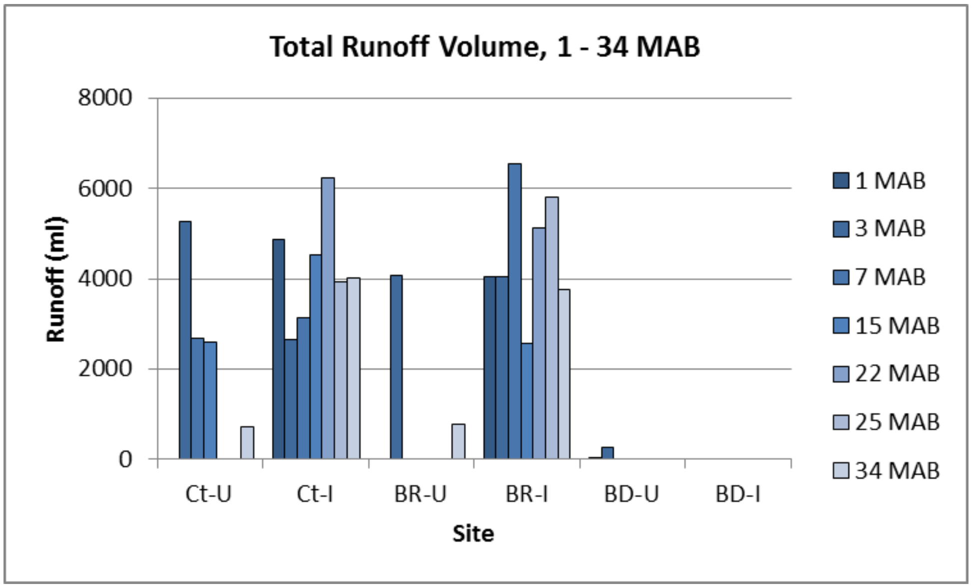


Figure 24. Total runoff (milliliters, ml) collected at Jacob fire site. (B = burned, U = unburned, R = ridge, D = drainage, U = plant understory, I = interspace sites.)

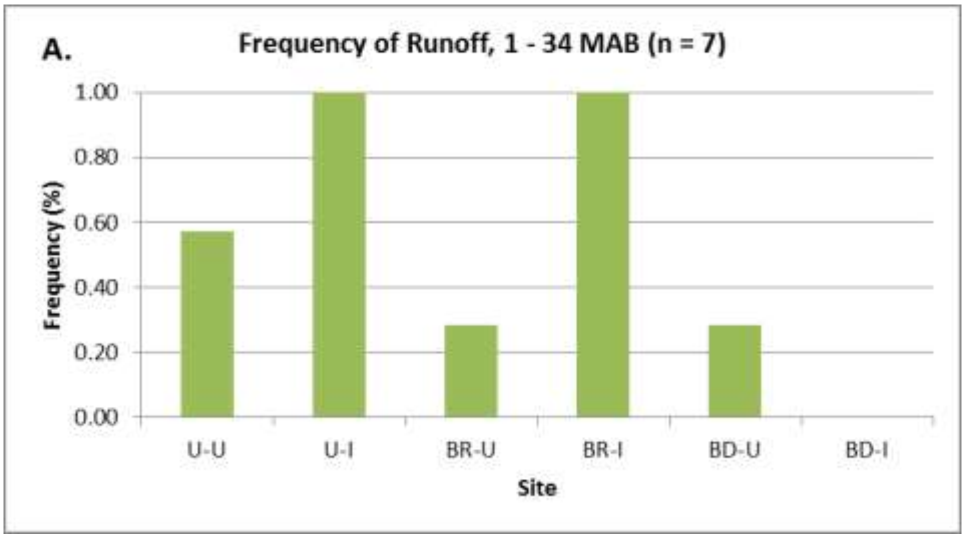
Averaged runoff data are shown in Figure 25. The frequency of runoff (Figure 25A) was highest at unburned and burned ridge interspace sites (U-I, BR-I) where some runoff was recorded for each rainfall simulation test. These sites, along with the unburned understory site (U-U), were those having relatively fine-textured soils (high % clay and low % sand). Throughout the 34-month monitoring period, no measurable runoff was ever recorded at the burned drainage interspace sites. In general, runoff from interspace sites was greater than for understory sites, except at burned drainage sites where finer soil particles may have been deposited beneath vegetation.

Averaged runoff and standard error bars are shown in Figure 25B. Three groups of runoff response are apparent: U-I and BR-I averaged the highest runoff response, followed by U-U and BR-U sites. Little or no runoff was measured on the drainage sites.

The similarity of response between unburned and burned ridge sites, whether on interspace or understory soils, indicated a lack of runoff response to the burn at the locations tested. More and frequent runoff was recorded on ridge sites. Little, if any, runoff was recorded at the drainage sites. These trends appear related to the trend in soil texture where higher and more frequent runoff was associated with those sites having textures with higher clay and lower sand contents.

Sediment Analysis

Suspended sediment data are sparse as not all runoff simulation tests produced measurable sediment. Average sediment concentrations (milligrams/liter, mg/l) for rainfall simulator sites are shown in Figure 26. As with runoff data, the sediment concentrations are high and similar for the ridge interspace sites, whether burned or unburned. Sediment concentrations from interspace sites were higher than from understory sites. However, a single, small test at 1 MAB with little runoff (25 milliliter, ml; 1.5 in^3) at a burned drainage understory site produced a large amount of sediment (10555 milligrams/liter, mg/l; 0.7 pounds/ft^3), almost 17 percent higher than the otherwise largest sediment concentration recorded throughout the 34-month project. While unusual, this amount of variability is not unexpected after a destructive event such as a fire.



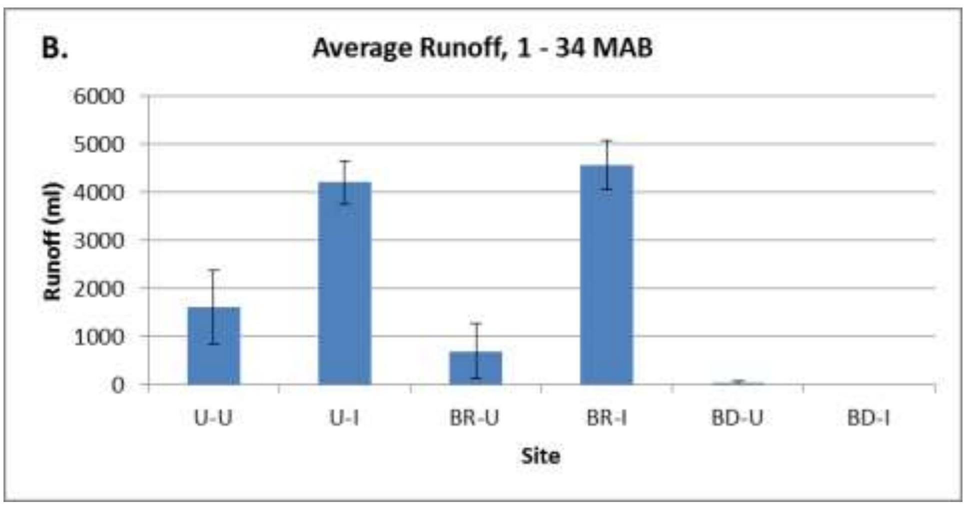


Figure 25. (A) Averaged runoff frequency (%) and (B) average runoff volumes (milliliters, ml) calculated for the Jacob fire site. (B = burned, U = unburned, R = ridge, D = drainage, U = plant understory, I = interspace sites.) Error bars are standard error (standard deviation / no. of samples^{½}).

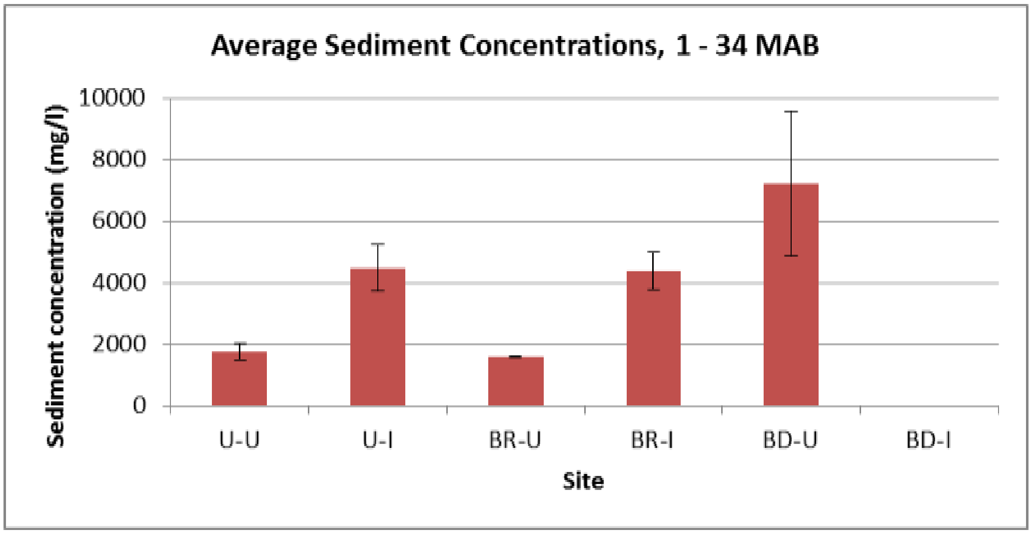


Figure 26. Average sediment concentrations for measurable runoff collected at the Jacob fire site. (B = burned, U = unburned, R = ridge, D = drainage, U = plant understory, I = interspace sites.) Error bars are standard error (standard deviation / no. of samples^{½}).

The sizes for which suspended sediment were analyzed ranged from 0.01 µm (3.9 x 10^{-7} in) to 1,000 µm (0.039 in). The resultant distributions appear in Figure 27. A vertical line was added to the graphs at 10 µm to view runoff sediment sizes in same context as wind emission particle sizes. Generally, the shapes of the distributions appeared similar, whether from burned or unburned sites. The size distributions of suspended sediment from unburned sites were consistent over time and there was little difference between understory and interspace sites. Suspended sediment from some burned plots may have exhibited a fire effect. Sediment from the burned ridge interspace sites was smaller (60 percent of the sediment finer than 10 µm) at 1 MAB than at later sampling dates (40 percent of the sediment finer than 10 µm); the same result was evident for burned drainage understory sites. These differences disappeared by 3 MAB. It appears there was a flush of fine particles washed off these sites immediately after the fire. Also, the 10 µm particle size used in wind erosion analysis appears to indicate the central tendency (median) of the distributions fairly well.

<u>Curve Numbers (CN)</u>

Curve numbers integrate several aspects of runoff related landscape characteristics including soil type, vegetation cover, land use, and initial soil moisture, and are used to estimate the amount of direct runoff expected from a given rainfall event. As such, they can be used to express the potential for runoff with higher values (CN ≤ 100) indicating a higher runoff potential.

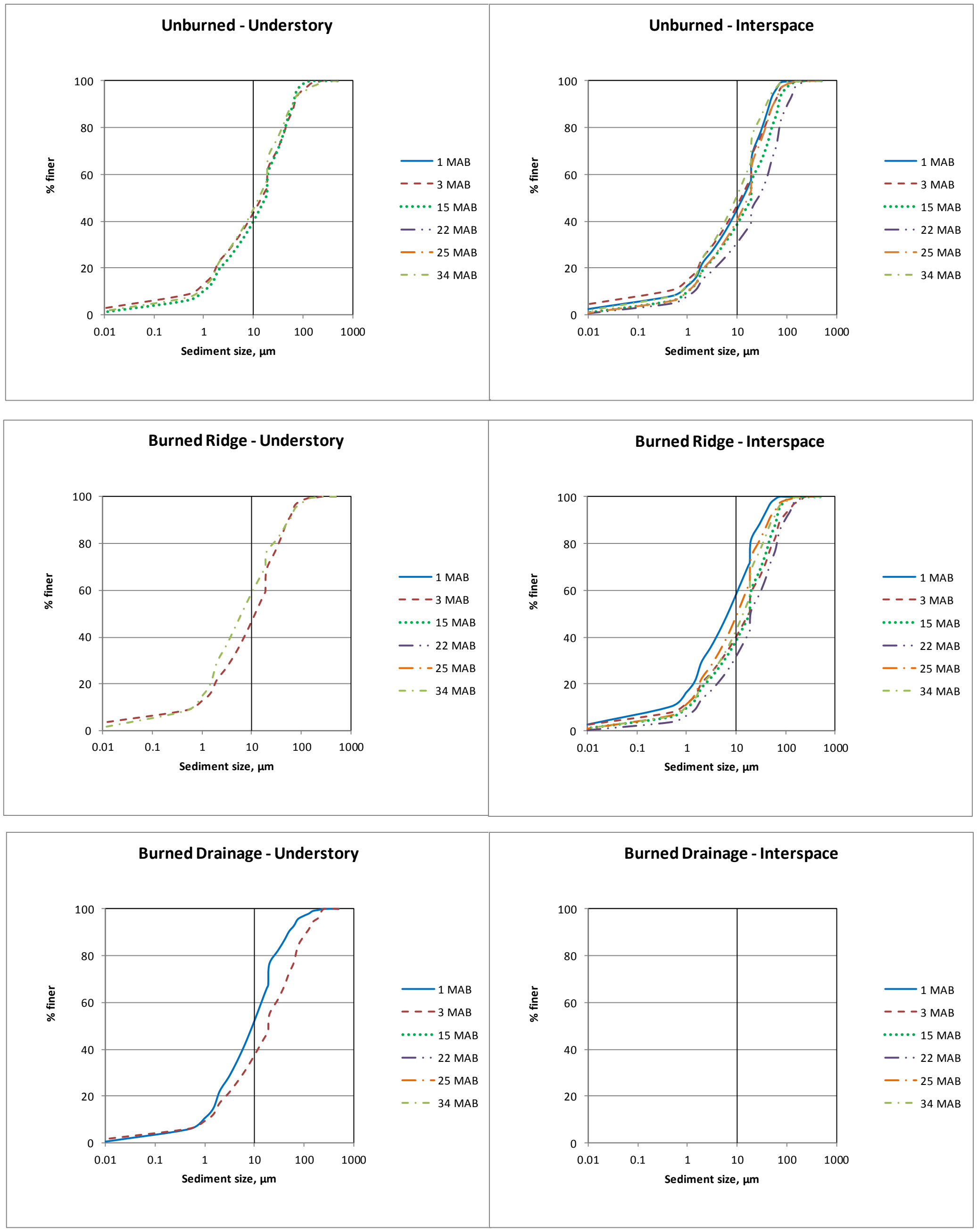


Figure 27. Particle size distribution of suspended sediment for 1 MAB – 34 MAB. (Reference line at 10 µm—used for wind erosion analysis—added for clarity).

During each field visit, only one rainfall simulation was performed on each of the six sample sites (no replicates). During each simulation, the timing and type of runoff on the soil surface beneath the simulator was noted, including: time (elapsed since rainfall began) of isolated ponding, time runoff appeared, time runoff occurred on all four quadrants beneath the simulator, and time when any runoff appeared in the downstream collection trough. CNs were calculated based on these flow characteristics; the resulting calculated

CNs from the rainfall simulator experiments for all four initial runoff situations appear in Appendix H. The CNs may not exactly correspond to runoff data shown in Figures 24 and 25 as those figures were based on runoff that reached the collection trough. However, from previous rainfall simulation studies performed on the NNSS (Miller and French, 2001), it was determined that the most appropriate CN for soil surfaces tested came from the "All Quadrant Runoff" measurement.

Curve numbers based on runoff from all quadrants beneath the rainfall simulator were calculated for the Jacob Fire site and are shown in Figure 28. Curve numbers for burned ridge interspace and unburned interspace sites were relatively high. While the occurrence of runoff appeared to decrease over time for the burned ridge understory and burned drainage understory sites, so too did runoff at unburned understory sites, indicating burned and unburned sites exhibited the same runoff potential with respect to CNs. The maximum CN was calculated for the burned drainage understory site (CN = 96), slightly higher than for the other sites (with maximum CNs = 95), but only three of the seven rainfall simulations at that site produced runoff, and the CNs calculated for that site were within the variability evident throughout the study. There were no consistent increases or decreases in CNs over time—no temporal trends were evident in the calculated CNs. Nor were there notable differences between CNs calculated for burned and unburned sites.

Overall, the integrative CNs reflected a runoff potential at the Jacob fire site consistent with other runoff-related data (e.g., soil texture, rainfall simulator runoff frequency and amount, and sediment), with runoff likely on interspace sites on ridge (upland) sites, less likely on understory sites, and rarely if ever occurring in the drainage areas. The potential for runoff at the site appeared unaffected by the fire.

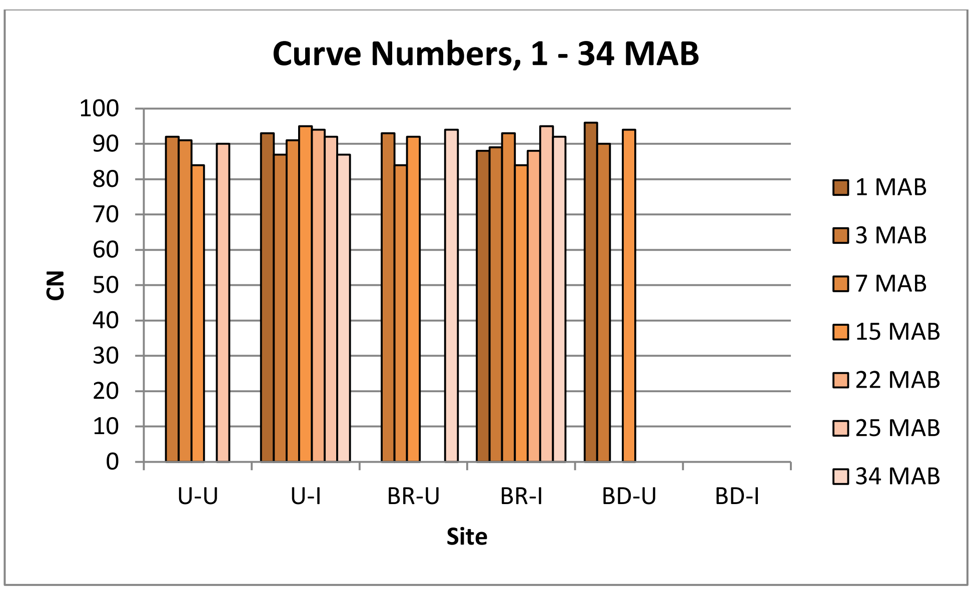


Figure 28. Curve numbers (CNs) calculated from all rainfall simulations (using data from All Quadrant Runoff simulations) at the Jacob fire site, 2008 – 2011.

Conclusions

Rainfall simulation measurements at the Jacob fire site suggested that fire may have an ambiguous effect on runoff and sediment transport, under the conditions tested at the site.

As expected, the runoff responses in ridge areas were different from those in drainage areas, with burned ridge areas responding like unburned areas. The runoff response of burned drainage area soils was almost opposite to those on ridges, apparently due to the more coarse-textured soils there. For example, runoff always occurred on burned ridge interspace and unburned interspace sites; no runoff occurred on burned drainage interspace sites. Except in the drainage area, the interspace sites were runoff source areas whether burned or unburned. Runoff from interspace soils was greater than from understory soils for both burned ridge and unburned sites. Again, the opposite response was observed on burned drainage soils where runoff occasionally occurred on burned drainage understory soils. The data show that coarse soil there may have inhibited surface runoff. Overall, no clear temporal trend was evident for runoff after a fire at the Jacob Fire site, under the simulated rainfall conditions applied there. Either there was no fire effect, or there was not enough data to discern a trend. However, the amount and frequency of runoff did appear to be positively associated with the clay content in the surface soil. Over time, and in all instances (burned vs. unburned, drainage vs. ridge, burned vs. unburned), runoff was greater on finer textured soils with higher clay contents than on coarser textured soils. Thus, surface soil texture provided more of an indication about the potential for overland flow generation than did fire-related effects.

Like runoff, there was little difference between the concentration of sediment from burned ridge interspace and unburned interspace sites. Relatively high suspended sediment loads were consistently measured on interspace areas, indicating the interspace soils are source areas for suspended sediment.

A fire effect was detected on the size distribution of sediment suspended in the runoff. Sediment on burned ridge interspace sites and on burned drainage understory sites was more fine at 1 MAB (approximately 60 percent sediment was finer than 10 µm) than at any other site, but the effect was not apparent at any time after that (40 percent of the sediment was finer than 10 µm). There was an initial flush of fine particles off these sites immediately after the fire. The 10 µm particle size used to characterize wind emissions was indicative of the central tendency of the suspended sediment distributions.

Curve numbers were calculated for all rainfall simulations. Some burned surfaces initially (1 to 3 MAB) exhibited high curve numbers (indicating high runoff potential) followed by lower curve numbers (at 7 months MAB), with a return to higher curve numbers by 15 MAB. Generally, CNs reflected runoff results showing no clear spatial or temporal effect of fire on runoff, under the rainfall conditions applied at the fire site.

VEGETATION RESPONSE FOLLOWING A WILDFIRE

Vegetation Survey Measurements

Methodology

Vegetation surveys were conducted on three dates in 2009 (1-May, 6-July, and 20-October), on two dates in 2010 (14-May and 28-May), and once in 2011 (4-May), roughly 9 MAB – 33 MAB. Plot size was 50 cm x 50 cm (19.6 in x 19.6 in; $0.25m^2$, $2.7 ft^2$). The geomorphic setting of the plots was recorded as either 'drainage' if located in a drainage or 'upland' (i.e, ridge) if located outside of a drainage channel. For each geomorphologic setting, the plot was located either within the shaded extent of a shrub canopy (understory) or in the inter-shrub space (interspace). Burned understory and interspace sites were identified by the charred soil surface or visible shrub remains. Species and the number of individual plants by species were recorded within each plot. Field data were recorded on paper data sheets and digitized for analysis. Plot locations were recorded using a handheld global positioning system (GPS) unit in UTM zone 11, NAD83 datum.

Several standard metrics were used to condense the vegetation data, quantify vegetation diversity, and describe the vegetation response to fire. For example, greater plant density generally provides more fuel for fire, although high densities of grasses and forbs may act to stabilize the soil and reduce erosion. The density and richness of vegetation relates to the persistence of repeat fire susceptibility.

The vegetation analysis in this report focuses on blackbrush (*Coleogyne ramosissima*) as the dominant shrub species and cheatgrass (*Bromus tectorum*), the invasive non-native grass which has been shown to increase fire risk (Brooks and Matchett, 2003; Haubensak *et al.*, 2009). The full, detailed vegetation analysis is included in Appendix H.

Diversity Metrics and Indices

Density: calculated as the total number of plants per plot.

Richness: calculated as the number of different species recorded for each plot.

Dominance: calculated as the number of plants of species *i* divided by the total density across all plants in the plot. It was used to characterize specifically blackbrush and cheatgrass.

Shannon-Weiner Diversity Index (H') (Lande, 1996) is calculated as:

$$H' = -\sum_{i=1}^{S} p_i \ln p_i \qquad (6)$$

where p_i is the proportion of individual plants of the *i*-th species. The Shannon-Weiner index, which accounts for both abundance and evenness, is an entropy index and is sensitive to the rarity and commonness of species recorded. H' is maximized when plots have equal counts for species recorded and decreases with unequal abundance across species. Lower values of H' indicate low diversity (low richness and evenness).

Simpson's Diversity Index (D): (Lande, 1996) is calculated as:

$$D = \sum_{i=1}^{S} p_i^2 \qquad (7)$$

where p_i is the number of species S. Simpson's Diversity Index was calculated for each plot. This index is a measure of biodiversity that accounts for abundance, richness and evenness and returns values ranging from 0 (infinite diversity) to 1 (no diversity). D values near 0 correspond to highly diverse, heterogeneous plots while D values closer to 1 are more homogeneous. It is a probability function that determines the likelihood that any two plants randomly sampled would be from the same species. Both Simpson's Index and Shannon-Weiner index were calculated because of the relatively low richness of the area, particularly post-fire.

Evenness (J'): is calculated as:

$$J' = \frac{H'}{H'_{max}} \qquad (8)$$

where H' is Shannon-Weiner index at a particular sample location, and H'_{max} is the maximum H' for all sample locations. Evenness is a measure of relative abundance among the species observed and ranges from 0 to 1, where values closer to 1 indicate evenness or equal distribution of abundance of species while values approaching 0 indicate unequal (*e.g.* skewed) distribution of species abundance.

Statistical Analyses

Basic summary statistics were calculated for all sample groups within the burned and unburned categories. Spearman correlation was conducted to assess the degree to which a relationship existed between richness and density of burned and unburned plots, respectively, and plotted. F-tests for dispersion were run to test for equal variability between groups for each of the metrics, *i.e.*, richness and density. If, based on the F-test results, the groups exhibited unequal variances then Welch's approximation for unequal variances was used in independent t-tests. The t-tests were used to compare means between groups for each metric of richness and density. Analysis of variance (ANOVA) was conducted to compare burned with unburned plots at the gross comparative level for both richness and density.

Vegetation Survey Results - 2009

A total of 60 plots were sampled in the unburned area, and a total of 80 plots were sampled in the burned area in 2009. Twelve different species were identified across all plots. Plant lists and basic summary statistics of density and richness are presented in Appendix H for each sampling unit.

All of the species identified in burned plots were found in unburned plots. Three species were unique to the unburned plots: cushion cryptantha (*Cryptantha circumscissa*), western tansymustard (*Descurainia pinnata*), and purplemat (*Nama demissum*). Whitestem blazingstar (*Mentzelia albicaulis*), an annual native forb, was found three times as often in burned plots than in unburned plots, and was the most frequently identified species in burned plots. Sixweeks fescue (*Vulpia octoflora*) was the next most prevalent species, although this annual native grass was less often reported in burned than unburned plots. In the unburned landscape, sixweeks fescue and cheatgrass were the most frequently reported species, followed by whitestem blazingstar and blackbrush. In comparison, the most frequently reported species in the burned plots were whitestem blazingstar, sixweeks fescue, blackbrush and cheatgrass in that order. Cheatgrass was reported nearly four times as often in the unburned landscape than the burned.

Within one year of the fire, drainages exhibited similar plant densities and richness. Burned uplands were significantly less dense and less rich primarily as a function of the difference between understory sites.

Both burned and unburned sites had at least one plot where no vegetation were present. Burned and unburned mean density and richness in 2009 are compared for drainage and upland plots in Figure 29. Mean plot density for burned and unburned locations by plot type and geomorphic setting is presented in Figure 30 and Figure 31, respectively. At the landscape level, results showed no significant difference in the mean plant density between burned (44.3 ± 58/plot) and unburned (53.6 ± 48.5/plot) plots ($t(138) = 1.01$, $p = 0.3162$) nor was there a significant difference in variability of density between burned and unburned plots ($F(59, 79) = 0.7$, $p = 0.149$). Mean richness of unburned plots was 2.6 ± 1.1 and for burned plots was 1.8 ± 1.1. Unburned plots showed significantly higher richness than burned plots ($t(138) = 3.96$, $p < 0.0001$) but were similar in richness variability ($F(59, 79) = 1.15$, $p = 0.565$).

Results from Shannon-Weiner index showed unburned areas ($\overline{H}' = 0.477$) were significantly more diverse than burned ($\overline{H}' = 0.267$) areas ($t(128) = 3.86$, $p = 0.0002$). Simpson's Index (D) showed that unburned areas ($\overline{D} = 0.743$) were significantly more diverse than burned areas ($\overline{D} = 0.845$) ($t(138) = -2.93$, $p = 0.004$). It should be noted that both of these $\overline{D}$ values are indicative of low diversity in general for both burned and unburned plots as they approach a value of 1. Mean evenness for burned ($J' = 0.464$) and unburned ($J' = 0.529$) areas was not significantly different ($t(89) = 1.05$, $p = 0.296$). Both values of J' indicate some inequality of the distribution of plant abundance but not heavily skewed towards one species.

Diversity and evenness comparisons for burned and unburned, drainage and upland plots, respectively, are presented in Figure 32. Burned drainages exhibited almost no diversity. There were relatively few different species recorded in burned drainages and, of those recorded, only a small few were found in abundance. In unburned drainages, fewer different species were recorded however those species tended to occur in relatively even proportions. Upland areas were no different in diversity although burned uplands exhibited a more even distribution of species.

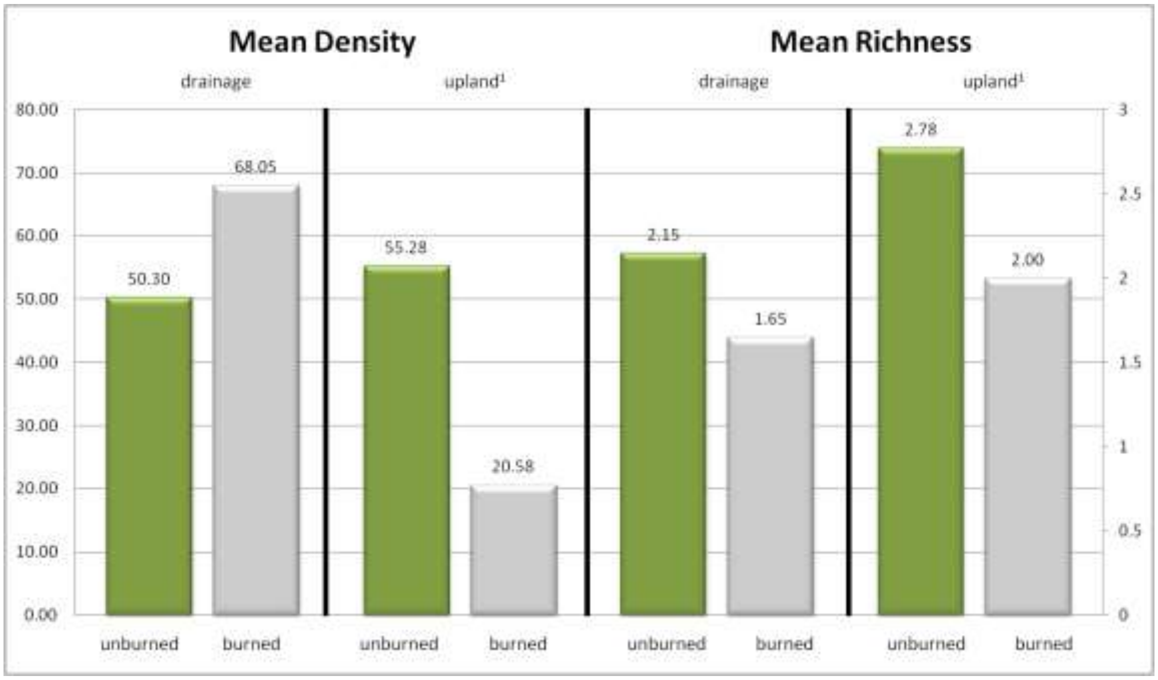


Figure 29. Mean density and mean richness (labeled) for all unburned and burned plots in each configuration of drainage or upland in 2009. [1] indicates a significant difference.

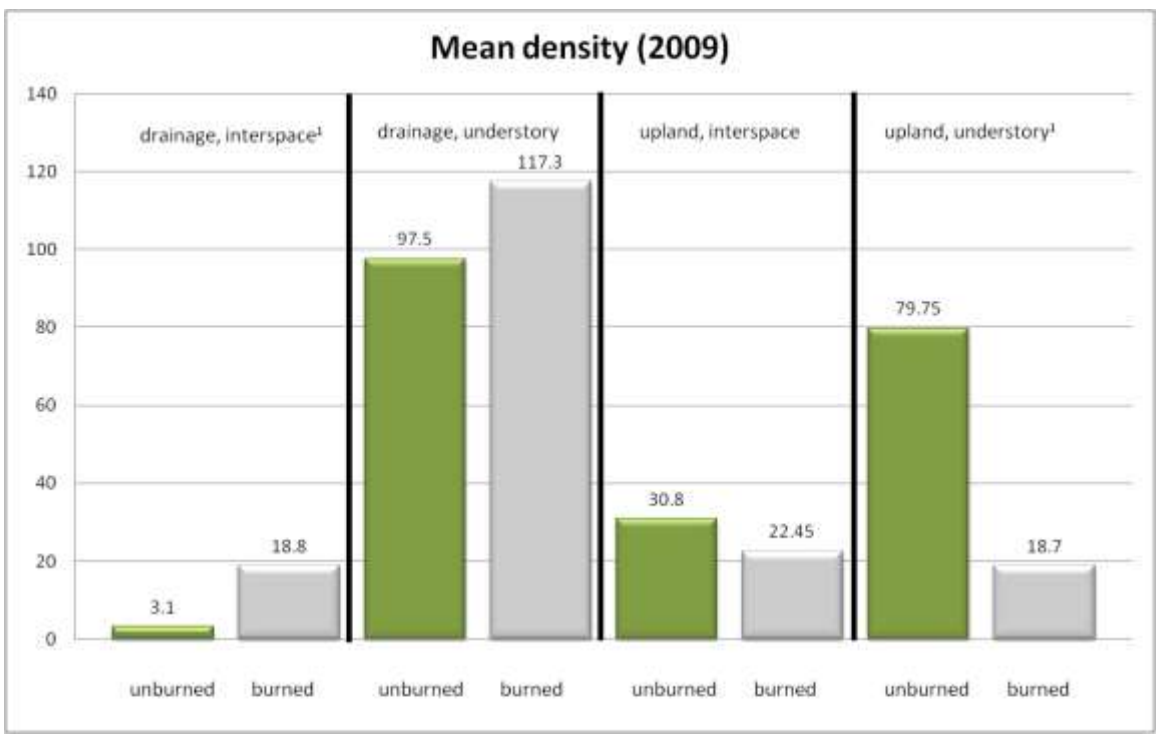


Figure 30. Mean density (labeled) by plot type for 2009. [1] indicates a significant difference.

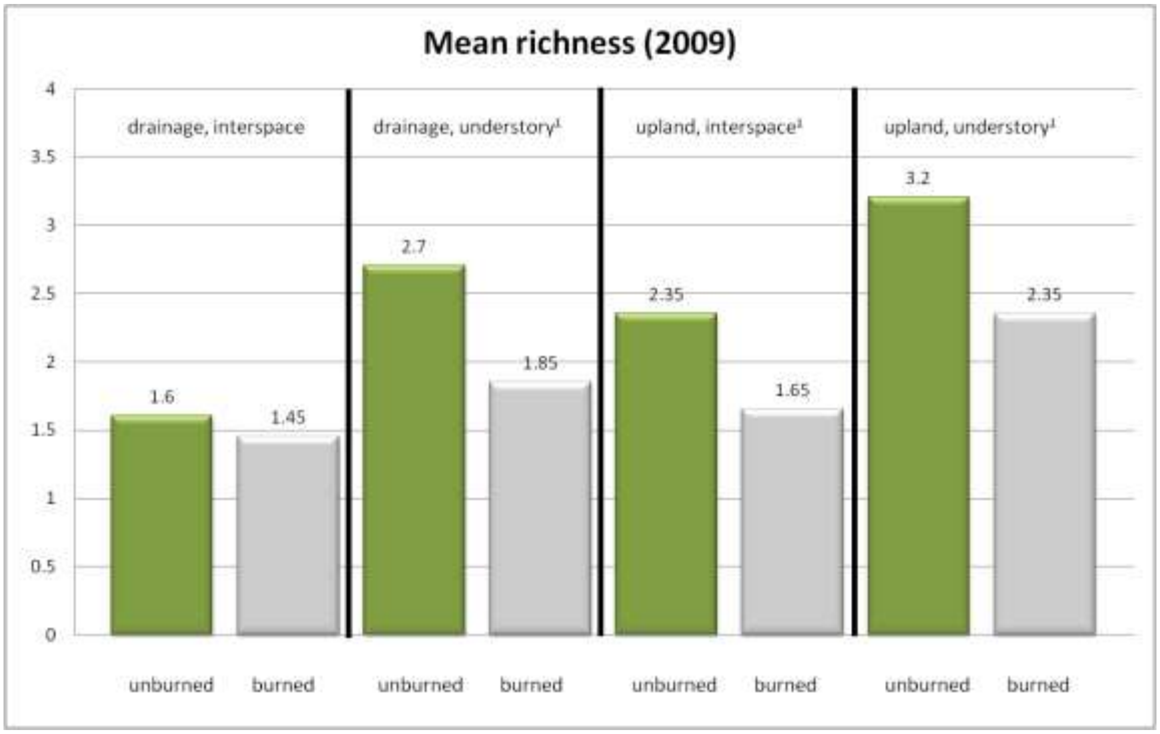


Figure 31. Mean richness (labeled) by plot type for 2009. [1] indicates a significant difference.

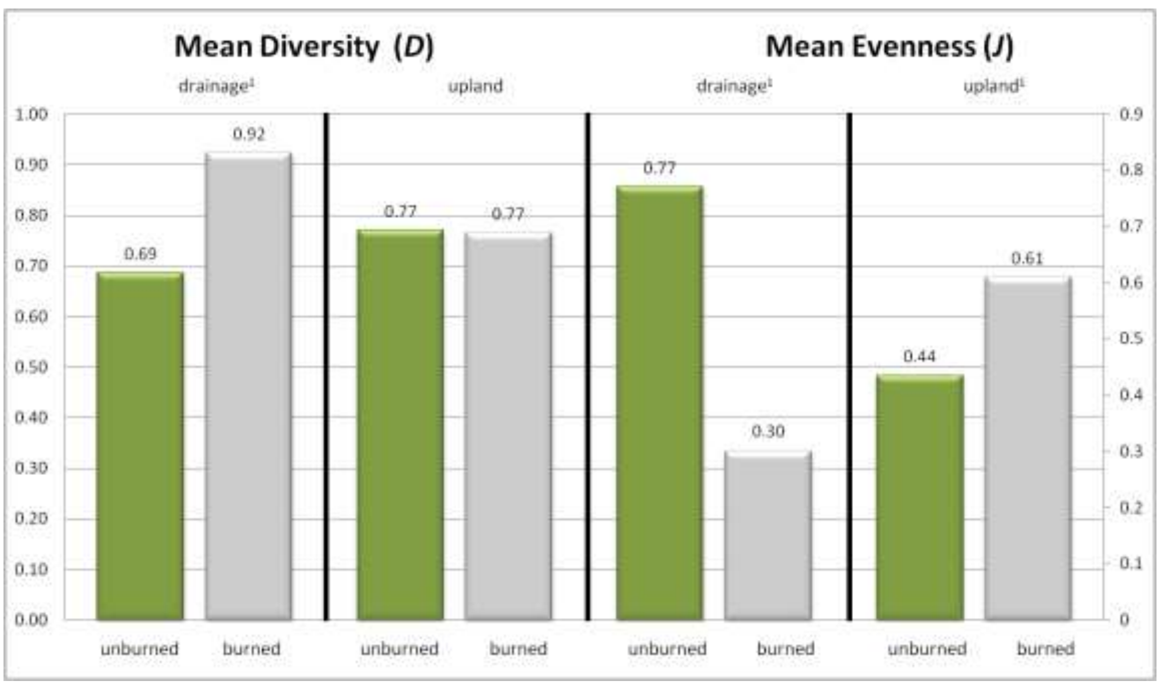


Figure 32. Mean diversity (Simpson's Index) and evenness for drainage and upland locations, com paring burned and unburned plots (2009). [1] indicates a significant difference. Diversity (*D*) ranges from 0 (infinite diversity) to 1 (no diversity). Evenness (*J*) ranges from 0 (skewed) to 1 (even).

Evaluating interspace plots, there was no difference in diversity between unburned ($\overline{D}$ =0.782) and burned ($\overline{D}$ =0.855) plots ($t(68) = -1.44$, $p = 0.1548$) and both exhibited similar evenness ($t(35) = 0.27$, p =0.7868) (unburned J'= 0.547; burned J'= 0.521). These sites have low diversity in general. For understory plots, unburned plots ($\overline{D}$ =0.704) were more diverse than burned ($\overline{D}$ =0.836) (t(65 = -2.76, $p = 0.0075$) although both returned values showing relatively homogeneous conditions. Burned (J'= 0.424) and unburned (J'= 0.516) understory sites showed no difference in evenness ($t(52) = 1.14$, $p = 0.259$).

The pattern of evenness was random in both the unburned plots (I = 0.004, z = 0.369, $p = 0.712$) and burned plots (I = 0.084, z = 1.60, $p = 0.109$). For diversity (Simpson's Index) the pattern in the unburned plots was random (I = -0.093, z = -1.32, $p = 0.186$) while the diversity pattern in burned plots was clustered (I = 0.118, z = 2.185, $p = 0.029$).

Cheatgrass was identified in 52 of 140 plots surveyed. Eighty-eight plots (63%) had no cheatgrass in 2009. It was identified in 41 (59%) unburned plots and in 11 (16%) burned plots. Mean dominance of cheatgrass in unburned plots where cheatgrass was identified was 0.584 ± 0.329. For burned plots the mean was 0.031 ± 0.029. The difference in cheatgrass dominance between burned and unburned plots was significantly different where unburned plots had predominantly more cheatgrass than burned plots ($t(50) = 5.53$, $p < 0.0001$). The dominance distribution for B. tectorum in burned and unburned plots is shown in Figure 33.

For plots surveyed in drainages, cheatgrass was more predominant in unburned plots (0.56 ± 0.25) than in burned plots (0.033 ± 0.033) ($t(19) = 0.525$, $p < 0.0001$). In total, 39 of 50 drainage plots had no cheatgrass identified, 13 unburned plots had none and eight burned plots had none. Of the 80 plots surveyed in upland areas, cheatgrass was identified in 28 unburned plots and in only 3 burned plots. Mean dominance in the unburned plots (0.596 ± 0.365) was significantly higher than in burned plots (0.025 ± 0.018) ($t(29) = 2.67$, $p = 0.0123$).

Blackbrush was identified in 36 of 140 plots surveyed. Mean blackbrush dominance for unburned plots (0.076 ± 0.062) was significantly lower than burned plots (0.466 ± 0.334) ($t(34) = -4.31$, $p = 0.0001$). Blackbrush was identified in two unburned drainage plots and was not found in any burned drainage plots. Blackbrush was identified in 22 burned upland plots and in 12 unburned upland plots. Blackbrush exhibited significantly higher dominance in burned upland plots (0.466 ± 0.334) than in unburned upland plots (0.064 ± 0.058) ($t(32) = -4.11$, $p = 0.0003$). This is not an unexpected result within one year post-burn.

<u>Vegetation Survey Results – 2010</u>

A total of 27 species were recorded across the 80 plots sampled in the unburned area and the 70 plots sampled in the burned area in 2010. Vegetation lists and basic summary statistics of density and richness are presented in Appendix H for each sampling unit category. Differences in vegetation density were identified between burned drainage sites, density in unburned upland sites, and in richness of unburned drainage sites. The spatial landscape pattern for 2010 was not analyzed due to insufficient sample size.

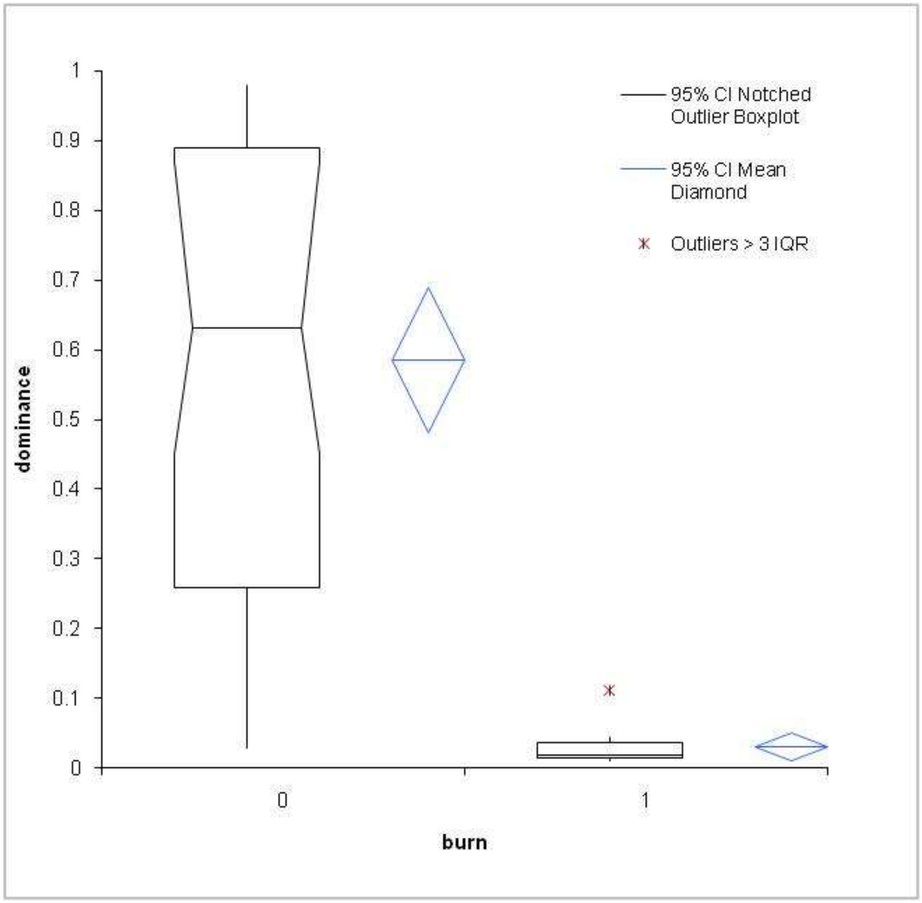


Figure 33. *Bromus tectorum* dominance for 2009 plots comparing unburned (0) and burned (1) conditions.

In 2010, approximately two years post-fire, the species recorded in burned plots were similar but also showed signs of diversion to those in unburned plots with the exception of cheatgrass. Approximately 50 percent of the species reported were shared in both the burned and unburned landscape. Twenty-two percent of species were unique to the burned landscape and just over one-fourth of the reported species were unique to the unburned landscape. Two exotic species were found—cheatgrass and redstem stork's bill (*Erodium cicutarium*). Cheatgrass was identified three times more often in unburned plots than in burned plots. Redstem stork's bill was reported only in one burned plot. Whitestem blazingstar, a native, was highly prevalent across the landscape regardless of having been burned or not. Sixweeks fescue was found nearly three times as often and blackbrush was recorded four times as often in the burned landscape.

By 2010, two years after the fire, the annual native whitestem blazingstar was more than 75 percent dominant in 41 burned plots of which 30 were drainage plots and was 100 percent dominant (the only species recorded) in seven burned plots. Five different species were more than 50 percent dominant in burned plots, and six species were the same in unburned plots. Only blackbrush and whitestem blazingstar were greater than 50 percent dominant in both the burned and unburned landscape. This is important in terms of diversity, which could be considered relevant to fire, soil stability, biomass, and erosion.

Both burned and unburned plots had a minimum of zero plants recorded. Burned and unburned mean density and richness for drainage and upland plots are presented in Figure 34. Mean plot density by plot type is presented in Figure 35 and Figure 36, respectively. At the landscape level, results showed a significant difference in the mean plant density between burned (174.5 ± 180.9/plot) and unburned (45.9 ± 45.5/plot) plots ($t(148) = -6.14$, $p < 0.0001$) as well as a significant difference in variability of density between burned and unburned plots ($F(79, 69) = 0.06$, $p < 0.0001$). Mean richness of unburned plots was 3.7 ± 1.4 and for burned plots was 3.5 ± 1.7. There was no significant difference in richness or richness variability of unburned and burned plots ($t(148) = 0.74$, $p = 0.46$) and ($F(79, 69) = 0.70$, $p = 0.131$), respectively.

Post-burn drainages exhibited greater density but fewer species than the unburned drainages. The higher density of plants would be expected to have greater soil stability properties. Because the dominance of highly flammable cheatgrass was significantly lower in burned areas and specifically less dominant in drainages ($t(44) = 4.48$, $p < 0.0001$), the promulgation of fire from cheatgrass would be low until there is a significant change in vegetation characteristics at this site. The upland areas exhibited similar properties regardless of burn status, i.e., greater density but fewer species.

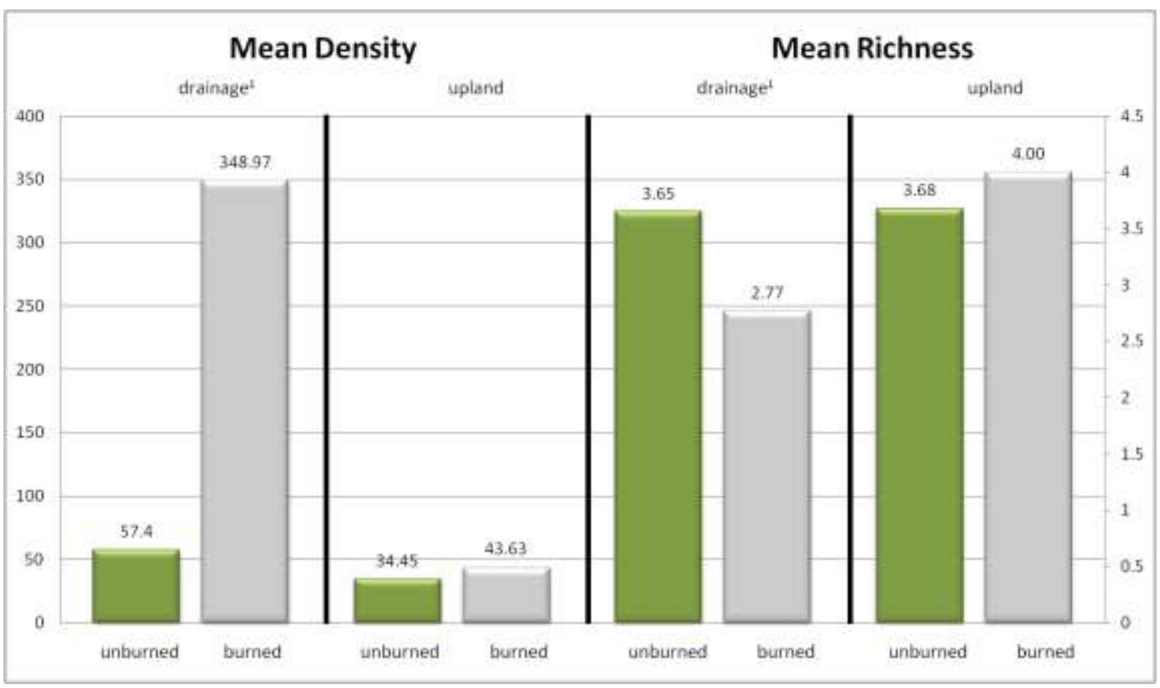


Figure 34. Mean density and mean richness (labeled) for all unburned and burned plots in each geomorphologic setting; drainage or upland (2010). [1] indicates a significant difference.

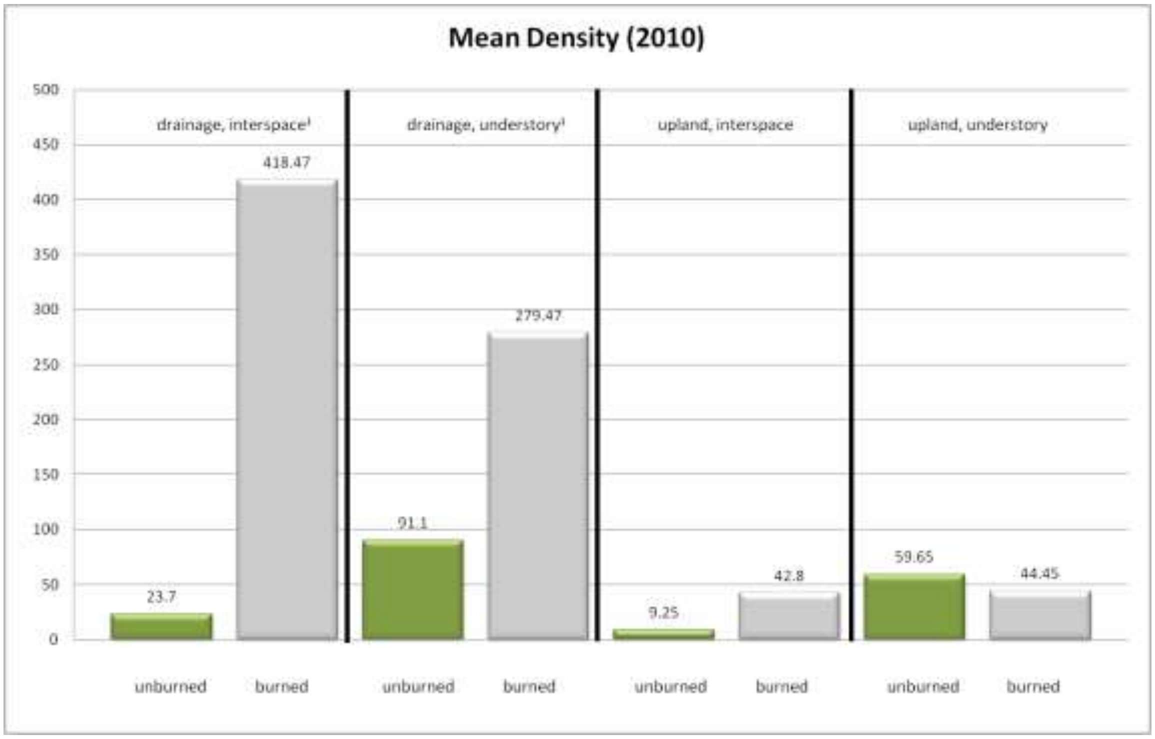


Figure 35. Mean density (labeled) by plot type for 2010. [1] indicates a significant difference.

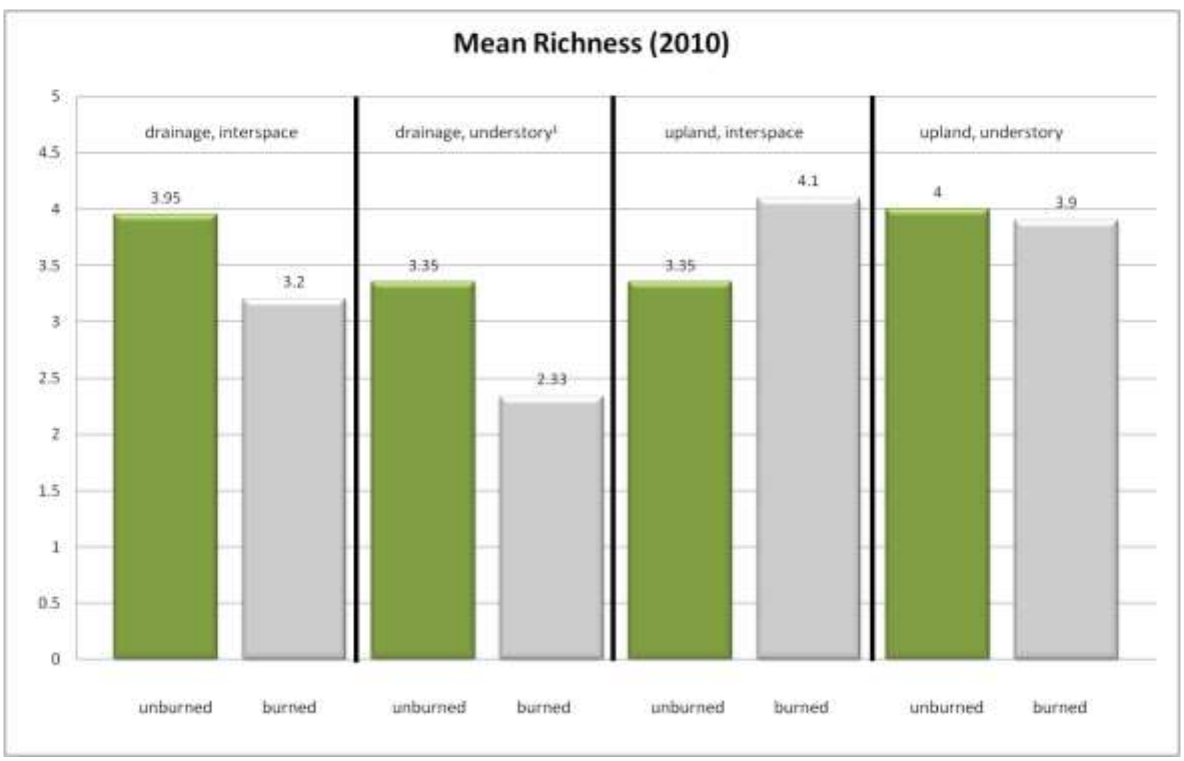


Figure 36. Mean richness (labeled) by plot type for 2010. [1] indicates a significant difference.

Results from Shannon-Weiner index showed unburned areas ($\overline{H'} = 0.906$) were significantly more diverse than burned ($\overline{H'} = 0.588$) areas ($t(135) = 4.02$, $p < 0.0001$). Simpson's Index (D) showed that unburned areas ($\overline{D} = 0.511$) were significantly more diverse than burned areas ($\overline{D} = 0.727$) ($t(142) = -5.38$, $p < 0.0001$). Burned area $\overline{D}$ approaching 1 indicate low diversity. Mean evenness for burned ($J' = 0.426$) and unburned ($J' = 0.696$) areas was also significantly different ($t(135) = 5.32$, $p < 0.0001$). Diversity and evenness comparisons for burned and unburned, drainage and upland plots, respectively, are presented graphically in Figure 37. In evaluating interspace sites, there was a significant difference in diversity between unburned ($\overline{D} = 0.463$) and burned ($\overline{D} = 0.728$) plots ($t(68) = -4.87$, $p < 0.0001$) and in evenness ($t(66) = 5.30$, $p < 0.0001$) (unburned $J' = 0.767$; burned $J' = 0.407$). Unburned interspace sites were more even in species distribution and the mean value approached 1. For 'understory' microsite plots, unburned plots ($\overline{D} = 0.555$) were more diverse than burned ($\overline{D} = 0.726$) ($t(72) = -2.93$, $p = 0.005$). Burned ($J' = 0.447$) and unburned ($J' = 0.634$) understory sites showed a difference in evenness ($t(67) = 2.49$, $p = 0.015$) although neither condition represented a distribution that was either even or skewed.

Two years post-fire, cheatgrass was identified in 81 of 150 plots surveyed. Sixty-nine plots (46%) had no cheatgrass in 2010. It was identified in 61 (76%) unburned plots and in 20 (29%) burned plots. Mean dominance of cheatgrass in unburned plots where identified was 0.377 ± 0.269. For burned plots the mean was 0.032 ± 0.059.

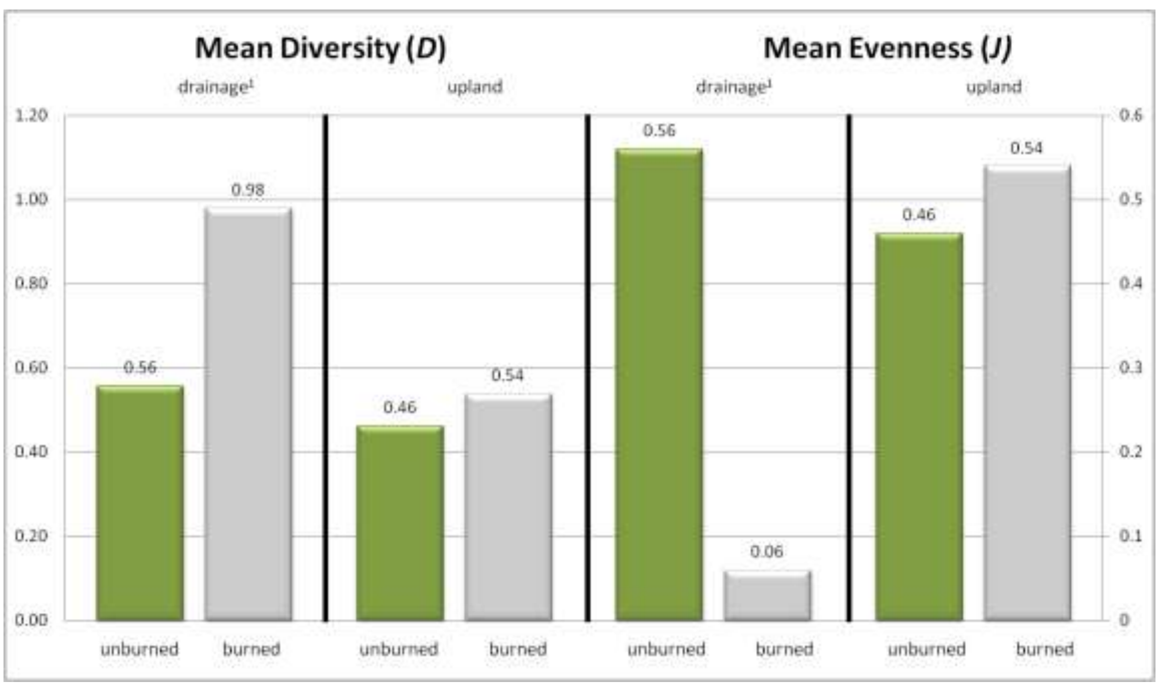


Figure 37. Mean diversity (Simpson's Index) and evenness for drainage and upland locations, comparing burned and unburned plots (2010). [1] indicates a significant difference. Diversity (D) ranges from 0 (infinite diversity) to 1 (no diversity). Evenness (J) ranges from 0 (skewed) to 1 (even).

The difference in cheatgrass dominance between burned and unburned plots was significantly different where unburned plots had predominantly more cheatgrass than burned plots (t(79) = 5.68, $p < 0.0001$). The dominance distribution for *B. tectorum* in burned and unburned plots is shown in Figure 38.

For plots surveyed in drainages, cheatgrass was more predominant in unburned plots (0.26 ± 0.26) than in burned plots (0.002 ± 0.003) (t(68) = 5.52, $p < 0.0001$). In total, 24 of 70 drainage plots had no cheatgrass identified, seven unburned plots had none and 17 burned plots had none. Of the 80 plots surveyed in upland areas, cheatgrass was identified in 28 unburned plots and in only 7 burned plots. Mean dominance in the unburned plots (0.32 ± 0.311) was significantly higher than in burned plots (0.015 ± 0.045) (t(78) = 6.05, $p < 0.0001$).

Blackbrush was identified in 30 of 150 plots surveyed. Mean blackbrush dominance in unburned plots where it occurred (0.242 ± 0.38) was not significantly different than burned plots where it was recorded (0.140 ± 0.16) (t(28) = 1.04, $p = 0.309$). Blackbrush was not identified in any unburned drainage plots and was found in three burned drainage plots. It was identified in 24 burned upland plots and in six unburned upland plots.

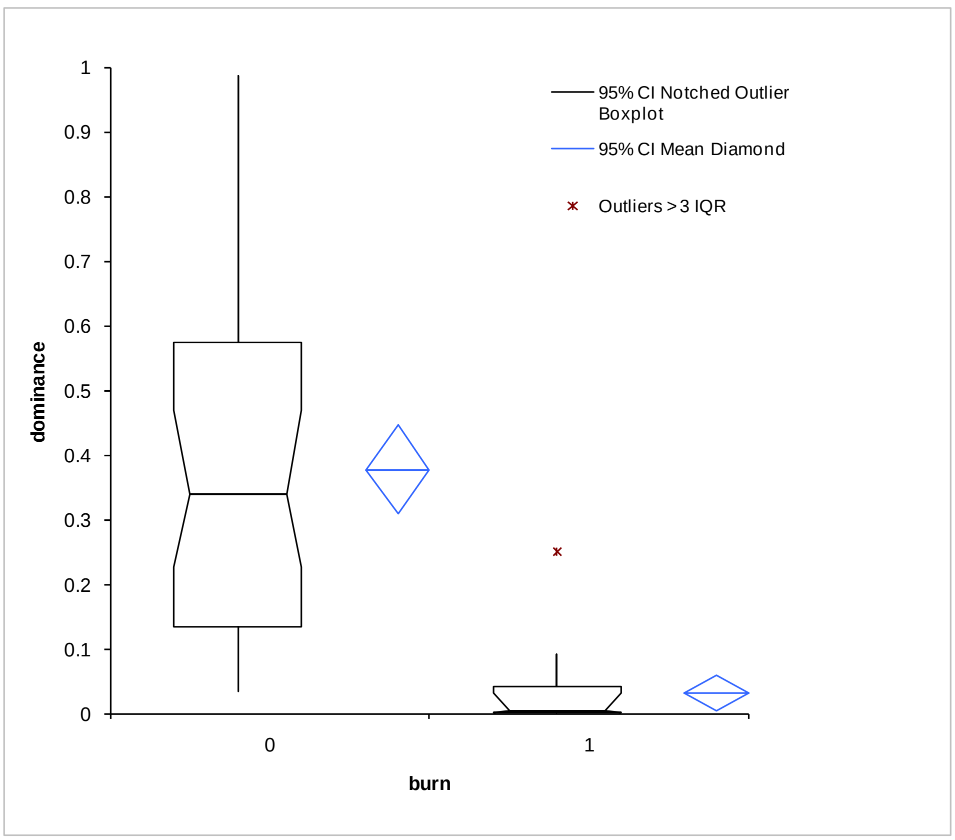


Figure 38. *Bromus tectorum* dominance for 2010 plots comparing unburned (0) and burned (1) conditions.

Vegetation Survey Results - 2011

A total of 80 plots were sampled in each of the unburned and burned areas for a total 160 plots, in 2011 and a total of 26 species were identified. Vegetation lists and basic summary statistics of density and richness are presented in Appendix H. Significant differences in density were identified between burned upland sites, density and richness in unburned upland and drainage microsites, respectively.

Sixty-five percent of the species recorded in 2011, approximately three years post-fire, were reported in both burned and unburned areas. Twenty-three percent of the species recorded were found only in the burned landscape and approximately 11 percent were reported only in unburned areas. Two exotic species, cheatgrass and redstem stork's bill, were reported in both burned and unburned plots, which is not surprising given their occurrence the year prior. Sixweeks fescue and desert pincushion were both reported nearly twice as often in unburned areas than burned. The prevalence of cheatgrass was nearly equal in both burned and unburned landscapes.

Burned drainages were much less dense but exhibited higher species richness than unburned drainages. Although the upland area plant density was no different regardless of burn status, the burned uplands exhibited higher richness than unburned areas. The understory sites in unburned drainages exhibited six times the density of unburned drainages and more than twice the density in unburned uplands. The reverse held true for interspace sites, where burned areas had much higher recorded plant densities and richness. In terms of overall metrics of diversity, unburned drainages were less diverse than burned, and unburned drainages exhibited almost no diversity. Burned drainages also exhibited a more even distribution of species richness than unburned drainages. Upland areas were equally homogeneous in terms of diversity and both burned and unburned uplands exhibited a more even distribution of species, although unburned areas significantly more so.

Unburned plots had a minimum of zero plants recorded while burned plots had a minimum of three. Burned and unburned mean density and richness results for drainages and uplands are presented in Figure 39. Mean plot density by plot type is presented in Figure 40 and Figure 41, respectively. At the landscape level, results showed a significant difference ($t(158) = 2.86$, $p = 0.0049$) in the mean plant density between burned (70.6 ± 91.9/plot) and unburned (38.3 ± 41.7/plot) plots as well as a significant difference in variability of density between burned and unburned plots ($F(79, 79) = 4.86$, $p < 0.0001$). Mean richness of unburned plots was 3.1 ± 2.1 and for burned plots was 4.7 ± 1.8. There was a significant difference in richness of unburned and burned plots ($t(158) = -5.03$, $p < 0.0001$). Richness variability within plots was not different ($F(79, 79) = 1.46$, $p = 0.092$).

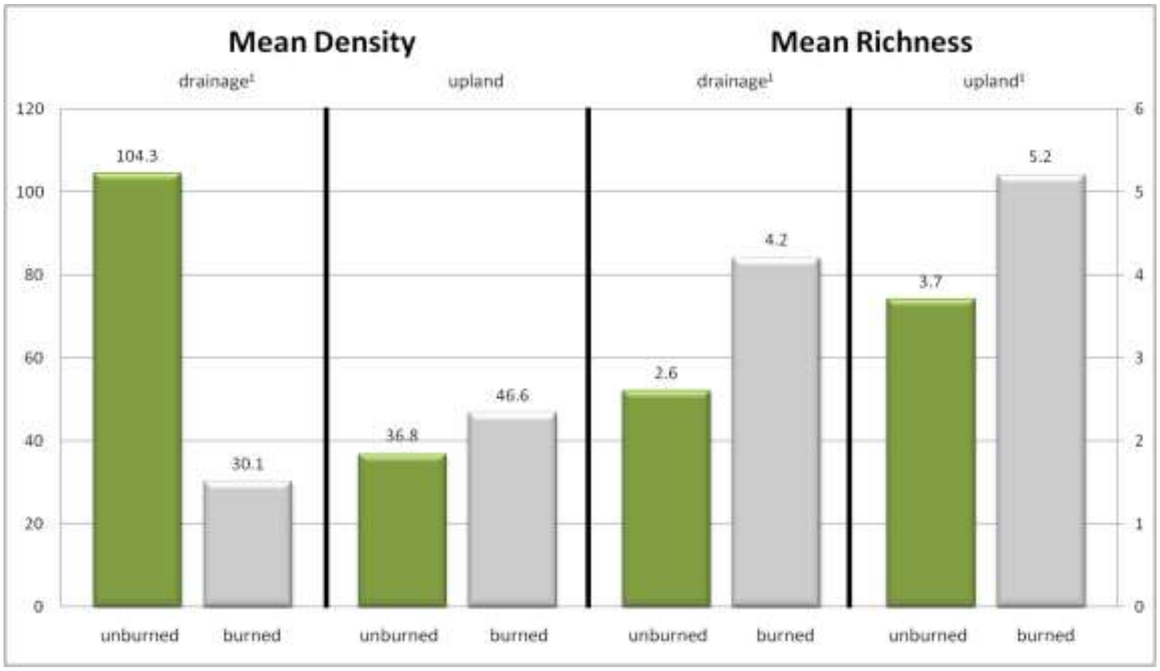


Figure 39. Mean density and mean richness (labeled) for all unburned and burned plots in each geomorphic setting, drainage or upland (2011). [1] indicates a significant difference.

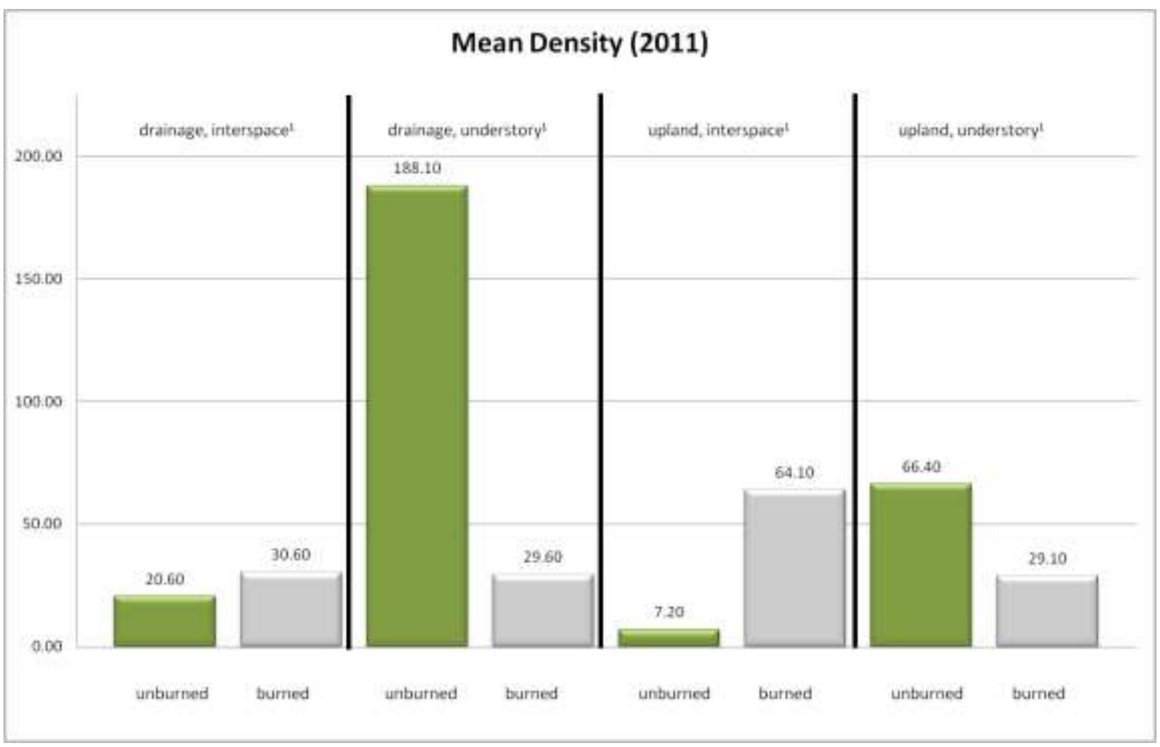


Figure 40. Mean density (labeled) by plot type for 2011. [1] indicates a significant difference.

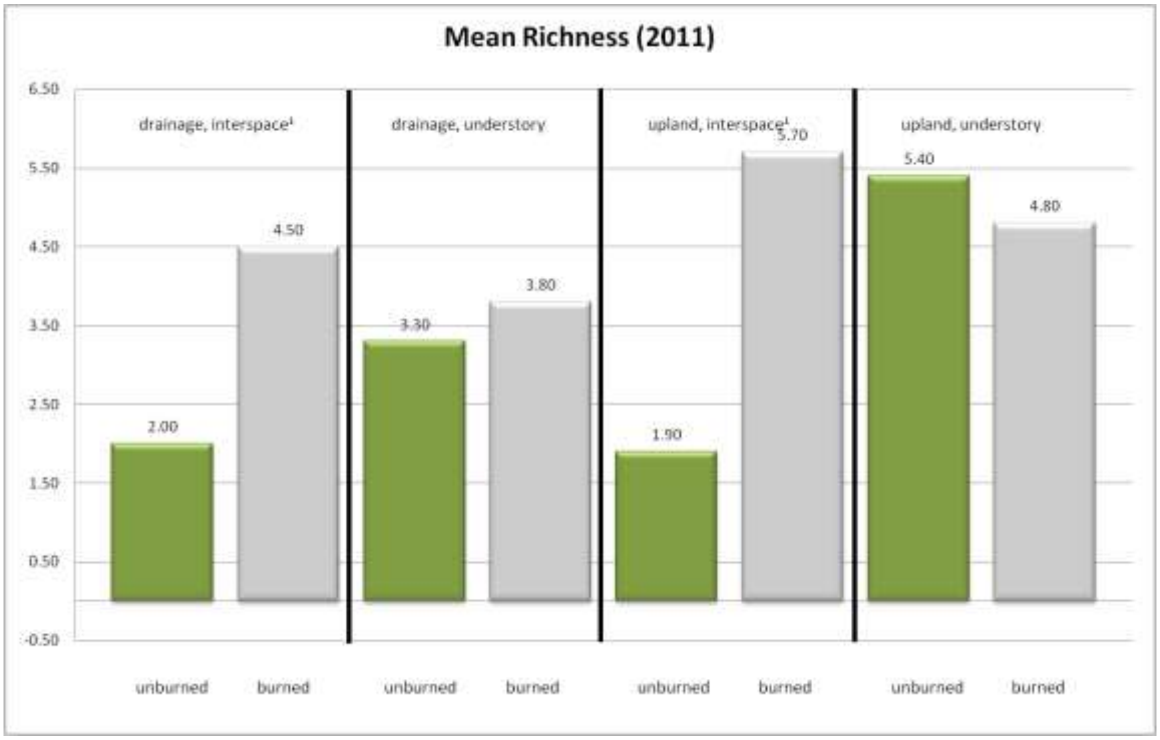


Figure 41. Mean richness (labeled) by plot type for 2011. [1] indicates a significant difference.

Results from Shannon-Weiner index showed a difference in diversity ($t(134) = -3.11$, $p = 0.002$) between unburned areas ($\overline{H}' = 0.784$) and burned areas ($\overline{H}' = 1.011$) areas. Simpson's Index (D) likewise showed that burned areas ($\overline{D} = 0.492$) were significantly more diverse ($t(152) = 4.74$, $p < 0.0001$) than unburned areas ($\overline{D} = 0.674$). As $\overline{D}$ approaching 1 is indicative of low diversity and although there was a difference between burned and unburned areas, neither would be considered relatively high in diversity by this metric. Mean evenness for burned ($J' = 0.682$) and unburned ($J' = 0.588$) areas was also significantly different ($t(134) = -2.25$, $p = 0.026$). Diversity and evenness comparisons for burned and unburned, drainage and upland plots, respectively, are presented graphically in Figure 42.

There was a significant difference ($t(72) = 4.22$, $p < 0.0001$) in diversity between unburned ($\overline{D} = 0.734$) and burned ($\overline{D} = 0.518$) interspace sites but not a difference in evenness ($t(60) = 1.15$, $p = 0.255$) (unburned $J' = 0.689$; burned $J' = 0.620$). For understory sites, unburned plots ($\overline{D} = 0.622$) were less diverse ($t(78) = 2.82$, $p = 0.006$) than burned ($\overline{D} = 0.465$). Burned ($J' = 0.748$) and unburned ($J' = 0.526$) understory sites showed a difference in evenness ($t(72) = -3.97$, $p = 0.0002$).

Three years post-fire, cheatgrass was identified in 124 of 160 plots (78%) surveyed. Fifty-six plots had no cheatgrass in 2011. It was identified in 64 (80%) unburned plots and in 60 (75%) burned plots. Mean density of cheatgrass was higher in unburned plots (68.4 ± 95.96) than in burned plots (5.6 ±12.12) Mean dominance of cheatgrass in unburned

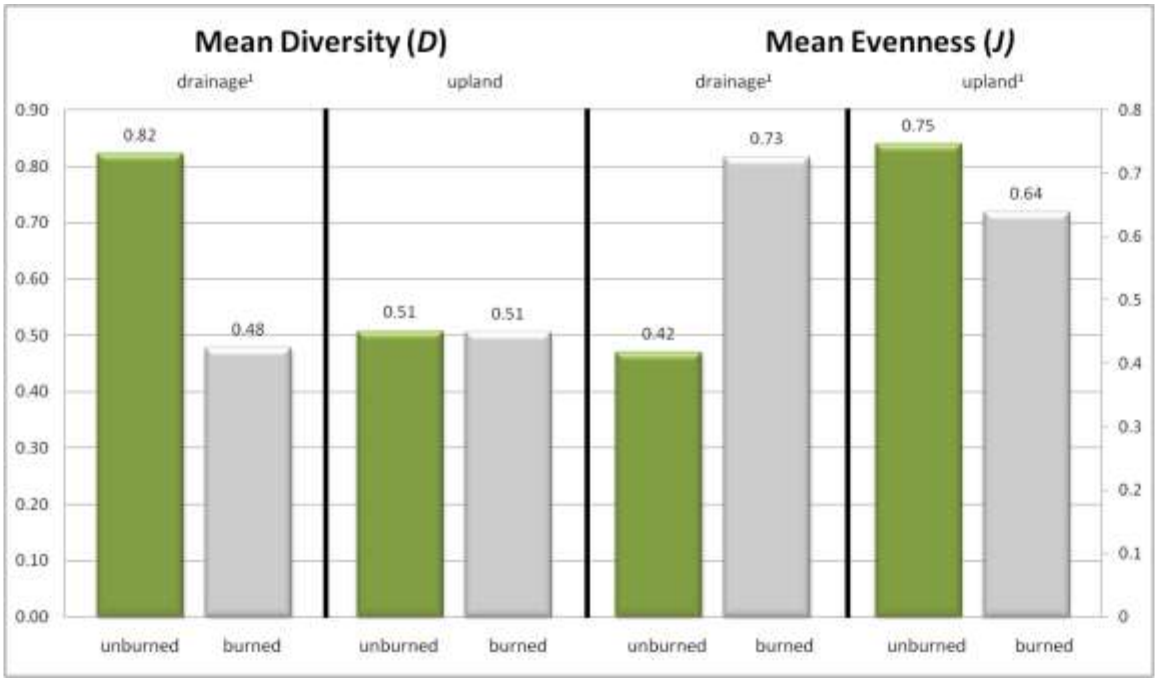


Figure 42. Mean diversity (Simpson's Index) and evenness for drainage and upland locations, comparing burned and unburned plots (2011). [1] indicates a significant difference. Diversity (*D*) ranges from 0 (infinite diversity) to 1 (no diversity). Evenness (*J*) ranges from 0 (skewed) to 1 (even).

plots where identified was 0.638 ± 0.335. For burned plots the mean was 0.182 ± 0.203. The difference in cheatgrass dominance between burned and unburned plots was significantly different where unburned plots had predominantly more cheatgrass than burned plots (t(122) = 9.08, $p < 0.0001$). The dominance distribution for *B. tectorum* in burned and unburned plots is shown in Figure 43.

In drainages, cheatgrass was more predominant (t(67) = 10.95, $p < 0.0001$) in unburned plots (0.848 ± 0.224) than in burned plots (0.248 ± 0.229). In total, 10 of 80 drainage plots had no cheatgrass identified, one unburned plots had none and nine burned plots had none. Of the 80 plots surveyed in upland areas, cheatgrass was identified in 26 (33%) unburned plots and in 29 (36%) burned plots. Mean dominance in the unburned plots (0.33 ± 0.207) was significantly higher (t(53) = 4.58, $p < 0.0001$) than in burned plots (0.1125 ± 0.144).

Blackbrush was identified in 22 of 160 plots (14%) surveyed. Mean blackbrush dominance in unburned plots where it occurred (0.465 ± 0.5) was marginally significantly different (t(20) = 2.11, $p = 0.0472$) than burned plots where it was recorded (0.123 ± 0.219). Blackbrush was not identified in any burned drainage plots and was found in six unburned drainage plots. It was identified in 13 burned upland plots and in three unburned upland plots.

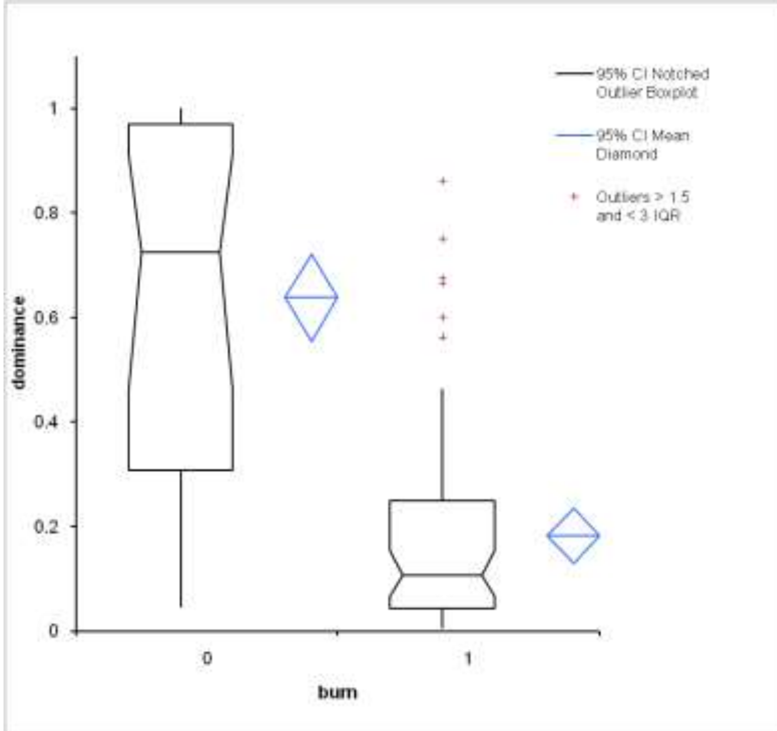


Figure 43. *Bromus tectorum* dominance for 2011 plots comparing burned (1) and unburned (0) conditions.

Vegetation - Discussion

In 2009, the species composition of burned areas was similar to the unburned areas. However, vegetation densities, frequency of occurrence, and diversity were different. The relationship between richness and density in both burned and unburned landscapes was positive although the correlation in unburned sites was stronger. The primary differences were found in the upland sites, which had lower density and richness where burned. Within the upland landscape, the primary difference was observed where fire had burned shrub canopies. Where shrubs burned, so did the understory vegetation and it did not regenerate to reflect unburned densities within one year post-fire. Nor did the burned uplands return to the unburned richness within one year.

There was a difference in regeneration in drainages one year post-fire in the burned landscape as compared to unburned areas. Diversity in burned drainages was significantly lower than in unburned drainages, and was almost non-existent. The distribution of the species and abundance in burned drainages was skewed towards a few species unlike unburned drainages, which exhibited a much more even species distribution.

Plant density was influenced by site location within the burned landscape but this did not hold true in terms of diversity. Understory sites had significantly higher plant densities and richness. There was no difference in diversity or evenness within the burned landscape whether interspace or understory. The vegetation community structure of the unburned landscape, however, had significantly higher plant densities and species richness under the canopy than in interspace sites. Diversity and evenness between interspace and understory sites was comparable for both unburned and burned sites.

Upland areas that had burned were found to have nearly half the plant density as in the unburned landscape. The burned upland sites also had significantly lower species richness. However, the difference in cheatgrass predominance between the burned (very low) and unburned (high) landscapes should not be overlooked. Cheatgrass was 35 times denser in the unburned sites than the burned sites overall and 46 times as dense in unburned uplands than burned uplands. The risk of fire recurrence due to cheatgrass regeneration in previously burned sites is relatively low compared to fire risk in unburned areas. However, any soil stability benefits from cheatgrass in burned areas would not likely be realized due to the low numbers of plants. Because the overall density in unburned uplands was more than 4.5 times greater than in the burned upland areas, soil stabilizing benefits from sheer number of plants one year post-fire would likely be less until vegetation recovers to a greater extent.

In drainages, the plant density in unburned sites was approximately twice that of burned sites. For cheatgrass specifically, the density difference in unburned versus burned sites was 31 times greater. Soil stability benefits from generic plant density would be expected to be greater in drainages than in upland sites, but still would not be comparable to an unburned landscape based solely on plant density.

In 2010, two years post-fire, the species composition of burned areas was less similar to the unburned areas than observed in 2009. In 2010 vegetation densities, species composition, richness, and diversity differed between burned and unburned sites. Within burned and unburned sites neither was found to exhibit a significant relationship between richness and density. The primary differences were found between drainage and upland areas. Burned drainages were six-times denser than unburned drainages while plant densities in burned and unburned uplands did not differ. The density differences in interspace and understory drainage sites were observed. Richness was lower in burned than unburned drainages but was not different in uplands. Burned drainages exhibited an eight-fold higher vegetation density over burned uplands. However, the upland landscape had greater richness than drainages. From a diversity perspective, drainages exhibited almost no diversity and upland sites were significantly more diverse than drainages. Upland sites also showed more even species distribution.

Two years post-fire showed large differences in drainage sites in particular. Density and richness was much greater in burned drainages than unburned drainages whereas there were no differences in these metrics for uplands. Both interspace and understory sites within burned drainages showed consistently greater plant density than unburned drainages. Richness of burned drainages was consistently lower but only significantly so for understory sites. Diversity in burned drainages was significantly lower than in unburned drainages, and was almost none. The distribution of species and abundance in burned drainages was skewed towards one or two species, namely Indian ricegrass > blackbrush > cheatgrass. The first two species are natives. While significantly different from skewed, unburned drainages were not entirely even in distribution.

Interspace or understory locations did not appear to afford any advantage in terms of vegetation re-establishment or community diversity. Neither plant density nor richness varied by interspace or understory site within the burned landscape. There was also no difference in diversity or evenness within the burned landscape by site. These results indicate that two years after the fire, the burned landscape had not regained a vegetation community structure similar to the unburned landscape. The unburned landscape had significantly higher plant densities. However, as in the burned landscape, species richness within unburned microsites did not differ. Diversity and evenness of unburned interspace sites were greater than unburned understory sites.

Plant densities were found to be consistently greater in the burned than in the unburned landscape. Within the burned landscape, it is more likely erosion would occur in upland areas versus drainages based solely on plant density. Upland areas that had burned were found to have 12 percent of the density of burned drainages. Species richness was no different between the burned and unburned landscape. However, the difference in cheatgrass predominance between the burned (very low) and unburned (high) landscapes should not be overlooked. Cheatgrass was 18 times denser in the unburned sites than in the burned sites overall, and nearly 30 times as dense in unburned uplands than burned uplands. The risk of fire recurrence due to cheatgrass regeneration in previously burned sites is relatively low compared to fire risk in areas previously unburned. However, any soil stability benefits from cheatgrass in burned areas would not likely be realized. Because the overall density in unburned uplands was nearly 3 times greater than in the burned upland landscape, soil stabilizing benefits from sheer number of plants two years post-fire would likely be less until vegetation recovers to a greater extent.

In drainage areas, the plant density in unburned sites more closely approximated that of burned sites and was less than twice as great. For cheatgrass specifically, the density difference in unburned versus burned sites was 15 times greater for the unburned drainages. Soil stability benefits from generic plant density would be expected to be greater in drainages than in upland sites, but still would not be comparable to an unburned landscape based solely on plant density.

In 2011, three years after the fire and the last year of measurements, the species composition of burned areas more closely resembled the species overlap observed in 2009 than 2010. More species were identified in the burned landscape than the unburned landscape. Cheatgrass and sixweeks fescue were the most commonly identified species found in the unburned landscape and these two species were also highly prevalent in burned sites along with desert pincushion. Cheatgrass frequency in burned plots approached that of unburned plots in 2011.

Comparing burned and unburned sites, a significant relationship was reported between richness and density and was stronger in the unburned landscape. The primary differences were found between upland and drainage sites. Vegetation densities in burned drainages were three times more dense than unburned drainages while plant densities in burned and unburned uplands were similar. Density differences were observed comparing interspace and understory sites as well. Richness was lower in burned versus unburned drainages and in both understory and interspace sites in upland burned and unburned locations. Burned drainages and uplands were not different in density, although the upland sites had greater richness than did the drainage sites. From a diversity perspective, there was no difference in diversity or richness across the burned landscape for both drainage and upland locations.

Three years after the fire, there were large differences in drainage sites both in density and richness. Vegetation density was much lower in burned drainages than unburned drainages whereas richness was greater in burned than unburned drainages. Density by microsites within burned drainages showed higher densities in burned interspace drainage sites and lower densities in burned drainage understory sites. Diversity in burned drainages was significantly higher than unburned drainages which were not very diverse. The distribution of the species and abundance in burned drainages was relatively even and significantly more so than unburned drainages. This is due to the fact that although three species were predominant (cheatgrass, sixweeks fescue, whitestem blazingstar) the other species identified were recorded in relatively greater numbers than in previous years.

In 2011, the interspace areas appeared to have provided some advantage in terms of vegetation re-establishment in both density and richness. All sites were significantly different in the burned landscape compared with the unburned landscape, and interspace sites within the burned area had greater plant density than in the equivalent unburned sites. The understory sites showed much lower densities in burned areas compared to unburned areas. The burned interspace sites were also consistently richer. In terms of biodiversity, drainages were approximately half as diverse in the burned compared to the unburned landscape, although that diversity was fairly evenly distributed. Uplands were no different between burned and unburned sites, and both exhibited an evenness metric that was closer to even than skewed.

These results indicate that three years post-fire, the burned landscape still does not reflect the vegetation community structure of the unburned landscape, and that site location has an effect, albeit counterintuitive. Because burning canopies release nutrients for subsequent vegetation germination and growth, understory canopy sites would be expected to have a higher plant density than interspace sites, which was not the case for this particular fire location.

The 2011 data suggest that drainages and burned understory sites would be more prone to erosion due to lower plant densities, but interspace sites are not as likely to promote increased erosion. This is because plant densities were found to be low in the burned area in understory sites whether in drainages or uplands. The understory sites in drainages are the primary source of the difference between the burned and unburned landscape. Species richness was consistently higher in the burned landscape. Again, the difference in cheatgrass predominance between the burned (low) and unburned (high) landscapes should not be overlooked. Cheatgrass was 12 times denser in the unburned sites than the burned sites overall, and seven times as dense in unburned uplands than burned uplands. In drainages, the difference in density was twelve and a half times for unburned compared to burned areas. The risk of fire recurrence due to cheatgrass regeneration in previously burned sites remains relatively low compared to fire risk in areas previously unburned even three years post-fire. With the relatively low overall species densities in drainages, any soil stability benefits in burned areas would not likely be realized. Because the overall density in burned uplands was greater than in the unburned upland landscape, soil stabilizing benefits from sheer number of plants three years after the fire would likely be greater there than on the previously unburned landscape.

In drainages, the plant density for burned sites was significantly less than that of unburned sites. For cheatgrass specifically, the density difference in unburned versus burned sites was 12 times greater. Soil stability benefits from generic plant density would be expected to be greater in uplands than in drainages; specifically, burned drainages would not be comparable to an unburned landscape based solely on plant density.

Vegetation - Summary and Conclusions

This study examined the regeneration and diversity of a burned and unburned blackbrush community at three discrete time periods after a fire. The resilience of a landscape to erosion from natural physical processes (e.g., wind, water) depends in a great part on the stability of the soil and, particularly, the vegetation cover. Vegetation can promote the integrity of the upper soil horizons, through direct water capture in rain events, and soil water use while subsurface biomass, namely the root structures of plants, help stabilize soil. Biomass varies with plant species, abundance, and spatial distribution. There is also a temporal component to biomass and root structure that will vary with community composition. Therefore, it is important not only to calculate basic metrics such as plant density and richness, but also abundance, dominance, and diversity metrics. Thus, there are several factors that come in to play in terms of maintaining a functional landscape where soil stability is concerned. Regeneration of the plant community can have a significant effect on landscape resilience, and return to ecological functionality after a wildfire.

One year after the fire, cheatgrass was observed in the burned area. Plant density in burned areas approximated that of the unburned landscape although burned areas lacked equivalent richness. An examination of the landscape in detail showed that although there was no difference at the broadest level, the vegetation density in burned drainage interspaces was 16 times denser than where burned, and was equally rich as unburned drainage interspace. Overall, the diversity one year post fire was substantially lower where the burn occurred with the exception of the drainage interspace as identified above. In terms of soil stability the density of plants should provide similar benefit in the burned landscape however the benefit of root mass of blackbrush or other shrub species would not be realized as shrubs did not substantially regenerate within one year. From an ecological standpoint the decrease in diversity would also be expected to have an effect on soil stability and the likelihood of fire recurrence. Burned drainages did not reflect the relative evenness of unburned drainages and exhibited almost no diversity. Diversity was equally low in burned and unburned uplands however the burned uplands exhibited a trend towards even distribution of the plant community diversity. Cheatgrass had not recolonized the burned area one year post fire to the extent that it was found in the unburned landscape. The majority of drainages sampled showed no cheatgrass at all, and where it was found, was usually in the unburned area. Cheatgrass in the burned uplands was rare. Blackbrush on the other hand did show signs of high regeneration within one year post fire in the uplands but was not found in any burned drainages.

Two years after the fire, the burned area exhibited very high plant densities, significantly greater than the adjacent unburned landscape. This was primarily a function of the six-fold higher densities measured in burned drainages over measurements from unburned drainages. Overall the richness of burned and unburned areas was comparable, but the unburned landscape remained overall more diverse. Examining drainages specifically, density in unburned drainages was no different between interspace and understory while in the burned drainages the interspace exhibited nearly twice the plant density of the understory. This difference was not observed in the burned uplands, which exhibited equal densities across sites. Burned drainages showed almost no diversity two years post fire. The burned uplands did not approximate the site density proportion of the unburned landscape, which was nearly twice as dense in the understory as the interspace but did exhibit comparable diversity. Although similarly even, neither burned nor unburned uplands would be considered to exhibit evenness in diversity, and burned drainages showed highly skewed diversity distributions indicating only a few species were dominating the community. Cheatgrass was three times as prevalent in the unburned landscape. Blackbrush occurrence was no different in burned and unburned areas although the size was different, where the shrubs had not yet established to the full size potential as compared with unburned areas. While there was no overall difference in blackbrush occurrence it was recorded four times more often in burned uplands than in unburned uplands. The implications for soil stabilization two years post fire, interpreted solely from the plant density, is that a positive effect simply due to biomass in the upper soil horizon could be expected. However the other aspect to biomass is its role as fuel and the likelihood of burning would be expected to increase with additional fuel loading. Shrubs had not regenerated sufficiently two years post fire for soil stabilization benefits to be realized through extensive root development.

Three years after the fire, there was a substantial difference in plant densities and richness between drainages and upland sites. In contrast, the burned landscape showed only a density difference between the understory and interspaces in sampled uplands. In the burned landscape, densities and richness were fairly uniform. This difference might appear counterintuitive initially, in that the understory would not be expected to have significantly lower plant densities three years into recovery; however due to the fact that the shrubs were completely killed in the fire, whether or not real effects gained from an understory are possible must be considered, however unlikely. The unburned drainages were shown to have three times the plant density as burned drainages although burned drainages exhibited greater species richness. Upland areas were equally dense although burned uplands also showed greater richness. Differences in richness were observed between burned interspace and unburned interspace areas; burned areas had higher richness. The understory sites exhibited similar richness for burned and unburned areas. From a diversity perspective, the uplands were equally diverse regardless of whether or not it had burned. The unburned uplands had a more even distribution of diversity compared to the burned landscape but neither was particularly even. Burned drainages were more diverse and the diversity was fairly evenly distributed while unburned drainages returned a generally low and skewed diversity. Cheatgrass was reported in three-quarters of the burned area, similar to the unburned area, and densities were higher in unburned plots. Cheatgrass was also more dominant in the community composition in unburned plots. Blackbrush was not reported in the burned drainages but was reported in unburned drainages. It was four times more prevalent in the burned uplands than the unburned uplands. Three years post fire the burned landscape continued to revegetate but had yet to approximate the condition of an unburned landscape. Cheatgrass, notorious for changing fire cycles, had increased in the burned area but so did other native species as well. Due to the uniformity in density and richness across the burned landscape, soil stabilization benefits from vegetation biomass would also be anticipated to perform consistently at a broad level. The lower density of the burned area would not be expected to support stabilization benefits to the same level as the unburned landscape. However there may be some additional value gained from the greater diversity of species in the burned landscape. The lower prevalence of cheatgrass coupled with higher diversity in the burned landscape might reasonably be anticipated to have an effect on fire recurrence, although no desert plant species can be considered fire resistant; any vegetation present whether annual forb, grass or perennial shrub constitutes fuel.

SUMMARY AND CONCLUSIONS

The purpose of this report was to present data and findings regarding the monitoring of wind and water erosion potential and vegetation response following a fire at a site comparable to the transition region between the Mojave and Great Basin on the NNSS. The site chosen—the Jacob Fire site—experienced a lightning-caused fire on August 6 – 8, 2008. Field measurements began one month after the fire and continued periodically for a three-year period. Burned areas sampled included ridge and drainage locations; plant canopies understory and between vegetation interspace sites were also sampled. These data were compared to data from comparable unburned sites to assess fire effects.

Wind erosion data were collected with a wind tunnel analog instrument (PI-SWERL) that measured emissions of particulate matter less than 10 µm (3.9×10^{-7} in) at different wind speeds. Emissions captured on exhaust filters on the PI-SWERL allowed for the determination of the chemical composition of the material. Water erosion data were generated by use of a rainfall simulator and measurement of collected runoff and suspended sediment. Laboratory analysis of the sediment provided data regarding the amount and size of particles present in the runoff. Vegetation was sampled in plots and analyzed for various diversity metrics and spatial pattern.

Analysis of the PM_{10} data indicated that the amount of emissions from burned soil surfaces was greater than for unburned soil surfaces, with emissions from burned drainage understory soils being consistently high. The data further showed the burned understory soils were major source areas for windborne material after a fire. These levels were nearly the same as unburned soils by 24 MAB. The chemical signature of the fire (EC:Soil ratio) began to fade about 12 MAB and was essentially absent by 24 MAB. Fire effects on wind erosion were conspicuous and lasted approximately two years. The amount of windborne particulate matter appeared unrelated to surface soil texture.

Little, if any, effect of the fire was apparent in either the occurrence or volume of runoff, or on the amount of sediment over the study period. Runoff from burned ridge interspace soils was similar to runoff from unburned interspace soil indicating these interspace areas were source areas of runoff and sediment, but there was no difference due to the fire. Runoff seldom occurred in drainage areas likely due to coarse soils there. The fire did affect the size of sediment in the runoff such that, on burned sites that had runoff, finer sediment was measured at 1 MAB, but by 3 MAB, this effect disappeared.

The marked difference in wind emission and runoff data highlights the complex relationship between fire and erosion. While wind and water erosion are different processes, observation and research indicate the overall erosion potential is elevated immediately after a fire then decreases over time. The wind erosion data from the Jacob Fire site substantiate this response. The water erosion data do not indicate a response to fire, although there may be some immediate effect on sediment size if runoff occurs after a fire. Thus, under the conditions simulated on the topographic, soil, and soil surface conditions at the Jacob Fire site, runoff may not be a significant erosional process and was relatively unaffected by fire there.

In contrast, the landscape was shown to be dynamic in terms of trajectory for revegetation. The three discrete time periods sampled showed the breadth of variability that exists when comparing burned and unburned areas. Of particular significance is the apparent opportunity to revegetate, to resist cheatgrass recolonization, and develop the biomass needed to assist in soil stabilization, which is provided by different geomorphologic landforms and by sites. The time period of vegetation recovery from a fire occurs on a different time scale than does soil stabilization based on the measurements reported here. The results from the vegetation surveys support the wind erosion results, where the primary source of windborne particles originate from the understory, where lower plant diversity and densities were found. However, the burned landscape did not reach the unburned community equivalent within 36 MAB. The soil appears to be more resilient and have a much shorter recovery time than the vegetation in this particular community.

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APPENDIX A: JACOB FIRE SITE MEASUREMENT DATES

Month	*Month*	*Year*	*MAB*	*Wind*	*Water*	*Vegetation*
8	8-Aug	2008	FIRE			
9	Sept	2008	1	X	X	
10	Oct	2008	2			
11	Nov	2008	3	X	X	
12	Dec	2008	4			
1	Jan	2009	5			
2	Feb	2009	6	X		
3	Mar	2009	7		X	
4	Apr	2009	8		X	
5	May	2009	9			X
6	Jun	2009	10			
7	Jul	2009	11			X
8	Aug	2009	12			
9	Sept	2009	13	X		
10	Oct	2009	14			X
11	Nov	2009	15		X	
12	Dec	2009	16			
1	Jan	2010	17			
2	Feb	2010	18			
3	Mar	2010	19			
4	Apr	2010	20			
5	May	2010	21	X		X
6	Jun	2010	22		X	
7	Jul	2010	23			
8	Aug	2010	24	X		
9	Sept	2010	25		X	
10	Oct	2010	26			
11	Nov	2010	27			
12	Dec	2010	28			
1	Jan	2011	29			
2	Feb	2011	30			
3	Mar	2011	31			
4	Apr	2011	32			
5	May	2011	33			X
6	Jun	2011	34	X	X	
7	Jul	2011	35			
8	Aug	2011	36	X		
9	Sept	2011	37			
10	Oct	2011	38			
11	Nov	2011	39			
12	Dec	2011	40			

APPENDIX B: SUPPLEMENTAL PI-SWERL CHEMICAL DATA

Table B-1. Emission profiles of PM_{10} components emitted from Unburned and Burned soils at one month after the fire (1 MAB).

	Unburned	**Burned Ridge**	**Burned Drainage**
Chloride,Cl	1.571 ± 6.733	0.285 ± 0.217	0.241 ± 0.204
Nitrate, NO_3		0.253 ± 0.218	0.286 ± 0.204
Sulfate, SO^{2-}	5.029 ± 6.748	8.533 ± 0.223	9.747 ± 0.212
Ammonium,NH_4^+	1.8 ± 6.783	0.219 ± 0.221	0.219 ± 0.207
Sodium, Na^+	2.371 ± 0.544	0.484 ± 0.046	2.027 ± 0.178
Magnesium, Mg^{2+}	7.886 ± 0.75	4.579 ± 0.105	4.723 ± 0.108
Potassium, K^+	2.6 ± 0.712	2.979 ± 0.079	3.461 ± 0.091
Calcium, Ca^{2+}	74.114 ± 7.278	86.929 ± 1.261	86.596 ± 1.252
Total OC	92.752 ± 62.186	73.5 ± 5.186	108.563 ± 6.8
Total EC	16.314 ± 12.923	123.25 ± 6.458	110.603 ± 5.806
Carbonate, CO_3	26.867 ± 59.322	84.608 ± 5.725	93.831 ± 6.076
Total Carbon	135.933 ± 92.092	281.358 ± 12.809	312.998 ± 13.869
Sodium, Na		4.305 ± 0.688	3.538 ± 0.637
Magnesium, Mg	0.867 ± 9.814	3.625 ± 0.336	2.756 ± 0.312
Aluminum, Al	20.543 ± 2.451	10.12 ± 0.109	5.911 ± 0.079
Silicon, Si	69.21 ± 6.301	28.171 ± 0.236	15.57 ± 0.146
Phosphorous, P		0.965 ± 0.038	1.318 ± 0.037
Sulfur, S	3.629 ± 1.639	0.831 ± 0.055	0.786 ± 0.051
Chlorine, Cl	0.924 ± 0.371	0.113 ± 0.012	0.08 ± 0.011
Potassium, K	22.314 ± 1.95	14.846 ± 0.06	8.559 ± 0.036
Calcium, Ca	84.124 ± 7.333	194.43 ± 0.689	139.414 ± 0.479
Scandium, Sc	0.01 ± 1.39		
Titanium, Ti	4.038 ± 0.426	1.858 ± 0.012	0.943 ± 0.009
Vanadium, V		0.03 ± 0.001	0.013 ± 0.001
Chromium, Cr			
Manganese, Mn	1.429 ± 0.483	1.008 ± 0.017	0.698 ± 0.015
Iron, Fe	45.143 ± 3.925	21.38 ± 0.085	10.417 ± 0.045
Cobalt, Co			
Nickel, Ni		0.004 ± 0.004	
Copper, Cu		0.063 ± 0.006	0.035 ± 0.006
Zinc, Zn	0.924 ± 0.242	0.225 ± 0.008	0.086 ± 0.007

Table B-1. Emission profiles of PM_{10} components emitted from Unburned and Burned soils at one month after the fire (1 MAB) (continued).

	Unburned	**Burned Ridge**	**Burned Drainage**
Gallium, Ga			
Arsenic, As			
Selenium, Se	0.019 ± 0.466		0.001 ± 0.025
Bromine, Br	0.952 ± 0.343	0.109 ± 0.011	0.061 ± 0.01
Rubidium, Rh	0.19 ± 0.238	0.089 ± 0.008	0.038 ± 0.007
Strontium, Sr	0.41 ± 0.44	1.054 ± 0.016	0.654 ± 0.014
Yttrium, Y		0.029 ± 0.011	
Zirconium, Zr	0.038 ± 0.77	0.112 ± 0.025	0.083 ± 0.024
Niobium, Nb	0.162 ± 0.591		
Molybdenum, Mo			
Palladium, Pd	0.343 ± 1.01		
Silver, Ag	0.01 ± 0.97	0.01 ± 0.031	
Cadmium, Cd			
Indium, In		0.004 ± 0.021	
Tin, Sn			
Antimony, Sb			
Cesium, Cs			
Barium, Ba			
Lanthanum, La			
Cerium, Ce	0.048 ± 0.288		
Samarium, Sa			
Europium, Eu		0.018 ± 0.048	
Terbium, Tb			
Hafnium, Hf	0.467 ± 3.155		
Tantalum, Ta			
Tungsten, W			
Iridium, Ir			
Gold, Au	0.438 ± 1.743		
Mercury, Hg			
Thallium, Th			
Lead, Pb	0.886 ± 0.586	0.063 ± 0.019	0.035 ± 0.018
Uranium, U	0.038 ± 0.941	0.012 ± 0.03	0.008 ± 0.028

Table B-2. Emission profiles of PM_{10} components emitted from Unburned and Burned soils at 3 MAB.

	Unburned	Burned Ridge	Burned Drainage
Chloride, Cl^-		0.302 ± 0.837	1.764 ± 0.589
Nitrate, NO_3	3.792 ± 2.138	1.581 ± 0.844	0.448 ± 0.361
Sulfate, SO^{2-}	6.181 ± 2.16	3.817 ± 0.88	3.916 ± 0.469
Ammonium, NH_4^+	0.292 ± 2.09	0.427 ± 0.826	0.23 ± 0.358
Sodium, Na^+	3.334 ± 1.938	1.623 ± 0.775	0.622 ± 0.317
Magnesium, Mg^{2+}	3.383 ± 0.271	2.434 ± 0.178	2.921 ± 0.21
Potassium, K^+	3.888 ± 1.944	2.248 ± 0.783	1.572 ± 0.334
Calcium, Ca^{2+}	112.912 ± 9.108	123.838 ± 9.191	113.33 ± 8.281
Total OC	105.626 ± 56.936	99.829 ± 47.635	70.413 ± 32.98
Total EC	31.665 ± 5.105	64.269 ± 6.884	101.447 ± 10.532
Carbonate, CO_3	5.161 ± 18.106	32.982 ± 10.424	86.01 ± 20.035
Total Carbon	142.452 ± 38.131	197.08 ± 41.982	257.871 ± 53.06
Sodium, Na		1.689 ± 2.321	5.698 ± 1.171
Magnesium, Mg		3.281 ± 1.235	3.051 ± 0.577
Aluminum, Al	3.495 ± 0.592	12.541 ± 0.948	6.724 ± 0.495
Silicon, Si	12.208 ± 1.154	37.076 ± 2.727	18.194 ± 1.316
Phosphorous, P			1.892 ± 0.149
Sulfur, S	0.938 ± 0.465	0.851 ± 0.192	0.238 ± 0.081
Chlorine, Cl	0.023 ± 0.113	0.147 ± 0.047	0.082 ± 0.021
Potassium, K	4.955 ± 0.408	13.771 ± 1.002	10.79 ± 0.772
Calcium, Ca	24.368 ± 1.969	104.305 ± 7.577	180.01 ± 12.869
Scandium, Sc		0.023 ± 0.164	
Titanium, Ti	1.034 ± 0.113	2.209 ± 0.164	1.145 ± 0.083
Vanadium, V	0.026 ± 0.007	0.035 ± 0.004	0.011 ± 0.001
Chromium, Cr	0.038 ± 0.069		
Manganese, Mn		0.664 ± 0.077	0.727 ± 0.058
Iron, Fe	16.782 ± 1.344	22.941 ± 1.67	12.393 ± 0.888
Cobalt, Co			
Nickel, Ni		0.009 ± 0.014	0.009 ± 0.006
Copper, Cu	0.096 ± 0.062	0.043 ± 0.024	0.03 ± 0.01
Zinc, Zn	0.264 ± 0.076	0.232 ± 0.033	0.104 ± 0.014
Gallium, Ga			
Arsenic, As			

Table B-2. Emission profiles of PM_{10} components emitted from Unburned and Burned soils at 3 MAB (continued).

	Unburned	Burned Ridge	Burned Drainage
Selenium, Se			
Bromine, Br	0.101 ± 0.103	0.113 ± 0.042	
Rubidium, Rh	0.036 ± 0.075	0.061 ± 0.03	0.059 ± 0.013
Strontium, Sr	0.219 ± 0.137	0.452 ± 0.064	0.833 ± 0.064
Yttrium, Y	0.219 ± 0.103	0.02 ± 0.04	
Zirconium, Zr		0.103 ± 0.095	0.031 ± 0.041
Niobium, Nb			0.002 ± 0.027
Molybdenum, Mo	0.11 ± 0.163		
Palladium, Pd		0.039 ± 0.123	
Silver, Ag			
Cadmium, Cd			
Indium, In			0.004 ± 0.039
Tin, Sn			
Antimony, Sb	0.445 ± 0.546		
Cesium, Cs			
Barium, Ba			
Lanthanum, La			
Cerium, Ce			
Samarium, Sa			0.009 ± 0.021
Europium, Eu		0.091 ± 0.176	
Terbium, Tb			
Hafnium, Hf	0.275 ± 0.975	0.19 ± 0.387	0.09 ± 0.168
Tantalum, Ta	0.224 ± 0.819		0.051 ± 0.14
Tungsten, W	0.323 ± 1.192	0.168 ± 0.472	0.05 ± 0.203
Iridium, Ir	0.089 ± 0.252		
Gold, Au		0.016 ± 0.215	
Mercury, Hg			
Thallium, Th	0.048 ± 0.172	0.056 ± 0.067	
Lead, Pb			0.037 ± 0.031
Uranium, U			0.043 ± 0.05

Table B-3. Emission profiles of PM_{10} components emitted from Unburned and Burned soils at 6 MAB.

	Unburned	**Burned Ridge**	**Burned Drainage**
Chloride, Cl^-		0.168 ± 0.805	4.162 ± 1.047
Nitrate, NO_3	29.146 ± 19.887	1.13 ± 0.807	0.527 ± 0.222
Sulfate, SO^{2-}	181.687 ± 53.936	37.479 ± 2.987	9.467 ± 0.745
Ammonium, NH_4^+	2.913 ± 17.999	0.209 ± 0.795	0.174 ± 0.214
Sodium, Na^+	153.965 ± 46.298	26.92 ± 2.179	4.881 ± 0.417
Magnesium, Mg^{2+}	6.254 ± 1.76	3.445 ± 0.252	2.385 ± 0.17
Potassium, K^+	31.731 ± 19.276	2.503 ± 0.759	1.517 ± 0.229
Calcium, Ca^{2+}	129.967 ± 36.398	160.068 ± 11.862	87.166 ± 6.329
Total OC	227.021 ± 213.752	111.648 ± 52.791	61.246 ± 27.956
Total EC	86.024 ± 40.483	154.192 ± 16.155	111.527 ± 11.521
Carbonate, CO_3	31.752 ± 155.925	51.48 ± 13.721	47.2 ± 11.014
Total Carbon	344.797 ± 210.225	317.319 ± 66.075	219.974 ± 44.879
Sodium, Na		2.373 ± 2.258	2.845 ± 0.68
Magnesium, Mg		1.748 ± 1.168	2.603 ± 0.376
Aluminum, Al	8.576 ± 5.019	10.602 ± 0.807	7.811 ± 0.563
Silicon, Si	58.105 ± 17.005	35.522 ± 2.61	22.26 ± 1.593
Phosphorous, P		0.119 ± 0.128	0.889 ± 0.073
Sulfur, S	46.594 ± 13.592	6.635 ± 0.521	1.24 ± 0.102
Chlorine, Cl	1.062 ± 1.027	0.652 ± 0.066	0.269 ± 0.023
Potassium, K	25.203 ± 7.082	13.944 ± 1.013	11.179 ± 0.795
Calcium, Ca	43.899 ± 12.689	121.103 ± 8.787	157.535 ± 11.192
Scandium, Sc			
Titanium, Ti	4.195 ± 1.334	1.839 ± 0.138	1.402 ± 0.1
Vanadium, V		0.033 ± 0.004	0.031 ± 0.002
Chromium, Cr			
Manganese, Mn		0.904 ± 0.088	0.958 ± 0.07
Iron, Fe	42.048 ± 11.827	20.301 ± 1.477	15.148 ± 1.077
Cobalt, Co			
Nickel, Ni	0.077 ± 0.298		
Copper, Cu		0.019 ± 0.023	0.043 ± 0.007
Zinc, Zn		0.292 ± 0.035	0.157 ± 0.014
Gallium, Ga			
Arsenic, As			

Table B-3. Emission profiles of PM_{10} components emitted from Unburned and Burned soils at 6 MAB (continued).

	Unburned	**Burned Ridge**	**Burned Drainage**
Gallium, Ga			
Arsenic, As			
Selenium, Se	0.019 ± 0.466		0.001 ± 0.025
Bromine, Br	0.952 ± 0.343	0.109 ± 0.011	0.061 ± 0.01
Rubidium, Rh	0.19 ± 0.238	0.089 ± 0.008	0.038 ± 0.007
Strontium, Sr	0.41 ± 0.44	1.054 ± 0.016	0.654 ± 0.014
Yttrium, Y		0.029 ± 0.011	
Zirconium, Zr	0.038 ± 0.77	0.112 ± 0.025	0.083 ± 0.024
Niobium, Nb	0.162 ± 0.591		
Molybdenum, Mo			
Palladium, Pd	0.343 ± 1.01		
Silver, Ag	0.01 ± 0.97	0.01 ± 0.031	
Cadmium, Cd			
Indium, In		0.004 ± 0.021	
Tin, Sn			
Antimony, Sb			
Cesium, Cs			
Barium, Ba			
Lanthanum, La			
Cerium, Ce	0.048 ± 0.288		
Samarium, Sa			
Europium, Eu		0.018 ± 0.048	
Terbium, Tb			
Hafnium, Hf	0.467 ± 3.155		
Tantalum, Ta			
Tungsten, W			
Iridium, Ir			
Gold, Au	0.438 ± 1.743		
Mercury, Hg			
Thallium, Th			
Lead, Pb	0.886 ± 0.586	0.063 ± 0.019	0.035 ± 0.018
Uranium, U	0.038 ± 0.941	0.012 ± 0.03	0.008 ± 0.028

Table B-4. Emission profiles of PM_{10} components emitted from Unburned and Burned soils at 13 MAB.

	Unburned	**Burned Ridge**	**Burned Drainage**
Chloride,Cl-			
Nitrate, NO3			0.161 ± 0.147
Sulfate, SO 2-	2.143 ± 16.838		2.068 ± 0.149
Ammonium,NH4+	16 ± 17.09	9.742 ± 7.659	0.491 ± 0.147
Sodium, Na+	7 ± 1.568	8.613 ± 0.822	0.589 ± 0.016
Magnesium, Mg2+	3.286 ± 0.863	3.097 ± 0.389	1.705 ± 0.029
Potassium, K+	7.143 ± 2.068	6.677 ± 0.923	2.192 ± 0.057
Calcium, Ca2+	75.643 ± 17.72	32.355 ± 6	54.624 ± 5.58
Total OC	136.714 ± 193.347	104.14 ± 98.035	36.673 ± 13.715
Total EC	17.643 ± 31.708	15.785 ± 15.165	25.537 ± 7.833
Carbonate, CO3			11.751 ± 3.304
Total Carbon -	409.881 ± 220.945	235.333 ± 110.398	78.44 ± 28.921
Sodium, Na			
Magnesium, Mg			
Aluminum, Al	58.238 ± 12.122	5.796 ± 3.234	18.202 ± 0.146
Silicon, Si	177.738 ± 31.421	19.538 ± 4.78	86.095 ± 0.467
Phosphorous, P			1.191 ± 0.073
Sulfur, S	4.976 ± 8.35		0.7 ± 0.074
Chlorine, Cl			0.033 ± 0.041
Potassium, K	23.143 ± 5.036	2.968 ± 1.469	12.7 ± 0.046
Calcium, Ca	121.571 ± 21.891	8.667 ± 3.714	100.154 ± 0.383
Scandium, Sc			
Titanium, Ti	3.214 ± 1.638	0.065 ± 0.704	2.009 ± 0.017
Vanadium, V	0.286 ± 0.693	0.108 ± 0.313	0.026 ± 0.006
Chromium, Cr		0.011 ± 0.286	0.017 ± 0.005
Manganese, Mn		0.237 ± 3.178	0.867 ± 0.063
Iron, Fe	40.643 ± 6.904	3.71 ± 0.644	22.505 ± 0.066
Cobalt, Co	0.143 ± 0.692	0.108 ± 0.313	
Nickel, Ni	0.286 ± 3.242	0.161 ± 1.46	0.014 ± 0.028
Copper, Cu			0.029 ± 0.068
Zinc, Zn	0.024 ± 7.848	0.333 ± 3.513	0.108 ± 0.068
Gallium, Ga			0.007 ± 0.065
Arsenic, As			0.003 ± 0.065

Table B-4. Emission profiles of PM_{10} components emitted from Unburned and Burned soils at 13 MAB (continued).

	Unburned	**Burned Ridge**	**Burned Drainage**
Selenium, Se			
Bromine, Br	0.333 ± 1.333	0.043 ± 0.602	0.083 ± 0.012
Rubidium, Rh			0.085 ± 0.008
Strontium, Sr	0.143 ± 1.311		0.43 ± 0.012
Yttrium, Y	0.238 ± 0.905	0.065 ± 0.412	0.012 ± 0.008
Zirconium, Zr	0.048 ± 3.504	0.032 ± 1.557	0.111 ± 0.031
Niobium, Nb			
Molybdenum, Mo	0.905 ± 2.268	0.011 ± 1.045	0.003 ± 0.022
Palladium, Pd	0.762 ± 2.86	0.667 ± 1.292	
Silver, Ag	1.595 ± 3.177	0.785 ± 1.431	
Cadmium, Cd	0.5 ± 5.048	0.215 ± 2.279	
Indium, In	0.071 ± 4.023	0.409 ± 1.83	
Tin, Sn		0.172 ± 2.655	
Antimony, Sb	3.238 ± 7.685	0.774 ± 3.462	0.015 ± 0.068
Cesium, Cs			
Barium, Ba	2.095 ± 12.17	5.183 ± 5.541	
Lanthanum, La	6.857 ± 21.221		
Cerium, Ce			
Samarium, Sa			0.074 ± 0.298
Europium, Eu	5.69 ± 28.418		
Terbium, Tb		2.774 ± 13.022	0.152 ± 0.254
Hafnium, Hf			0.016 ± 0.117
Tantalum, Ta	4.881 ± 9.843	0.28 ± 4.415	
Tungsten, W			
Iridium, Ir		0.065 ± 1.311	
Gold, Au			
Mercury, Hg			
Thallium, Th	0.333 ± 1.975	0.14 ± 0.894	0.001 ± 0.014
Lead, Pb	0.143 ± 2.097		0.029 ± 0.019
Uranium, U			

APPENDIX C: PI-SWERL EMISSION PROFILES

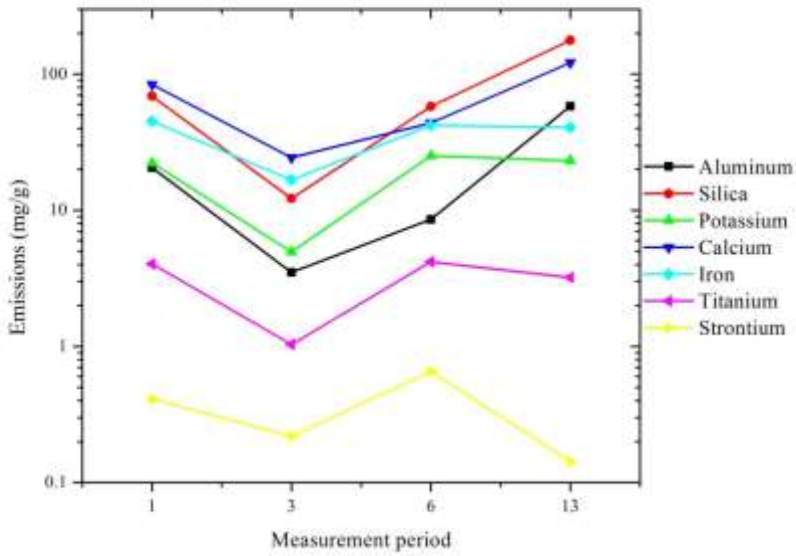


Figure C-1. Emission profiles (in µg/g) of elements associated with mineral soil particles from Control (unburned) soils. (Uncertainties overlap zero for: strontium at 1, 6, and 13 MAB).

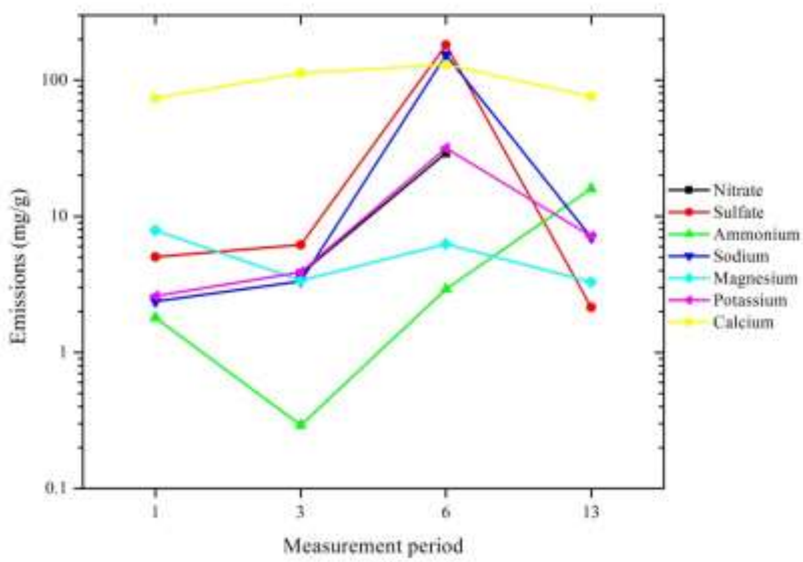


Figure C-2. Emission profiles (in µg/g) of water-soluble anions and cations from Control soils. (Uncertainties overlap zero for: ammonium at 1, 3, 6, and 13 MAB; sulfate at 1 and 3 MAB).

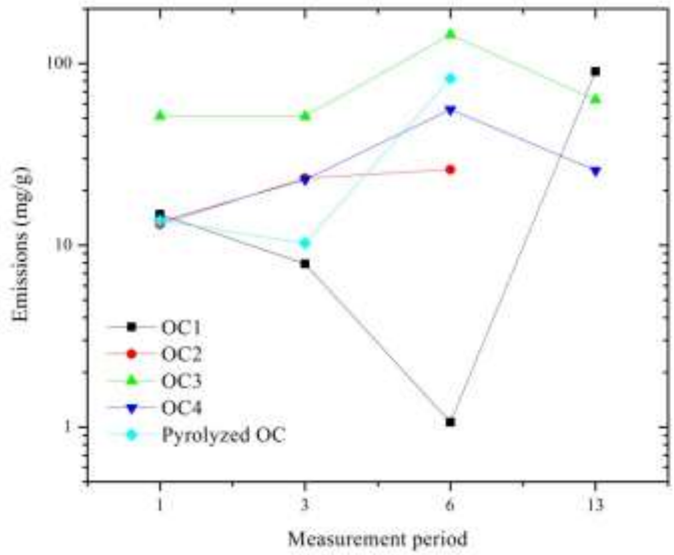


Figure C-3. Emission profiles (in µg/g) of carbonaceous particles from Control soils.

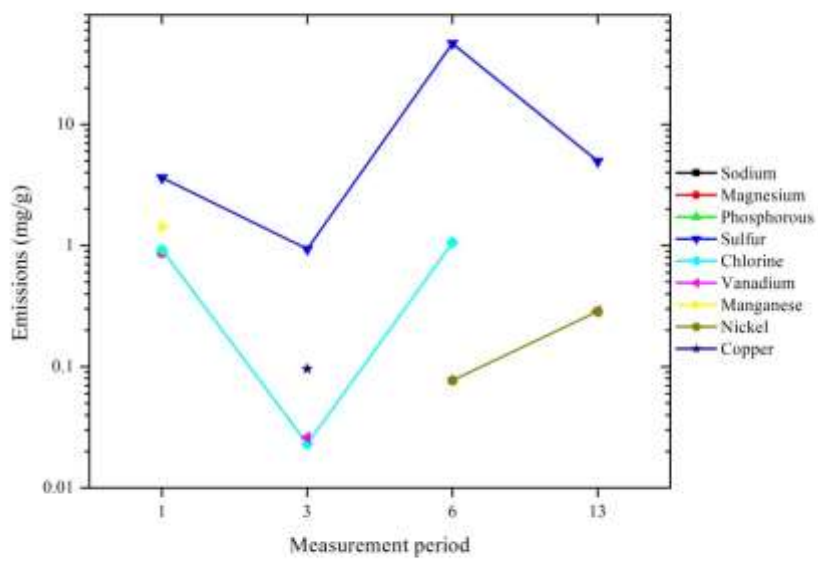


Figure C-4. Emission profiles (in µg/g) of common elements from Control soils. (Uncertainties overlap zero for: magnesium at 1 MAB; sulfur at 13 MAB, vanadium at 13 MAB; nickel at 6 and 13 MAB).

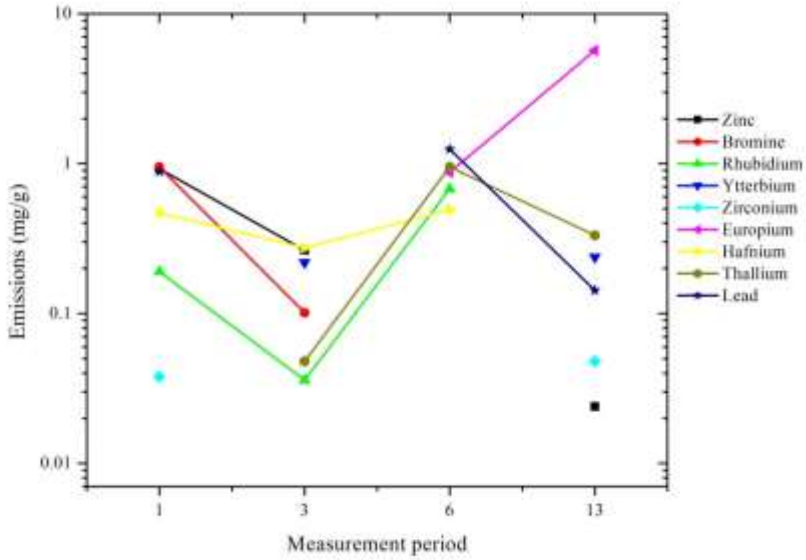


Figure C-5. Emission profiles (in µg/g) of rare elements from Control soils. (Uncertainties overlap zero for: zinc at 13 MAB; ytterbium at 13 MAB; bromine at 3 and 13 MAB; rhubidium at 1 and 13 MAB; zirconium at 1 and 13 MAB; europium at 6 and 13 MAB; thallium at 6 and 13 MAB; halfnium at 1, 3, and 6 MAB).

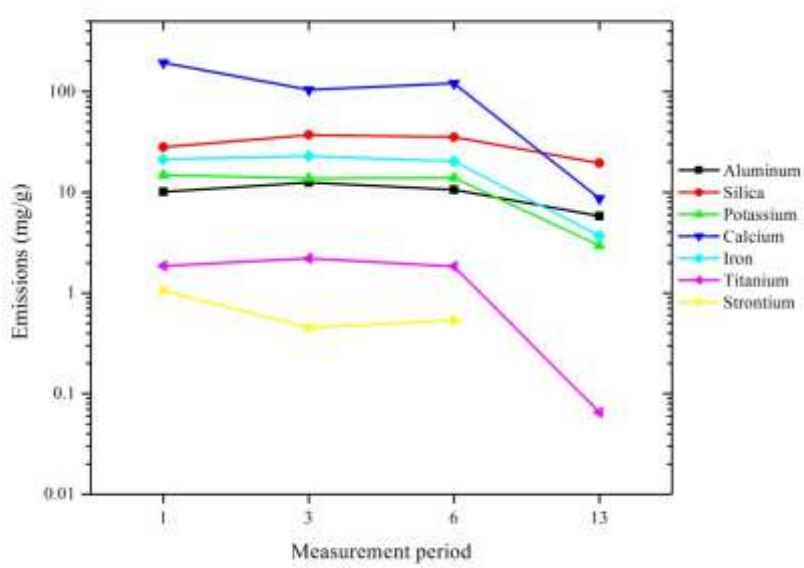


Figure C-6. Emission profiles (in µg/g) of elements associated with mineral soil particles from Burned Ridge soils. (Uncertainties overlap zero for: titanium at 13 MAB).

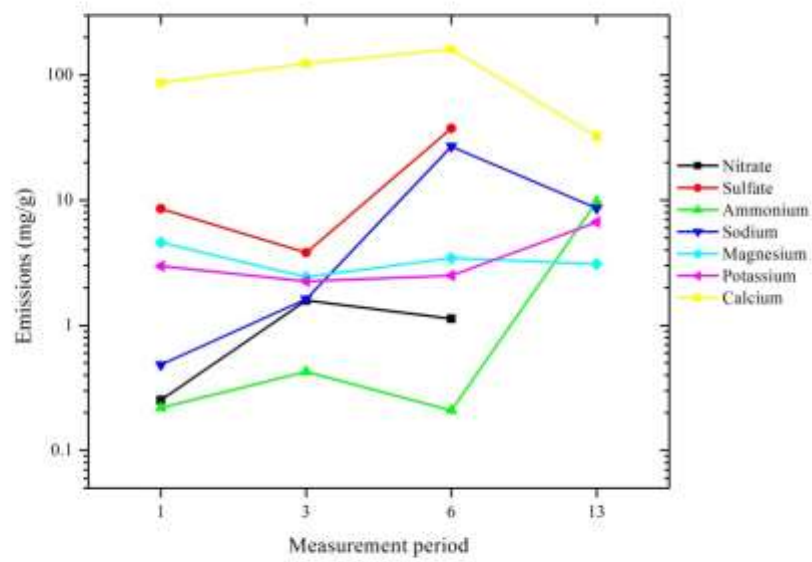


Figure C-7. Emission profiles (in µg/g) of water-soluble anions and cations from Burned Ridge soils. (Uncertainties overlap zero for: ammonium at 1, 3, and 6 MAB).

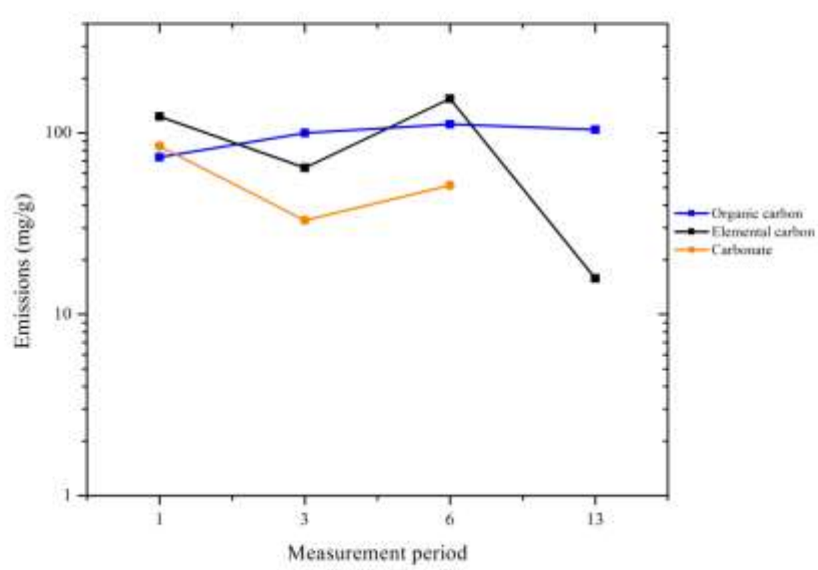


Figure C-8. Emission profiles (in µg/g) of carbonaceous particles from Burned Ridge soils.

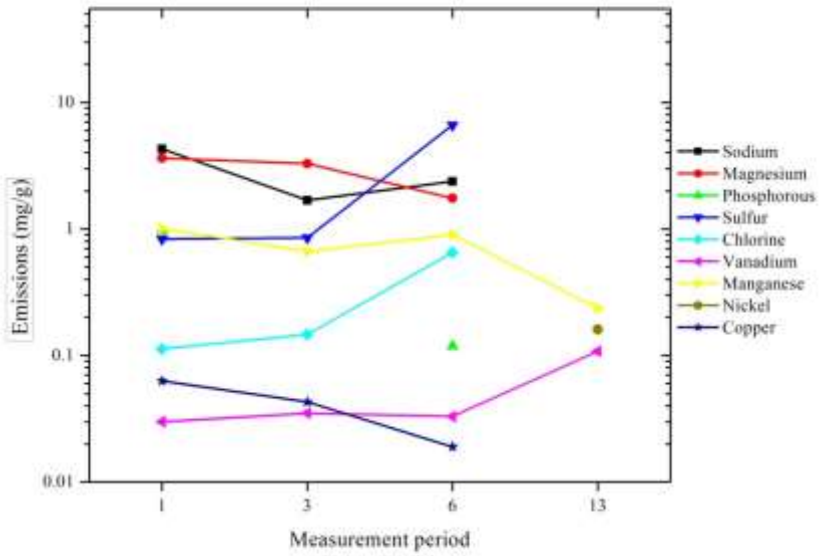


Figure C-9. Emission profiles (in µg/g) of common elements from Burned Ridge soils. (Uncertainties overlap zero for: sodium at 3 MAB; phosphorous at 6 MAB; copper at 6 MAB; vanadium at 13 MAB; manganese at 13 MAB; nickel at 3 and 13 MAB).

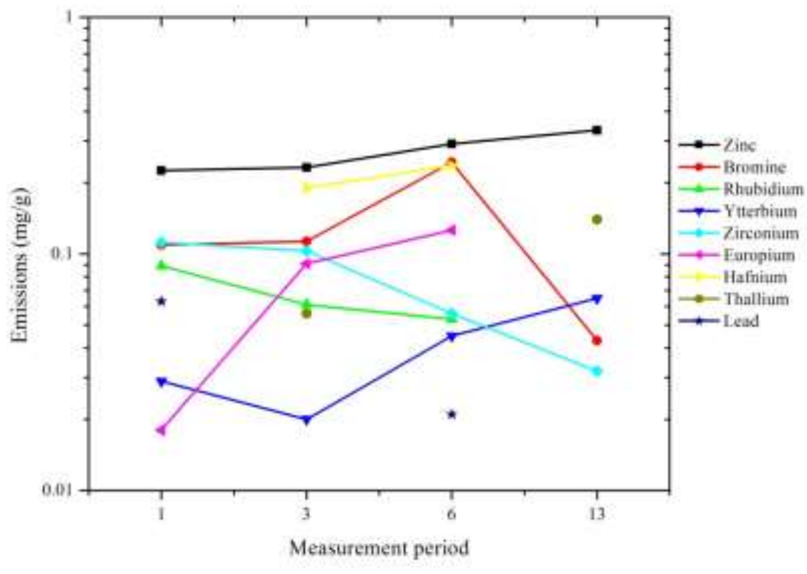


Figure C-10. Emission profiles (in µg/g) of rare elements from Burned Ridge soils. (Uncertainties overlap zero for: lead at 6 MAB; ytterbium at 3 and 13 MAB; thallium at 3 and 13 MAB; zirconium at 6 and 13 MAB; halfnium at 3 and 6 MAB; and europium at 1, 3, and 6 MAB).

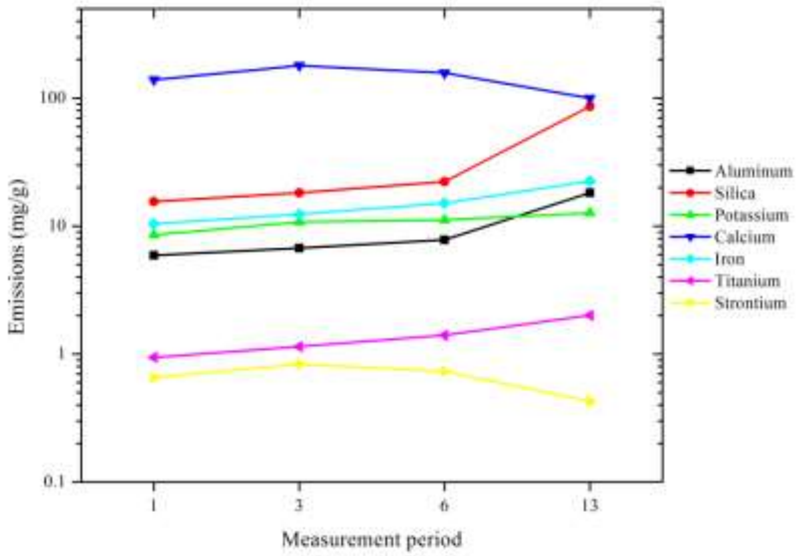


Figure C-11. Emission profiles (in µg/g) of elements associated with mineral soil particles from Burned Drainage soils.

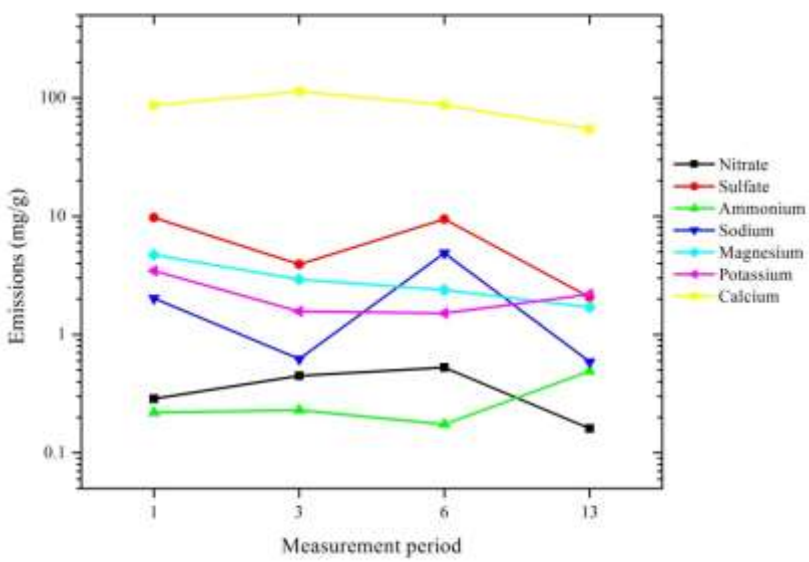


Figure C-12. Emission profiles (in µg/g) of water-soluble anions and cations from Burned Drainage soils. (Uncertainties overlap zero for: ammonium at 3 and 6 MAB).

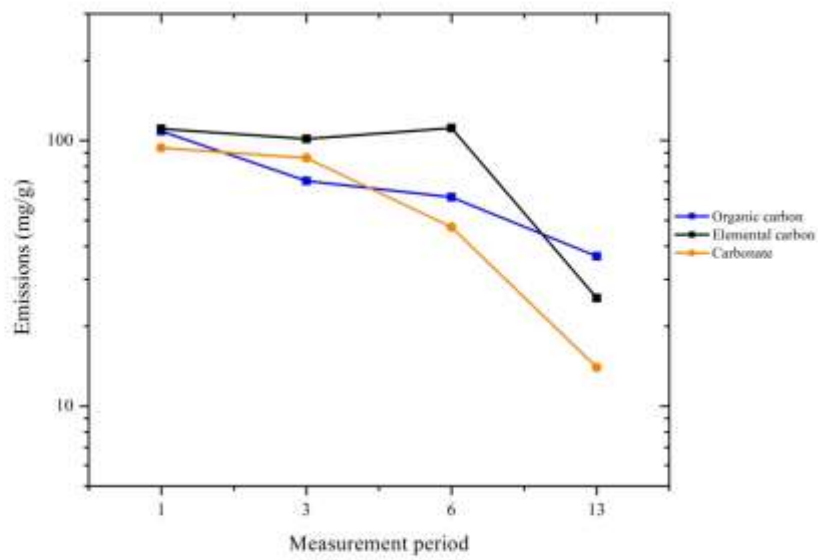


Figure C-13. Emission profiles (in μg/g) of carbonaceous particles from Burned Drainage soils.

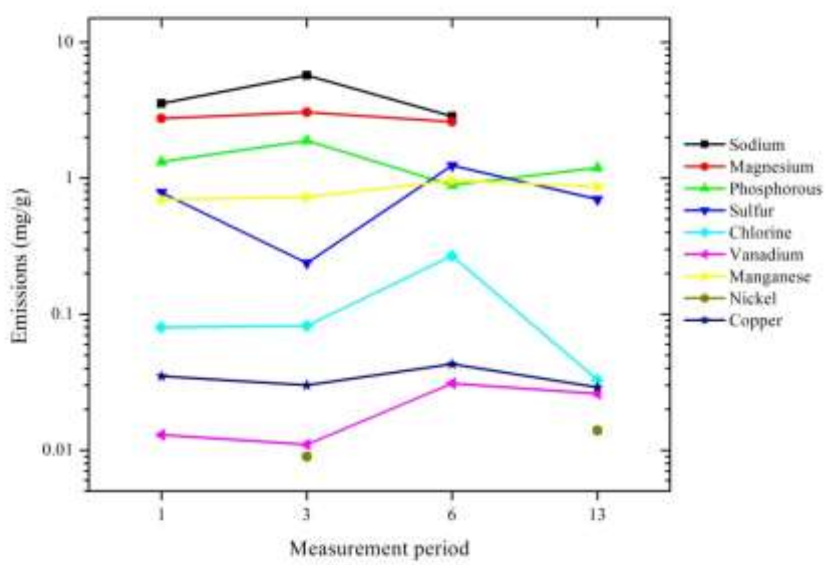


Figure C-14. Emission profiles (in μg/g) of common elements from Burned Drainage soils. (Uncertainties overlap zero for: chloride at 13 MAB; nickel at 13 MAB; and copper at 13 MAB).

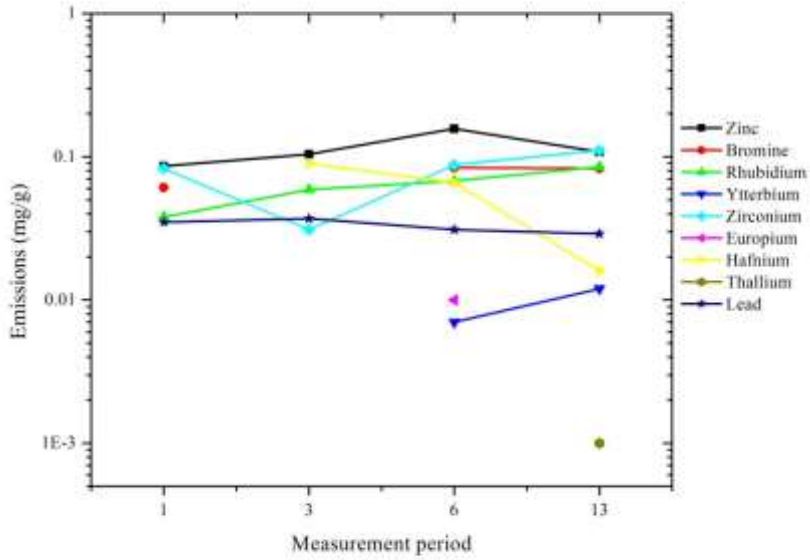


Figure C-15. Emission profiles (in μg/g) of rare elements from Burned Drainage soils. (Uncertainties overlap zero for: zirconium at 3 MAB; ytterbium at 6 MAB; europium at 6 MAB; thallium at 3 and 13 MAB; and halfnium at 3, 6, and 13 MAB).

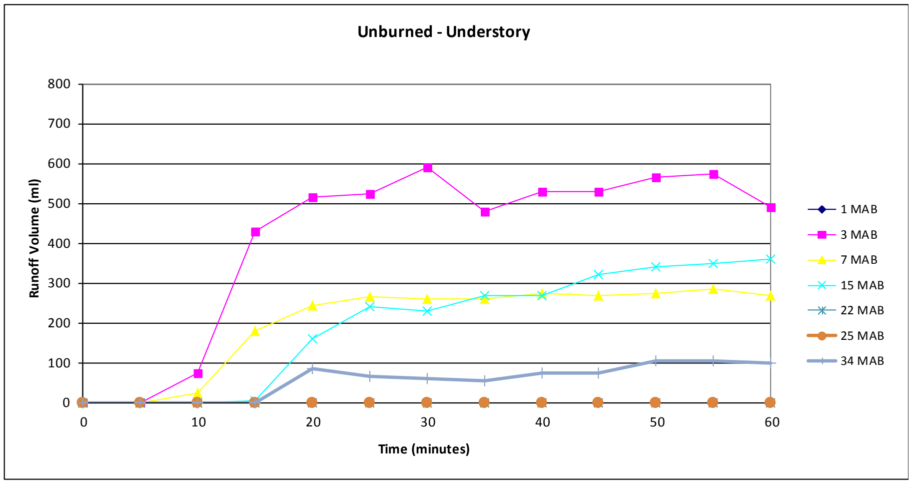
APPENDIX D: SOIL MOISTURE DATA FROM JACOB FIRE SITE (2008 – 2011)

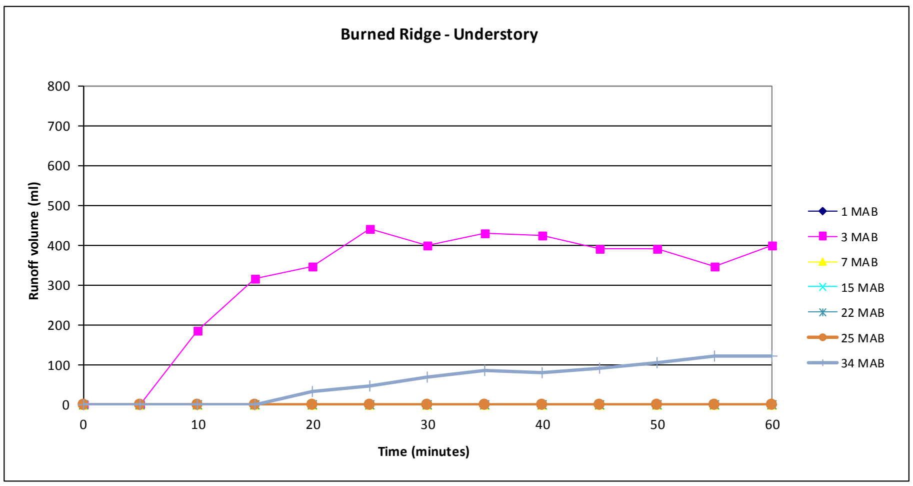
Site	Date (MAB)	Gravimetric Moisture Content (g/g)	
		Initial	Final
Unburned, understory	1	0.02	0.14
	3	0.05	0.12
	7	--	--
	15	0.02	0.13
	22	0.02	0.15
	25	0.02	0.19
	34	0.02	0.19
Unburned, interspace	1	0.01	0.08
	3	0.05	0.13
	7	--	--
	15	0.01	0.11
	22	0.03	0.08
	25	0.02	0.12
	34	0.02	0.10
Burned ridge, understory	1	0.02	0.19
	3	0.02	0.18
	7	--	--
	15	0.02	0.02
	22	0.02	0.13
	25	0.01	0.17
	34	0.06	0.19
Burned ridge, interspace	1	0.02	0.09
	3	0.07	0.13
	7	--	--
	15	0.01	0.12
	22	0.03	0.07
	25	0.03	0.04
	34	0.03	0.13
Burned drainage, understory	1	0.01	0.14
	3	0.04	0.14
	7	--	--
	15	0.03	0.15
	22	0.01	0.15
	25	0.01	0.14
	34	0.01	0.16
Burned drainage, interspace	1	0.01	0.10
	3	0.05	0.13
	7	--	--
	15	0.01	0.11
	22	0.01	0.15
	25	0.01	0.14
	34	0.01	0.16

APPENDIX E: SOIL BULK DENSITY AND POROSITY DATA FROM JACOB FIRE SITE (2010 – 2011)

Date	Site	Bulk Density	Porosity
22 MAB	Control, US	1.49	0.44
	Control, IS	1.60	0.40
	Burned Ridge, US	1.50	0.43
	Burned Ridge, IS	1.48	0.44
	Burned Drainage, US	1.47	0.45
	Burned Drainage, IS	1.66	0.37
25 MAB	Control, US	1.52	0.42
	Control, IS	1.52	0.38
	Burned Ridge, US	1.71	0.35
	Burned Ridge, IS	1.57	0.41
	Burned Drainage, US	1.59	0.40
	Burned Drainage, IS	1.63	0.38
34 MAB	Control, US	1.57	0.41
	Control, IS	1.71	0.36
	Burned Ridge, US	1.60	0.40
	Burned Ridge, IS	1.82	0.31
	Burned Drainage, US	1.41	0.47
	Burned Drainage, IS	1.74	0.34

APPENDIX F: RUNOFF HYDROGRAPHS FROM JACOB FIRE SITE (2008 – 2011)





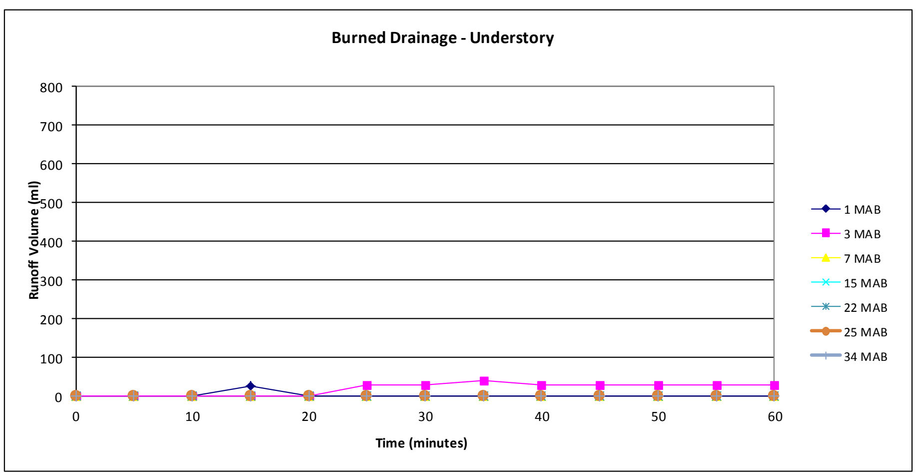
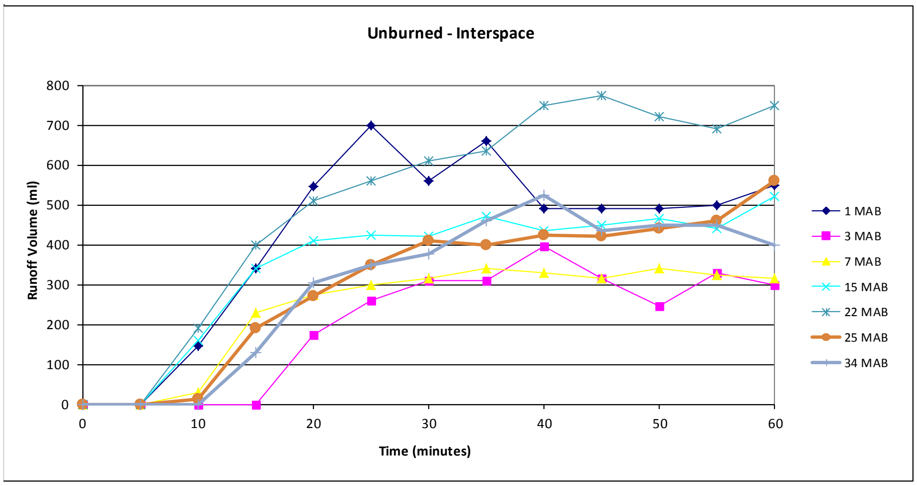
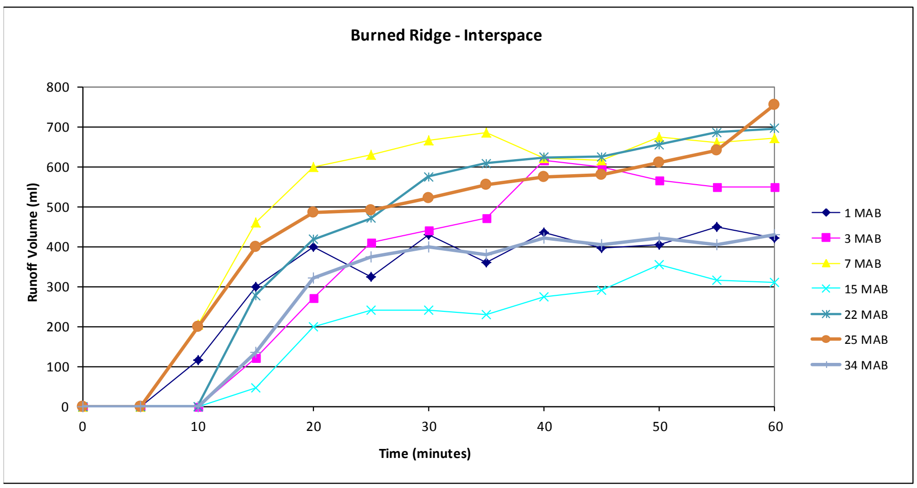


Figure F-1. Runoff Hydrographs From Jacob Fire Site (2008 – 2011)





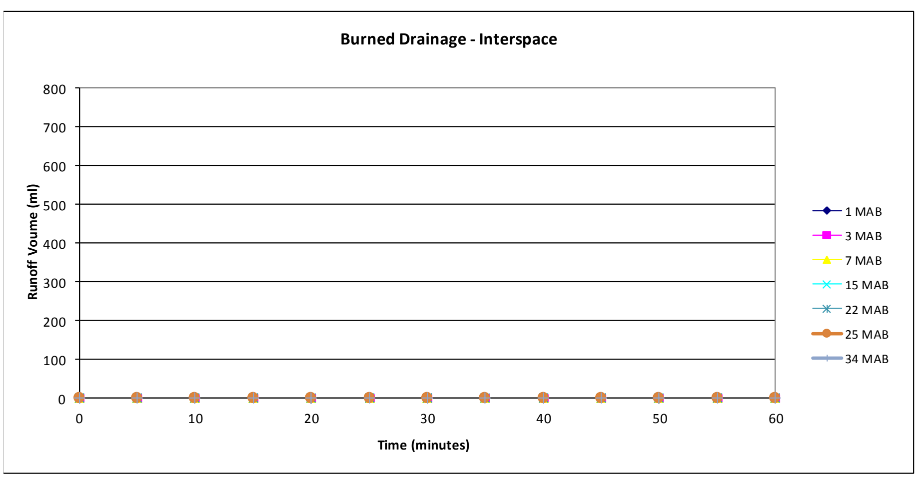


Figure F-1. Runoff Hydrographs From Jacob Fire Site (2008 – 2011) (continued).

APPENDIX G: CURVE NUMBERS (CNS) CALCULATED FOR THE JACOB FIRE SITE (2008 – 2010)

Table G-1. Curve Numbers (CNS) Calculated for the Jacob Fire Site (2008 – 2010)

		Curve Numbers for Rainfall Simulation Tests			
Surface	**MAB**	**Ponding**	**Initial Runoff**	**All Quadrant Runoff**	**Initial Runoff to Trough**
Burned Ridge, Understory	1	92	92	No Runoff	No Runoff
	3	96	96	93	93
	7	94	90	84	No Runoff
	15	97	96	92	84
	22	No Ponding	No Runoff	No Runoff	No Runoff
	25	96	96	No Runoff	No Runoff
	34	98	96	94	90
Burned Ridge, Interspace	1	93	93	88	94
	3	98	95	89	87
	7	95	94	93	92
	15	88	86	84	86
	22	92	91	88	87
	25	97	96	95	93
	34	94	94	92	88
Burned Drainage, Understory	1	98	97	96	92
	3	94	93	90	89
	7	No Ponding	No Runoff	No Runoff	No Runoff
	15	98	97	94	91
	22	No Ponding	No Runoff	No Runoff	No Runoff
	25	96	89	No Runoff	80
	34	No Ponding	No Runoff	No Runoff	No Runoff
Burned Drainage, Interspace	1	No Ponding	No Runoff	No Runoff	No Runoff
	3	No Ponding	No Runoff	No Runoff	No Runoff

Table G-1. Curve Numbers (CNS) Calculated for the Jacob Fire Site (2008 – 2010) (continued).

Surface	**MAB**	**Curve Numbers for Rainfall Simulation Tests**			
		Ponding	**Initial Runoff**	**All Quadrant Runoff**	**Initial Runoff to Trough**
	7	No Ponding	No Runoff	No Runoff	No Runoff
	15	No Ponding	No Runoff	No Runoff	No Runoff
	22	No Ponding	No Runoff	No Runoff	No Runoff
	25	92	89	No Runoff	No Runoff
	34	No Ponding	No Runoff	No Runoff	No Runoff
Unburned, Understory	1	72	No Runoff	No Runoff	No Runoff
	3	94	92	92	92
	7	96	93	91	90
	15	93	84	84	83
	22	90	66	No Runoff	59
	25	97	95	90	66
	34	93	89	No Runoff	88
Unburned, Interspace	1	96	95	93	93
	3	92	91	87	86
	7	95	92	91	90
	15	99	98	95	93
	22	99	97	94	93
	25	96	94	92	90
	34	96	89	87	88

APPENDIX H: ANALYSIS OF VEGETATION AT THE JACOB FIRE SITE (2008 - 2011)

INTRODUCTION

The increase in fires in arid and semi-arid parts of Nevada and elsewhere in the Southwest U.S. has implications for post-closure management or long-term stewardship for Soil Sub-Project Corrective Action Units (CAUs) for which the Department of Energy, Nevada Nuclear Security Administration Nevada Site office (NNSA/NSO) has responsibility for regulatory closure. For many CAUs and Corrective Action Sites (CASs) where closure-in-place alternatives are now being implemented or considered, there is a greater chance that they could experience wildfire at some point while they still pose a radiological risk, especially considering the long half-life of some of the radionuclide contaminants of concern (COCs) (Shafer et al., 2007; Shafer and Gomes, 2009).

In the 1970's the Nevada Applied Ecology Group (NAEG) examined a variety of disturbance mechanisms at Soil CAUs where isotopes of plutonium were among the most significant COCs, however wildfires were not considered because at that time they were comparatively infrequent and small in size (Bruce Church and Lynn Anspaugh, *personal communication*, 2007). The majority of NAEG studies were carried out before the spread of cheatgrass (*Bromus tectorum*) and red brome (*Bromus madritensis*) across the area encompassing the Nevada National Security Site (NNSS), the Tonopah Test Range (TTR), and the Nevada Test and Training Range (NTTR). Both of these grass species were introduced to the region as part of Euro-American settlement and cattle grazing in the nineteenth century (Knapp, 1996). Found locally on disturbed sites as early as the 1960s on the NNSS, both species had spread across the region during a period of above-average precipitation in the 1980s (Rickard and Beatley, 1965; Hunter, 1991). Today, these grasses can rapidly invade disturbed areas and in many plant communities they have colonized interspaces between shrubs, increasing the total fuel load and allowing fires to move more easily between shrubs (Knapp, 1996). Besides increasing the chance of a fire occurring and the likelihood that it will burn larger areas than historic fires, Hansen and Ostler (2004) have documented that invasive plants are quick to colonize areas that had previously burned on the NNSS, increasing the chance that subsequent fires will occur.

Most wildland fires on the NNSS have occurred in the blackbrush zone (Hansen and Ostler, 2004). Blackbrush (*Coleogyne ramosissima*) is a native rosaceous Mojave Desert perennial shrub that occurs at the transition between the Mojave and Great Basin deserts of the southwestern United States. Blackbrush communities are one of the dominant vegetation community types in southern Nevada, occurring between 1220 m (4003 ft) and 2000 m (6562 ft) elevation above the creosote bush-bursage communities on the lower mountain slopes and valley floors, and below the pinyon-juniper-sagebrush communities that occur higher on the mountain slopes. It has a relatively low growth rate limited by soil moisture associated with the shallow, gravelly soils on which it tends to associate (Brooks, 2005; Lei and Walker, 1995). Plant diversity in blackbrush communities is comparable to other major vegetation types although invasion by non-native species is increasingly reported (Brooks, 2005).

Over the past 30 years, there has been a significant increase in the frequency, extent and intensity of wildfires in native desert shrub communities in the southwestern United States (Knapp, 1998; Brooks and Matchett 2006; Abella, 2010). In particular, blackbrush communities have become increasingly susceptible to wildfire due to drought and invasion by non-native annual grasses (Abella, 2009; Rew et al., 2010). Fire has long-term effects on

Figure 1. Unburned (top) and burned (bottom) blackbrush community, Lincoln County, NV.

these communities which experience little regeneration (Callison *et al*. 1985; Brooks and Matchett 2003; Abella *et al*., 2009). Unlike other shrub species, blackbrush does not sprout from roots after fire and experiences slow recruitment rates (Humphrey, 1974; Brooks, 1995). Figure 1 shows an intact blackbrush community and a similar landscape post-fire. Because it is considered one of the most flammable native plant assemblages in the Mojave Desert, it is considered to be a hazardous fuel (Brooks, 2005).

Other non-fire disturbances to blackbrush communities also tend to result in little or no natural recruitment of blackbrush (Webb *et al*., 1987; Brooks, 1995). The lack of regeneration is often due in part to the rapid post-fire establishment of non-native annual grasses that are capable of perpetuating a grass/fire cycle which eliminates initial blackbrush regeneration that may have occurred after the first fire cycle (D'Antonio and Vitosek, 1992).

High intensity wildfires have the ability to kill most of the native soil seed bank including blackbrush (Esque *et al*., 2010). Finally, changes to the soil chemistry and physical and hydrological properties due to fire may also play a role in the minimal post-fire re-establishment of blackbrush (Lei, 1999).

To better understand the implications and risks should Soil Sub-Project CAUs/CASs experience wildfires, a series of studies have been initiated at uncontaminated analog sites to better understand the possible impacts of erosion and transport by wind and water, the risks and perceived risk they might pose to site workers and public receptors in communities around the NNSS, TTR, and NTTR; and to develop recommendations for stabilization and restoration of contaminated sites should they burn. The first of these studies was undertaken at the Jacob fire site located in Lincoln County, NV, 170 kilometers (106 miles) north of Las Vegas and 100 km (63 miles) northeast of Yucca Valley, NNSS (Figure 2). This location is representative of the transition zone between the Mojave and the Great Basin Desert on the NNSS (Ostler et al., 2000) where the largest number of Soil CAUs and CASs are Located.

The Jacobs wildfire occurred on August 6, 2008, and the fire burned approximately 172 ha (425 ac) within a perimeter of ~10.7 km (6.6 mi) in a previously unburned blackbrush community. The data and results presented here summarize a three-year monitoring effort (2009-2011) in burned and unburned portions of the Jacob Fire (Figure 2).

METHODS

Data – field collection

Vegetation surveys in 50 cm^2 (7.8 in^2) plots were conducted on three dates in 2009: 1-May, 6-July, and 20-October; on two dates in 2010: 14-May and 28-May; and on 4-May 2011. Geomorphologic setting of the plots was recorded as ‘drainage’ or ‘upland’ based on landscape locale. If located in a drainage the plot was labeled ‘drainage’ while if it were anywhere other than a drainage it was labeled ‘upland’. For each geomorphologic setting the plot was located either within the shaded extent of a shrub canopy (‘understory’) or in the inter-shrub space (‘interspace’). Species identified and the number of individual plants by species was recorded in each plot. Plot locations were recorded using a handheld global positioning unit (GPS) in UTM zone 11, NAD83 datum. Field data were recorded on paper data sheets and digitized for analysis. At each location the GPS coordinate was recorded coincident with vegetation plot data for both spatial and statistical analysis.

Diversity Metrics

Standard diversity metrics were calculated to quantify vegetation biodiversity. Density was calculated as the total number of plants per plot. Richness was calculated as the number of different species recorded for each 0.25 m^2 (2.7 ft^2) plot. The Shannon-Weiner diversity index (H') (Lande, 1996) was calculated as:

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

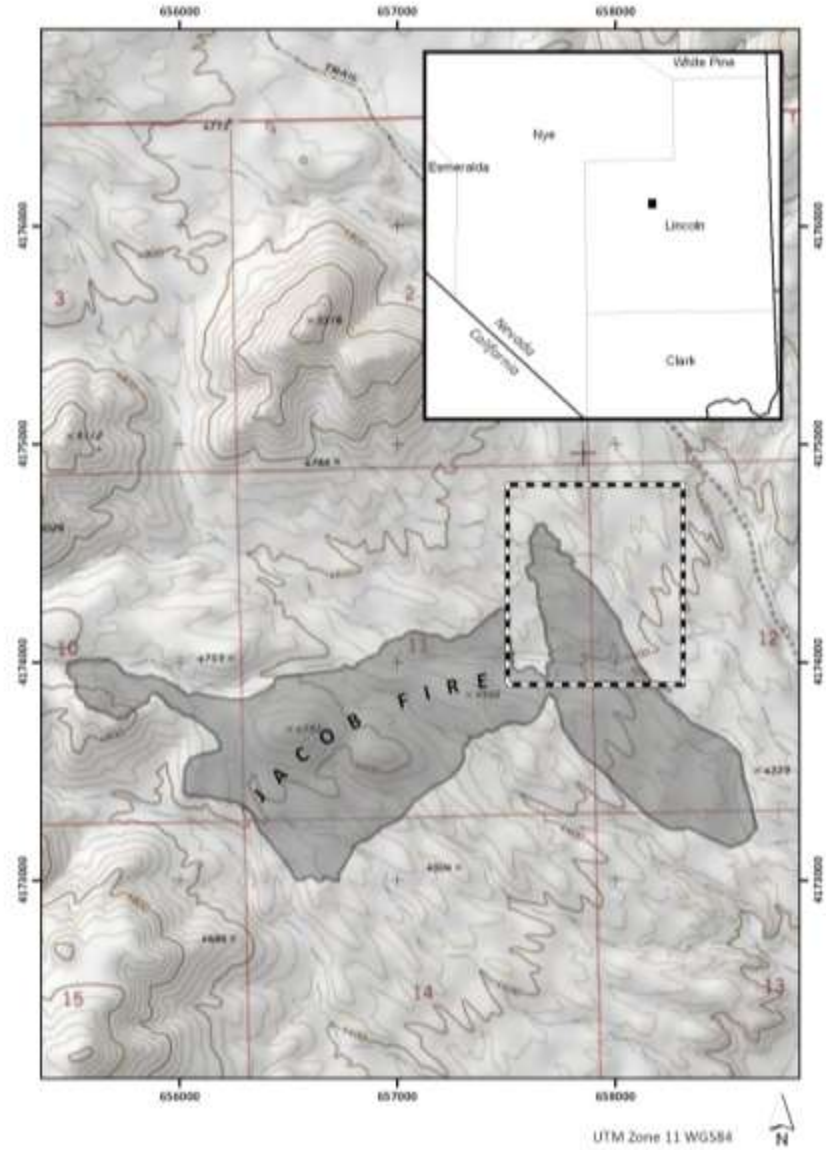


Figure 2. Study area location within Lincoln County, Nevada and 2008 Jacob Fire outline. Study plots were located within the area outlined by the dashed line.

where p_i is the proportion of individual plants of the i-th species. The Shannon-Weiner index, which accounts for both abundance and evenness, is an entropy index and is sensitive to the rarity and commonness of species recorded. H' is maximized when plots have equal counts for species recorded and decreases with unequal abundance across species. Lower values of H' indicate low diversity (low richness and evenness).

Evenness (J') is a measure of relative abundance among the species and returns values from 0 to 1 and is calculated as:

$$J' = \frac{H'}{H'_{max}}$$

where H' is Shannon-Weiner index at a particular sample location and H'_{max} is the maximum H' for all sample locations. Values closer to 1 indicate evenness or equal distribution of abundance of species while values approaching 0 indicate unequal (*e.g.* skewed) distribution of species abundance.

Simpson's diversity index (D) (Lande, 1996) was calculated per plot and is a measure of biodiversity that accounts for abundance, richness and evenness as follows:

$$D = \sum_{i=1}^{S} p_i^2$$

where p_i is the number of species S. This index returns values ranging from 0 (infinite diversity) to 1 (no diversity). D values near 0 correspond to highly diverse, heterogeneous plots while those plots returning a value closer to 1 are more homogeneous. It is a probability function that determines the likelihood that any two plants randomly sampled would be from the same species. Both Simpson's Index and Shannon-Weiner index were calculated because of the relatively low richness of the area, particularly post-fire. Finally, dominance for each species recorded in plots was calculated. Dominance is the number of plants of species *i* divided by the total density across all plants in the plot.

Statistical analyses

Basic summary statistics were calculated for all sample groups within the burned and unburned categories. Spearman correlation was conducted to assess the degree to which a relationship existed between richness and density of burned and unburned plots, respectively and plotted. F-tests for dispersion were run to test for equal variability between groups for each of the metrics, i.e., richness and density. If, based on F-test results, the groups exhibited unequal variances then Welch's approximation for unequal variances was used in independent t-tests. The t-tests were run to compare means between groups for each metric of richness and density. Analysis of variance (ANOVA) was conducted to compare burned with unburned plots at the gross comparative level for both richness and density.

Spatial statistical analyses

Spatial statistics provide a means to assess the relationship between geographic location and a response of interest (*e.g.* field measurement or calculated metric). The data were imported into ArcGIS to conduct spatial statistical analyses and to visualize both field recorded and calculated response values, in geographic space. Two different spatial statistical calculations were run to address spatial pattern of the biological response data, average nearest neighbor (ANN) and Moran's I, which is a commonly used measure of spatial autocorrelation, i.e., the similarity of data values as a function of their location or proximity to each other.

Average nearest neighbor analysis was performed to determine the likelihood that plot locations were the result of random chance and if not, to identify if the spatial pattern of the plots was clustered or dispersed (systematic). The ANN ratio (Ebdon, 1985) is calculated as:

$$ANN = \frac{\overline{D}_O}{\overline{D}_E}$$

where $\overline{D}_O$ is the observed mean distance between each point and its nearest neighbor:

$$\overline{D}_O = \frac{\sum_{i=1}^{n} d_i}{n}$$

and $\overline{D}_E$ is the expected mean distance for the data points given a spatially random pattern:

$$\overline{D}_E = \frac{0.5}{\sqrt{n/A}}$$

Above, d_i is the distance between feature *i* and its nearest neighbor, and *n* is the total number of features in the study area of size *A*. In this study Euclidean distance was used although Manhattan distance may also be used. Euclıean distance is the length of a straight line between two points. Manhattan distance is the distance calculated between two points measured by summing the vertical and horizontal distance if a grid were overlaid on the sampling area.

The null hypothesis in ANN is that the pattern exhibits complete spatial randomness, where the ratio equals a value of one. A z-score, i.e., the difference between one data value and the data mean divided by the data standard deviation, with a corresponding p-value (the probability or confidence that the null hypothesis is correct) determines whether or not to reject the null hypothesis. The z-score is the determinant of whether the pattern is clustered or dispersed and the magnitude of the spatial pattern, if any. An ANN ratio less than 1 indicates clustered spatial pattern while a value greater than 1 indicates dispersion.

Moran's I was calculated for each of density and richness metrics for all plots. Moran's I calculates spatial autocorrelation based on both the geographic point location and the point value at that location. Scores range from -1 (dispersed) to 1 (clustered). In this manner it identifies spatial pattern (dispersed, random, a zero value, or clustered) of the measured variable. A z-score and p-value was produced to determine the significance level of the Index value. Moran's I (Moran, 1950) is calculated as:

$$I = \frac{n}{S_O} \frac{\sum_{i=1}^{n} \sum_{j=1}^{n} w_{i,j} z_i z_j}{\sum_{i=1}^{n} z_i^2}$$

where z_i is the deviation of an attribute for feature *i* from its mean $(x_i - \bar{X})$, $w_{i,j}$ is the spatial weight between features *i* and j, n = the total number of features, and S_O is the aggregate of all spatial weights:

$$S_O = \sum_{i=1}^{n} \sum_{j=1}^{n} w_{i,j}$$

The null hypothesis in Moran's I is that the attribute analyzed (here density or richness) is randomly distributed among the locations sampled. The results of this analysis are interpreted in conjunction with the underlying spatial sampling pattern.

2009 RESULTS AND DISCUSSION

A total of 60 plots were sampled in the unburned area and a total of 80 plots were sampled in the burned area in 2009 (Table 1). Twelve different species were identified across all plots (Table 2). Basic summary statistics of density and richness are presented in Table 3 for each sampling unit category.

Table 1. Sample distribution for 2009.

	Burned		**Unburned**		
	Drainage	Upland	Drainage	Upland	*total*
Interspace	20	20	10	20	70
Understory	20	20	10	20	70
total	40	40	20	40	*140*

Table 2. Species identified in plots in 2009 surveys. Code = species abbreviation. Treatment = U (found only in unburned plots), Both (found in both burned and unburned plots).

Code	**Genus**	**Species**	**Duration**	**Growth**	**Native/ exotic**	**Treat-ment**
BRTE	*Bromus*	*tectorum*	annual	grass	exotic	Both
CHFR	*Chaenactis*	*fremontii*	annual	forb	native	Both
CORA	*Coleogyne*	*ramosissima*	perennial	shrub	native	Both
CRAN	*Cryptantha*	*angustifolia*	annual	forb	native	Both
CRCI	*Cryptantha*	*circumsissa*	annual	forb	native	U
DEPI	*Descurainia*	*pinnata*	annual	forb	native	U
GICA	*Gilia*	*cana*	annual	forb	native	Both
MEAL	*Mentzelia*	*albicaulis*	annual	forb/herb	native	Both
MIPA	*Mimulus*	*parryi*	annual	forb	native	Both
NADE	*Nama*	*demissum*	annual	forb	native	U
PHFR	*Phacelia*	*fremontii*	annual	forb	native	Both
VUOC	*Vulpia*	*octoflora*	annual	grass	native	Both

Table 3. Summary statistics (mean ± standard deviation) calculated for each sample plot category. Values are per 0.25m^2 plot.

	Burned				**Unburned**			
	Drainage		Upland		Drainage		Upland	
	Density[1]	*Richness*	*Density*	*Richness*	*Density*[1]	*Richness*[2]	*Density*[1]	*Richness*[1]
Interspace	18.8±17.2	1.5±0.9	22.5±28.8	1.7±1.23	3.1±2.6	1.6±1.3	30.8±32.1	2.4±1.0
Understory	117.3±72.3	1.9±0.98	18.7±13	2.4±1.1	97.5±48.1	2.7±0.8	80±39.3	3.2±1.0

[1] Significant at α = 0.01

[2] Significant at α = 0.05

Unburned plots

The most abundant species in unburned plots was *Vulpia octoflora*, which is a native annual grass commonly called sixweeks fescue or sixweeks grass (Figure 3). It is occurs widely across North America. The plant responds to fire by re-establishing from soil-stored seed in early post-fire plant communities. There are few studies reported to provide what would be considered a typical post-fire response (http://www.fs.fed.us/database/feis/plants/graminoid/vuloct/all.html#FIRE%20EFFECTS). *Bromus tectorum*, commonly called cheatgrass, was the second most abundant species. Cheatgrass is a highly invasive annual grass that occurs throughout North America. It establishes post-fire from soil seed banks and is highly adapted to frequent fire and is highly competitive with other species especially natives.

ANOVA results showed similar densities ($F(19,40) = 0.32$, $p = 0.995$) and richness ($F(19,40) = 0.63$, $p = 0.858$) across plots sampled in the unburned areas. Mean plant density in unburned plots was 53.6 ± 48.5 and mean richness was 2.6 ± 1.1. Spearman correlation (Figure 4) showed a positive relationship ($r^2 = 0.52$) between density and richness for unburned plots ($r(58) = 4.64$, $p < 0.0001$).

Plant densities in unburned plots were not significantly different between drainages (50.3 ± 58.7/plot) and uplands (55.3 ± 43.2/plot) ($t(58) = -0.37$, $p = 0.711$) and exhibited similar variability ($F(19, 39) = 1.84$, $p = 0.105$). Richness in unburned plots exhibited no difference in variability between drainages and upland sites ($F(19, 39) = 1.21$, $p = 0.595$). Drainages (2.2 ± 1.2) were significantly less rich than upland sites (2.8 ± 1.1) ($t(58) = -2.06$, $p = 0.0222$, 95% CI -∞ to -0.1). In unburned plots results from Shannon-Weiner index showed drainage areas ($\overline{H}' = 0.662$) were marginally significantly more diverse than upland areas ($\overline{H}' = 0.467$) ($t(49) = 2.27$, $p = 0.027$). Simpson's Index results showed that unburned drainages ($\overline{D} = 0.688$) were not significantly different than unburned uplands ($\overline{D} = 0.771$) ($t(58) = -1.54$, $p = 0.1299$). Mean evenness for unburned drainages ($J' = 0.772$) and unburned uplands ($J' = 0.436$) were significantly different ($t(45) = 4.66$, $p < 0.0001$). The unburned uplands exhibited more equal distribution of plant abundance whereas unburned drainages were skewed towards having only few species.

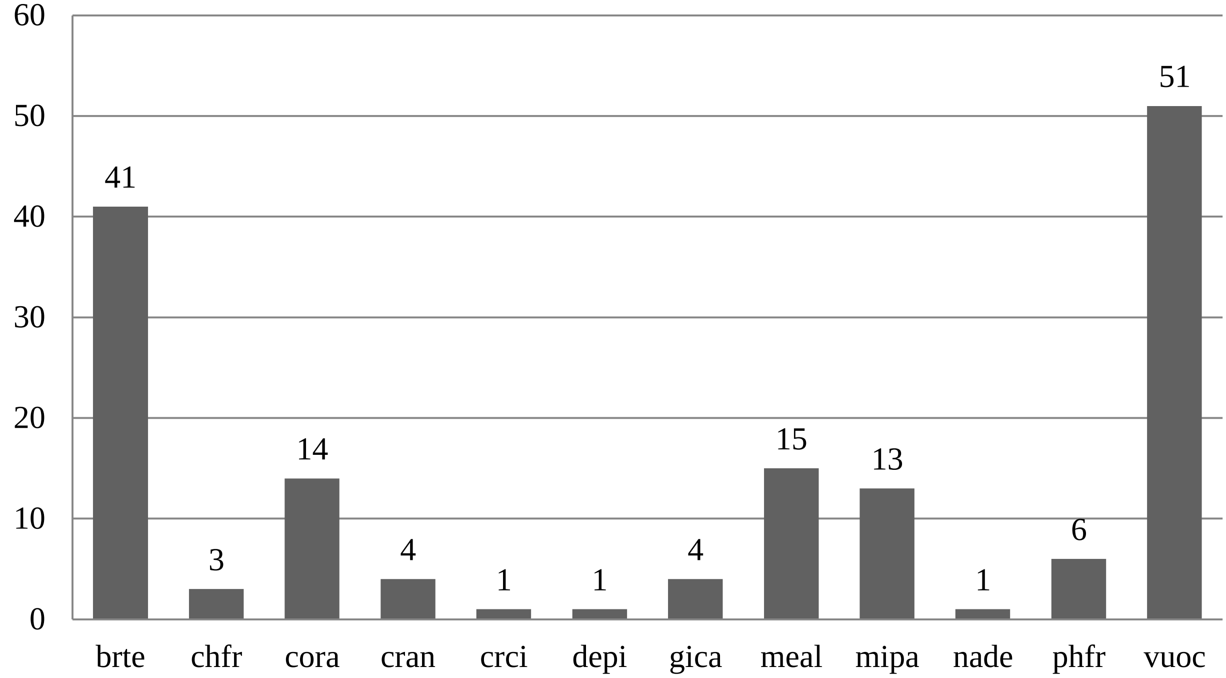


Figure 3. Frequency distribution (abundance) of species identified in unburned plots in 2009. See Table 2 for species code definition.

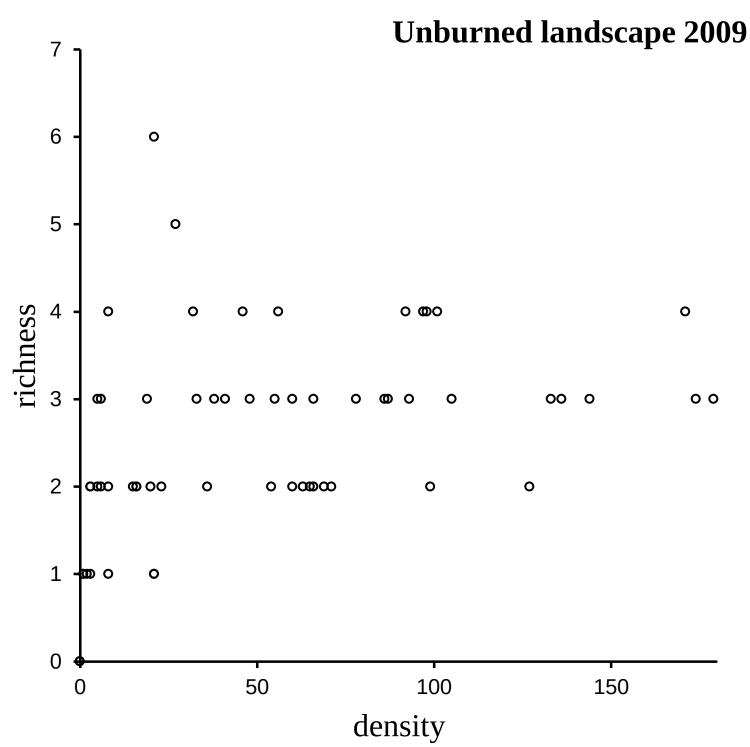


Figure 4. Correlation plot for diversity and richness in unburned plots (2009).

In examining microsite differences in unburned plots, the variability in plant density was marginally significantly different for ‘interspace’ (21.6 ± 29.2/plot) sites compared to ‘understory’ (85.7 ± 42.5/plot) sites (F(29, 29) = 0.47, p = 0.049). Sites characterized as ‘interspace’ were less variable than ‘understory’ sites (F(29, 29) = 0.47, p = 0.025) although the 95 percent CI included zero (0.00 to 0.88). Mean density was significantly lower in the ‘interspace’ sites than in the ‘understory’ sites (t(58) = -6.81, p < 0.0001). Richness of ‘interspace’ (2.1 ± 1.2/plot) and ‘understory’ (3.0 ± 0.9/plot) exhibited similar variability (F(29,29) = 1.55, p = 0.244) although mean richness was significantly less in ‘interspace’ sites than in ‘understory’ sites (t(58) = -3.45, p = 0.0005).

Results from Shannon-Weiner index showed ‘interspace’ ($\overline{H}' =$0.546) and ‘understory’ ($\overline{H}' =$ 0.509) microsites had similar diversity (t(49) = 0.45, p = 0.653). Simpson’s Index results also showed that ‘interspace’ ($\overline{D}$ = 0.782) and ‘understory’ ($\overline{D}$ = 0.704) microsites were not significantly different (t(58) = -1.52, p = 0.133). Both microsite types exhibited fairly low diversity. Mean evenness was similar for both ‘interspace’ (J'= 0.547) and ‘understory’ (J'= 0.516) microsites (t(45) = 0.39, p = 0.696).

Burned plots

A total of nine species were identified in the burned plots (Figure 5). *Mentzelia albicaulis* or whitestem blazingstar was the most abundant species identified in the burned plots. This is an annual native forb/herb found throughout the United States. The second most abundant species in burned plots was *Vulpia octoflora*, a native annual grass commonly called sixweeks fescue or sixweeks grass. *Bromus tectorum*, the highly flammable invasive cheatgrass, was the fourth most abundant species behind the native dominant shrub, blackbrush (*Coleogyne ramosissima*).

ANOVA results showed similar densities (F(19,60) = 0.50, p = 0.954) and richness (F(19,60) = 1.30, p = 0.221) between plots sampled in unburned areas. Mean plant density in burned plots was 44.3 ± 58 and mean richness was 1.8 ± 1.1. Spearman correlation (Figure 6) showed a positive relationship (r^2 = 0.37) between density and richness for unburned plots (r(78) = 3.50, p < 0.0008).

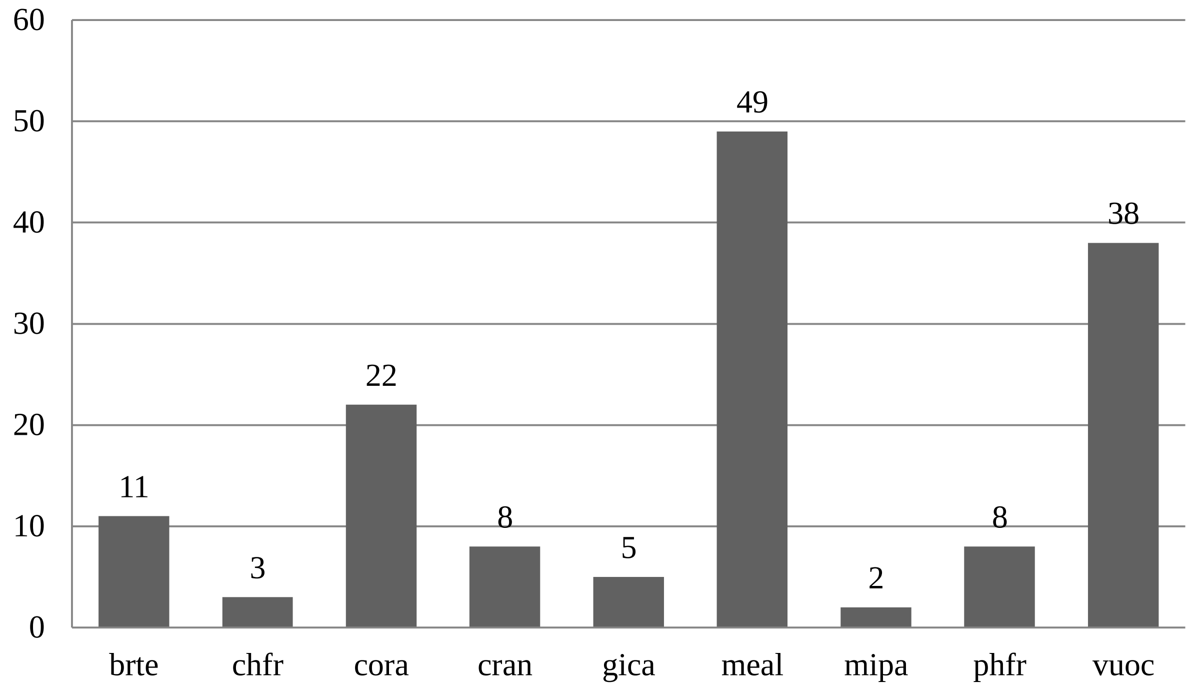


Figure 5. Species identified in burned plots in 2009. Total abundance presented for each species. See Table 2 for species code definition.

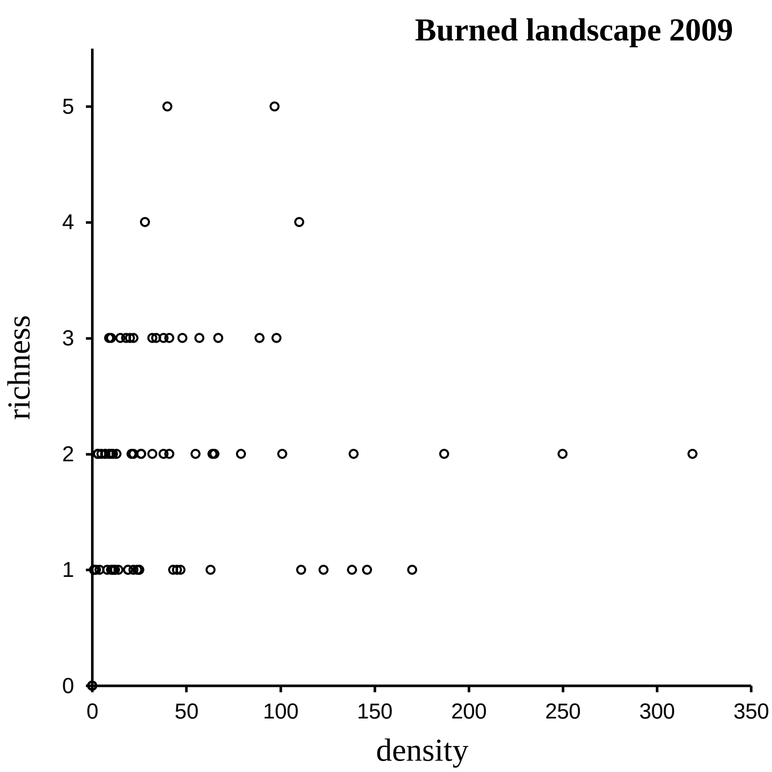


Figure 6. Correlation plot between density and richness for burned plots sampled in 2009.

Plant densities in the burned plots were significantly different between drainages (68.1 ± 72.0/plot) and uplands (20.6 ± 22.1/plot) ($t(46.3) = 3.99$, $p = 0.0002$). The variability of plant density was significantly greater in drainages than upland sites ($F(39, 39) = 10.59$, $p < 0.0001$). Richness in burned plots was not significantly different for drainages (1.7 ± 0.9) compared to upland sites (2.0 ± 1.2) ($t(78) = -1.48$, $p = 0.143$) and plots exhibited no difference in variability between the two ($F(39, 39) = 0.56$, $p = 0.07$).

In examining microsite differences, the variability in plant density was significantly different for 'interspace' (20.6 ± 23.5/plot) sites compared to 'understory' (68 ± 71.6/plot) sites ($F(39, 39) = 0.11$, $p < 0.0001$). Mean density was significantly lower in the 'interspace' sites than in the 'understory' sites ($t(78) = -3.98$, $p = 0.0002$). There was a significant difference in richness between 'interspace' (1.6 ± 1.1) and 'understory' (2.1 ± 1.0) sites ($t(78) = 2.38$, $p = 0.0199$) although the variability in plot richness was not significantly different ($F(39, 39) = 1.11$, $p = 0.749$).

Comparing burned versus unburned plots

All of the species identified in burned plots were found in unburned plots however, three species were unique to the unburned plots, namely cushion cryptantha (*Cryptantha circumscissa*), western tansymustard (*Descurainia pinnata*), and purplemat (*Nama demissum*). Whitestem blazingstar was found three times as often in burned plots compared to the unburned plots and was the most frequently identified species in burned plots. Sixweeks fescue was next most prevalent, although this native grass was less often reported in burned than unburned plots. In the burned landscape sixweeks fescue and cheatgrass were the most frequently reported species followed by whitestem blazingstar and blackbrush. In comparison, the most frequently reported species in the burned plots was whitestem blazingstar, sixweeks fescue, blackbrush and cheatgrass in that order. Cheatgrass was reported nearly four times as often in the unburned landscape than the burned.

Within one year post-fire drainages exhibited similar densities and richness. Burned uplands were significantly less vegetatively dense and less rich primarily as a function of the difference between sites under shrub canopy cover.

Both burned and unburned sites had at least one plot where no vegetation were present. Burned and unburned mean density and richness in 2009 are compared for drainage and upland plots in Figure 7. Mean plot density for burned and unburned locations by plot type and geomorphic setting is presented in Figure 8 and Figure 9, respectively. At the landscape level, results showed no significant difference in the mean plant density between burned (44.3 ± 58/plot) and unburned (53.6 ± 48.5/plot) plots ($t(138) = 1.01$, $p = 0.3162$) nor was there a significant difference in variability of density between burned and unburned plots ($F(59, 79) = 0.7$, $p = 0.149$). Mean richness of unburned plots was 2.6 ± 1.1 and for burned plots was 1.8 ± 1.1. Unburned plots showed significantly higher richness than burned plots ($t(138) = 3.96$, $p < 0.0001$) but were similar in richness variability ($F(59, 79) = 1.15$, $p = 0.565$).

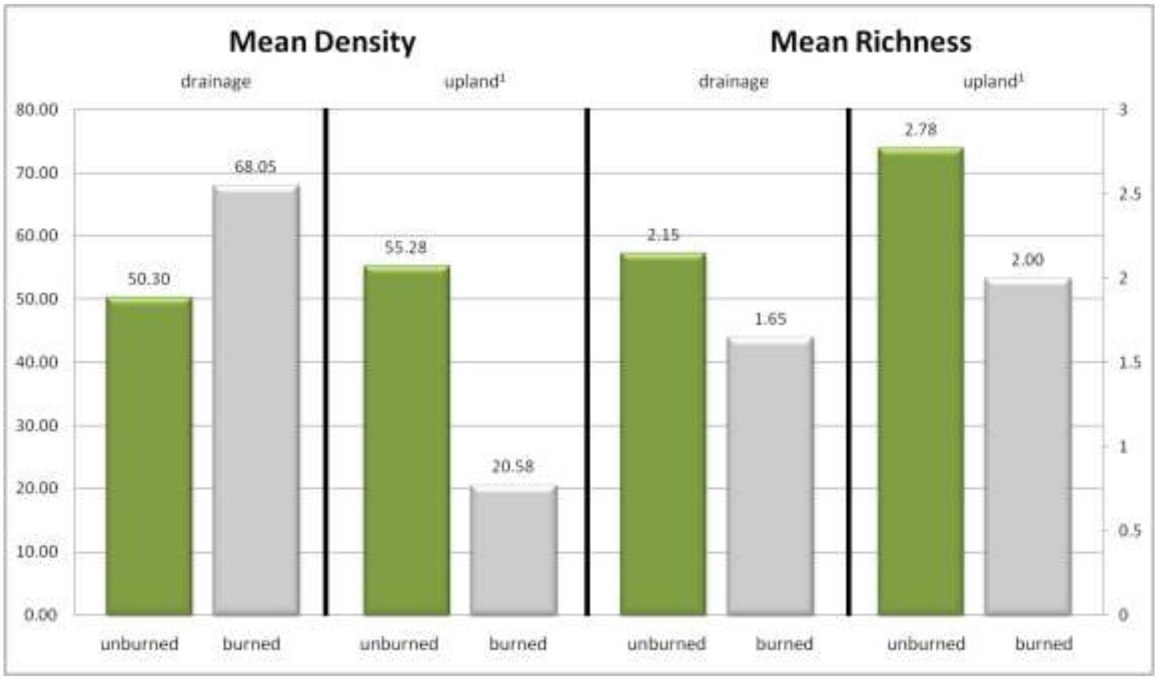


Figure 7. Mean density and mean richness (labeled) for all unburned and burned plots in each configuration of drainage or upland in 2009. [1] indicates a significant difference.

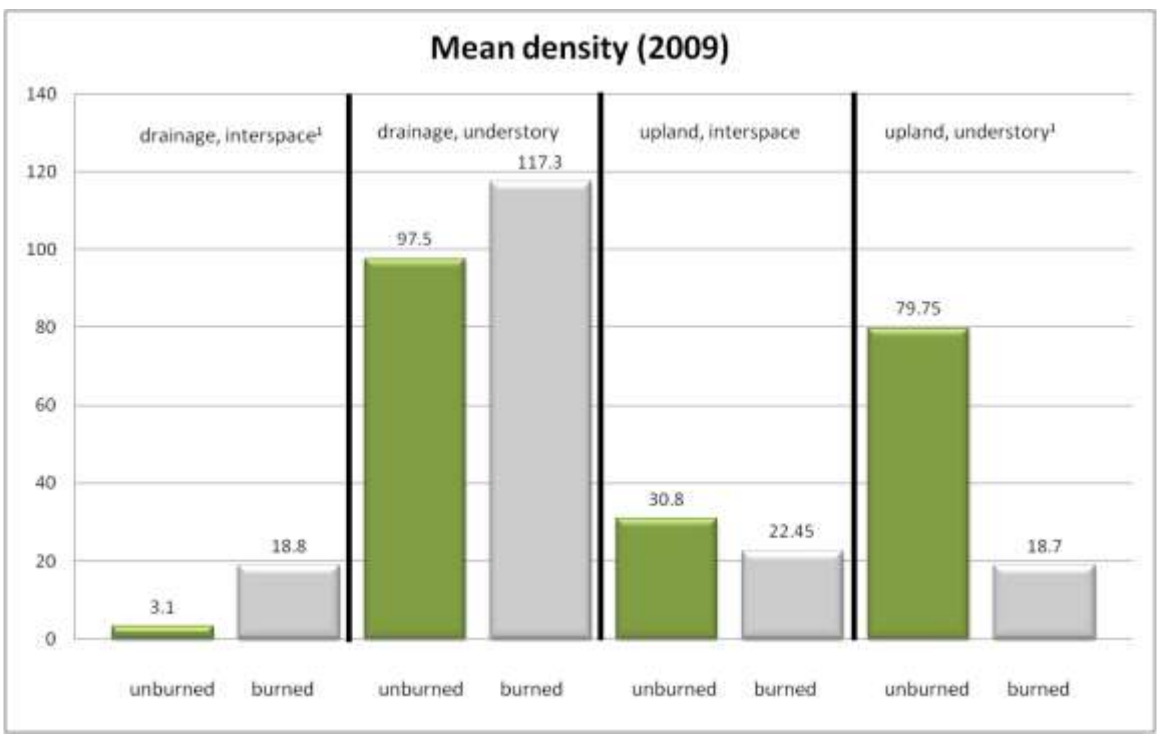


Figure 8. Mean density (labeled) by plot type for 2009. [1] indicates a significant difference.

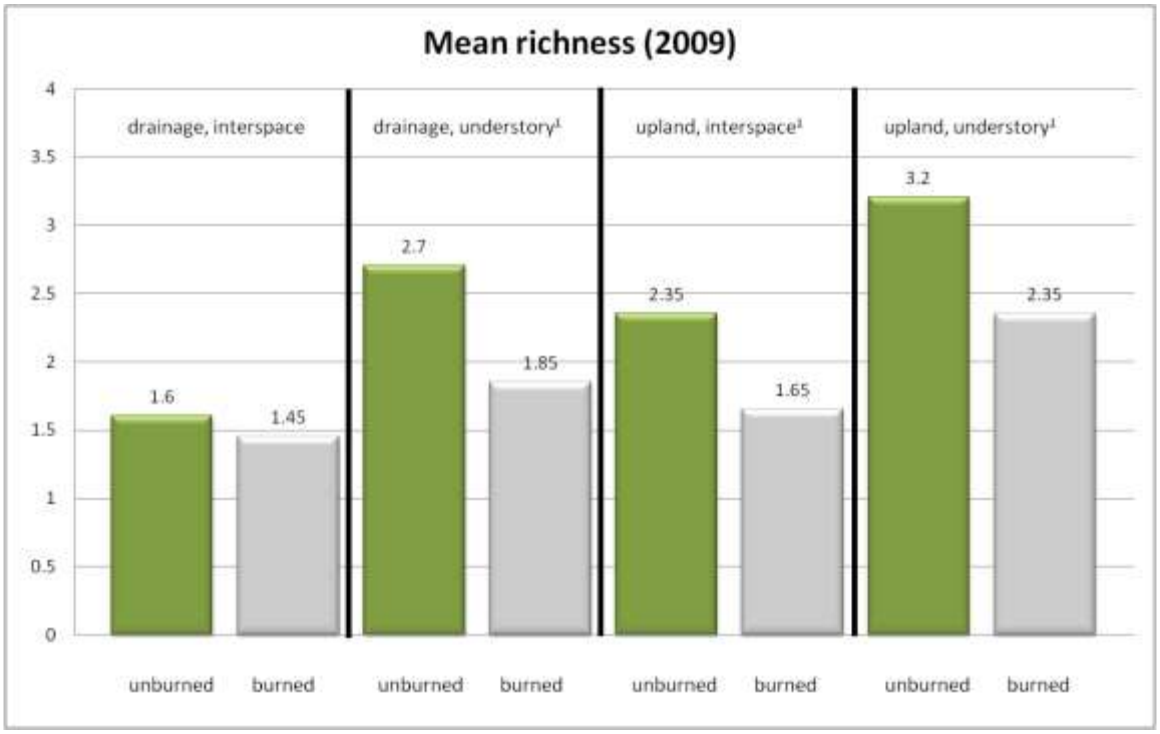


Figure 9. Mean richness (labeled) by plot type for 2009. [1] indicates a significant difference.

Results from Shannon-Weiner index showed unburned areas ($\overline{H}' = 0.477$) were significantly more diverse than burned ($\overline{H}' = 0.267$) areas ($t(128) = 3.86$, $p = 0.0002$). Simpson's Index (D) showed that unburned areas ($\overline{D} = 0.743$) were significantly more diverse than burned areas ($\overline{D} = 0.845$) ($t(138) = -2.93$, $p = 0.004$). It should be noted that both of these $\overline{D}$ values are indicative of low diversity in general for both burned and unburned plots as they approach a value of 1. Mean evenness for burned ($J' = 0.464$) and unburned ($J' = 0.529$) areas was not significantly different ($t(89) = 1.05$, $p = 0.296$). Both values of J' indicate some inequality of the distribution of plant abundance but not heavily skewed towards one species.

Diversity and evenness comparisons for burned and unburned, drainage and upland plots, respectively, are presented graphically in Figure 10. Burned drainages exhibited almost no diversity. There were relatively few different species recorded in burned drainages and of those recorded only a small few were found in abundance. In unburned drainages fewer different species were recorded however those species tended to occur in relatively even proportions. Upland areas were no different in diversity although burned uplands exhibited a more even distribution of species.

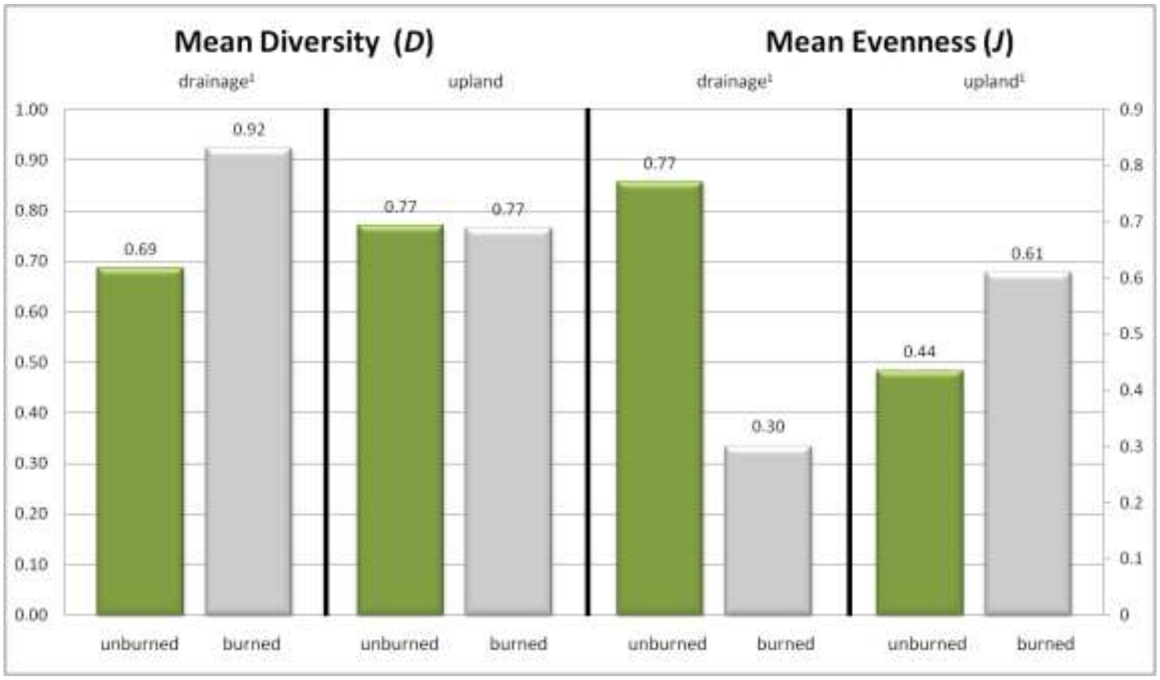


Figure 10. Mean diversity (Simpson's Index) and evenness for drainage and upland locations, comparing burned and unburned plots (2009). [1] indicates a significant difference. Diversity (*D*) ranges from 0 (infinite diversity) to 1 (no diversity). Evenness (*J*) ranges from 0 (skewed) to 1 (even).

Evaluating 'interspace' microsite plots, there was no difference in diversity between unburned ($\overline{D}$ =0.782) and burned ($\overline{D}$ =0.855) plots ($t(68) = -1.44$, $p = 0.1548$) and both exhibited similar evenness ($t(35) = 0.27$, p =0.7868) (unburned J'= 0.547; burned J'= 0.521). These sites have low diversity in general. For 'understory' microsite plots, unburned plots ($\overline{D}$ =0.704) were more diverse than burned ($\overline{D}$ =0.836) (*t*(65 = -2.76, $p = 0.0075$) although both returned values showing relatively homogeneous conditions. Burned (J'= 0.424) and unburned (J'= 0.516) 'understory' microsites showed no difference in evenness ($t(52) = 1.14$, $p = 0.259$).

<u>Drainages</u>

Comparing only those plots in drainages regardless of microsite, there was no significant difference in densities between burned (68.1 ± 72/plot) and unburned sites (50.3 ± 58.7/plot) ($t(58) = -0.95$, p =0.344). In drainages the burned plots and unburned plots exhibited similar variability ($F(19, 39) = 0.67$, $p = 0.343$). Neither mean richness ($t(58) = 1.83$, p =0.072) nor variability in richness ($F(19, 39) = 1.75$, p =0.137) was significantly different in burned (1.7 ± 0.9) and unburned (2.2 ± 1.2) drainage plots.

Simpson's Index returned significant differences in diversity between burned ($\overline{D}$ =0.924) and unburned ($\overline{D}$ = 0.688) drainages ($t(58) = -5.25$, $p < 0.0001$). Burned drainages exhibited almost no diversity. Burned drainages were significantly different (J'= 0.302) than unburned drainages (J'= 0.772) ($t(32) = 5$, $p < 0.0001$). The distribution of species in burned drainages is relatively skewed.

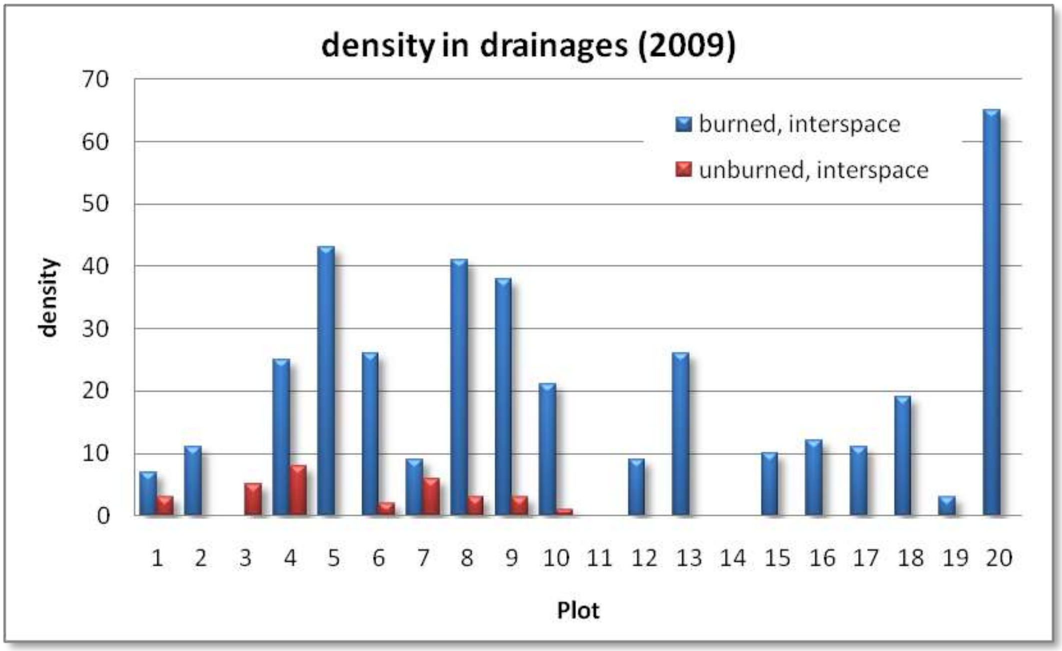
Figure 11 presents the density and richness data for drainage plots by 'interspace' microsite. Mean plant density in drainages was lower in unburned, 'interspace' areas of drainages (3.1 ± 2.6/plot) than in burned, 'interspace' areas in drainages (18.8 ± 17.2/plot) (t(20.7) = -3.99, p = 0.0007). The variability in plant density was significantly different; burned, 'interspace' sites in drainages exhibited greater variability than unburned, 'interspace' drainage sites (F(9, 19) = 0.02, p < 0.0001). Richness between burned (1.5 ± 0.9/plot) and unburned (1.6 ± 1.3/plot) 'interspace' drainage sites was not significantly different (t(28) = 0.38, p = 0.708) nor different in variability (F(9,19) = 2.03, p = 0.185).

Figure 12 presents the density and richness data for drainage plots by 'understory' microsite. There was no difference in mean density for plots in 'understory' microsites in drainages between the burned (117.3 ± 72.3) and unburned (97.5 ± 48.1) plots (t(28) = -0.78, p = 0.442). Drainage, 'understory' sites that had burned exhibited no difference in variability of plant density than unburned, drainage 'understory' sites (F(9, 19) = 0.44, p = 0.212). Richness between burned (1.9 ± 0.9/plot) and unburned (2.7 ± 0.8/plot) 'understory' drainage sites was significantly different (t(28) = 2.56, p = 0.016). The variability in richness was not significantly different (F(9,19) = 0.89, p = 0.89).

Upland sites

Comparing only those upland plots regardless of microsite, there was a significant difference in densities between unburned (55.3 ± 43.2/plot) and burned sites (20.6 ± 22.1/plot) (t(58.1) = 4.52, p < 0.0001) and in the variability of density (F(39,39) = 3.82, p < 0.0001). Mean richness was significantly different in burned (2.0 ± 1.2) and unburned (2.8 ± 1.1) upland plots (t(78) = 3.05, p = 0.003). Variability in richness was not significantly different (F(39, 39) = 0.80, p = 0.497).

There was not a significant difference in diversity (t(78) = 0.08, p = 0.9352) for burned ($\overline{D}$ =0.767) and unburned ($\overline{D}$ =0.771) upland plots. Unburned upland plots exhibited greater evenness (J'= 0.436) than burned (J'= 0.611) (t(55) = -2.63, p = 0.011). These results are the reverse for observations in drainages.



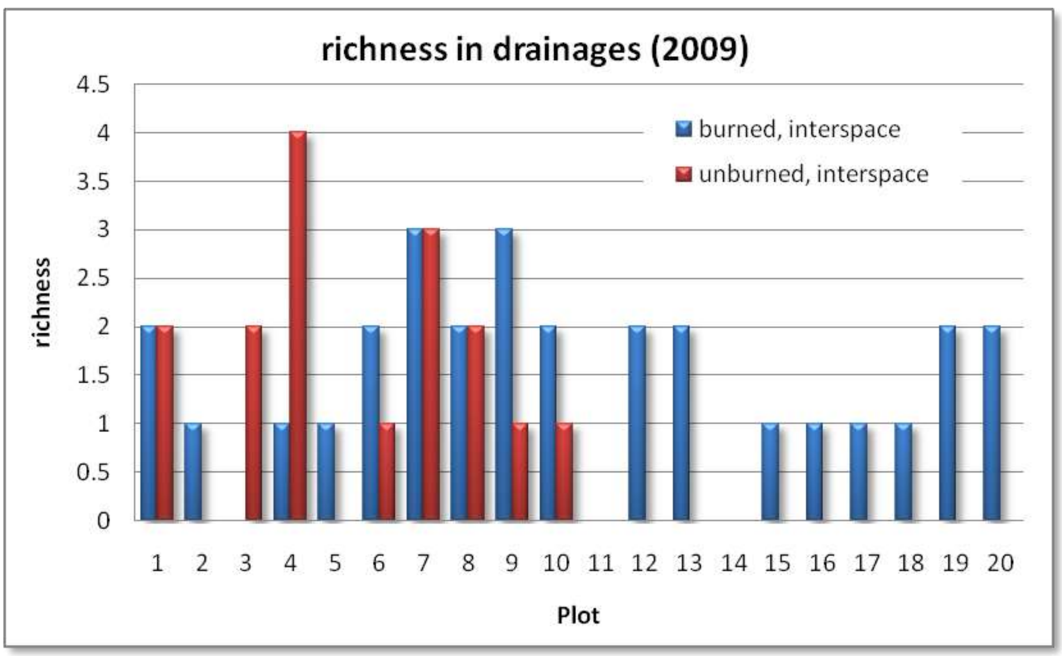


Figure 11. Density and richness for burned and unburned inter-canopy drainage sites.

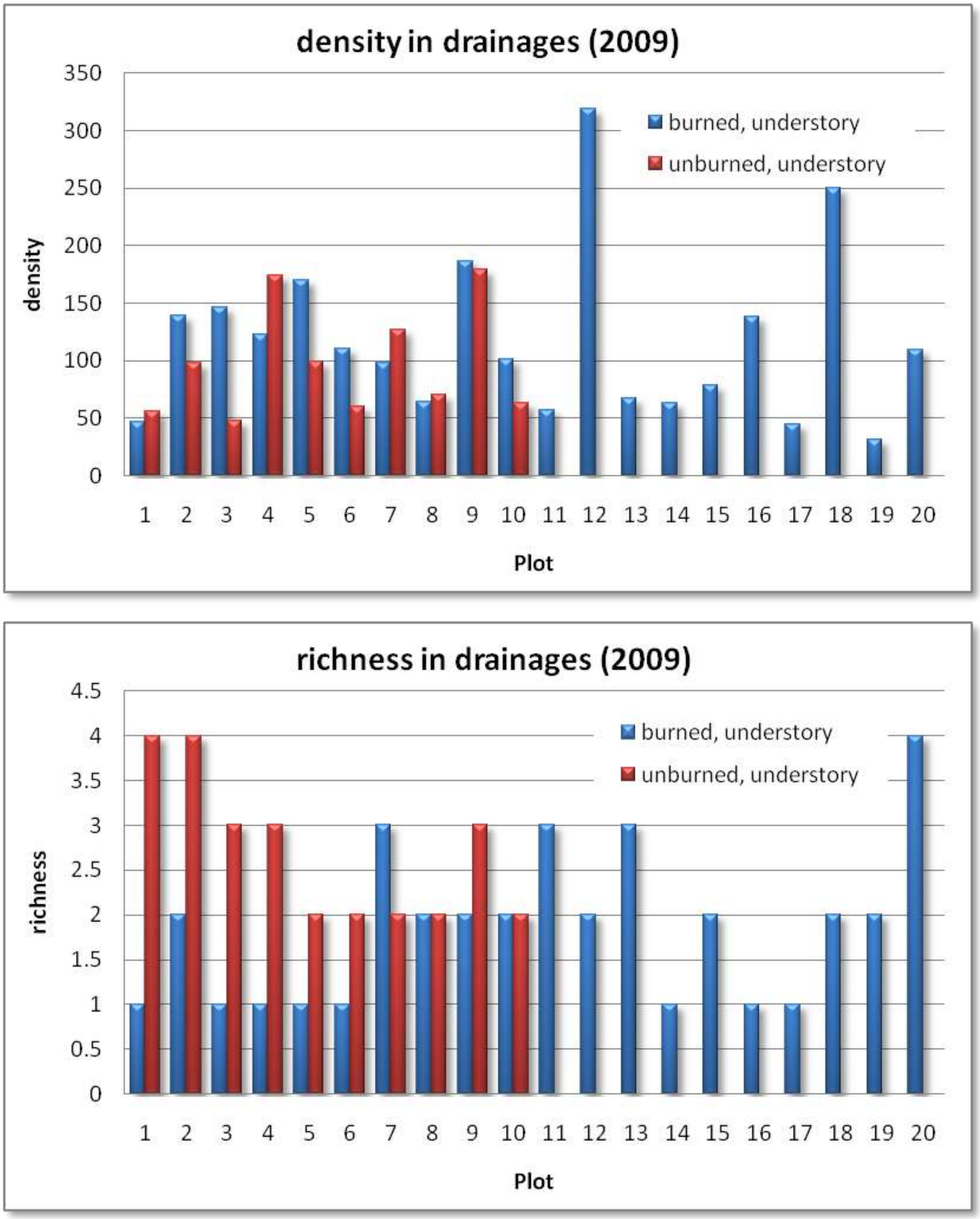


Figure 12. Density and richness data for burned and unburned drainages under the canopy.

Figure 13 presents the density and richness data for ‘interspace’ microsite location in upland plots. For upland, ‘interspace’ plots there was no difference in density ($t(38) = 0.87$, $p = 0.392$) of burned (22.5 ± 28.8) and unburned (30.8 ± 32.1) plots, nor a difference in variability of density between the two ($F(19, 19) = 1.25$, $p = 0.635$). Richness between burned (1.7 ± 1.2/plot) and unburned (2.4 ± 1.0/plot) upland ‘interspace’ sites was marginally significantly different ($t(38) = 1.95$, $p = 0.059$, 95% CI 0.0 to 1.4) and was not different in variability ($F(19,19) = 0.72$, $p = 0.480$).

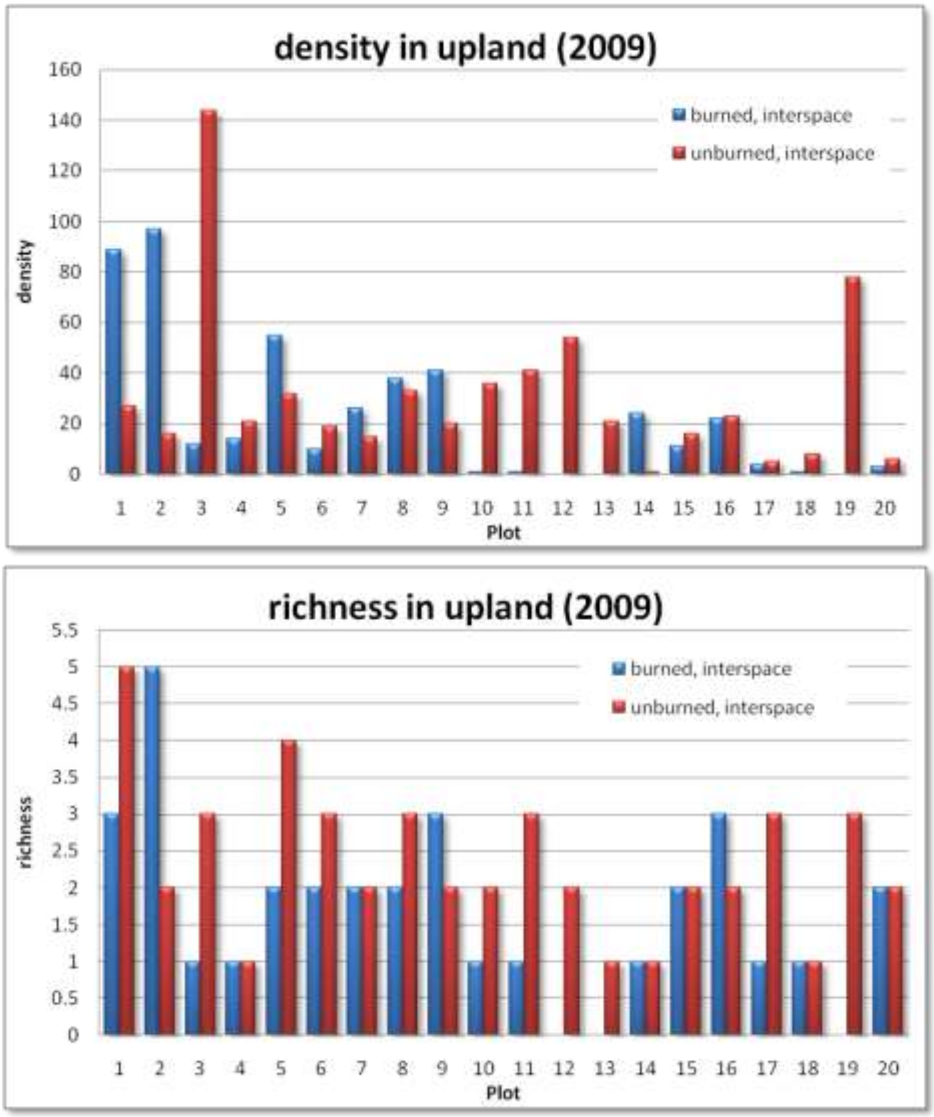


Figure 13. Density and richness for burned and unburned inter-canopy upland sites.

Figure 14 presents the density and richness data for 'upland' microsite plots under the canopy ('understory'). Mean density was significantly higher in unburned (79.8 ± 39.3/plot) than burned (18.7 ± 13/plot) upland 'understory' sites ($t(23.1) = 6.6$, $p < 0.0001$). There was a significant difference in the variability of density in upland 'understory' plots ($F(19,19) = 9.15$, $p < 0.0001$). Richness of unburned upland 'understory' plots (3.2 ± 1.0) was significantly different from burned (2.4 ± 1.1) upland 'understory' sites ($t(38) = 2.63$, $p = 0.012$) although the variability between the two was not significantly different ($F(19, 19) = 0.76$, $p = 0.561$).

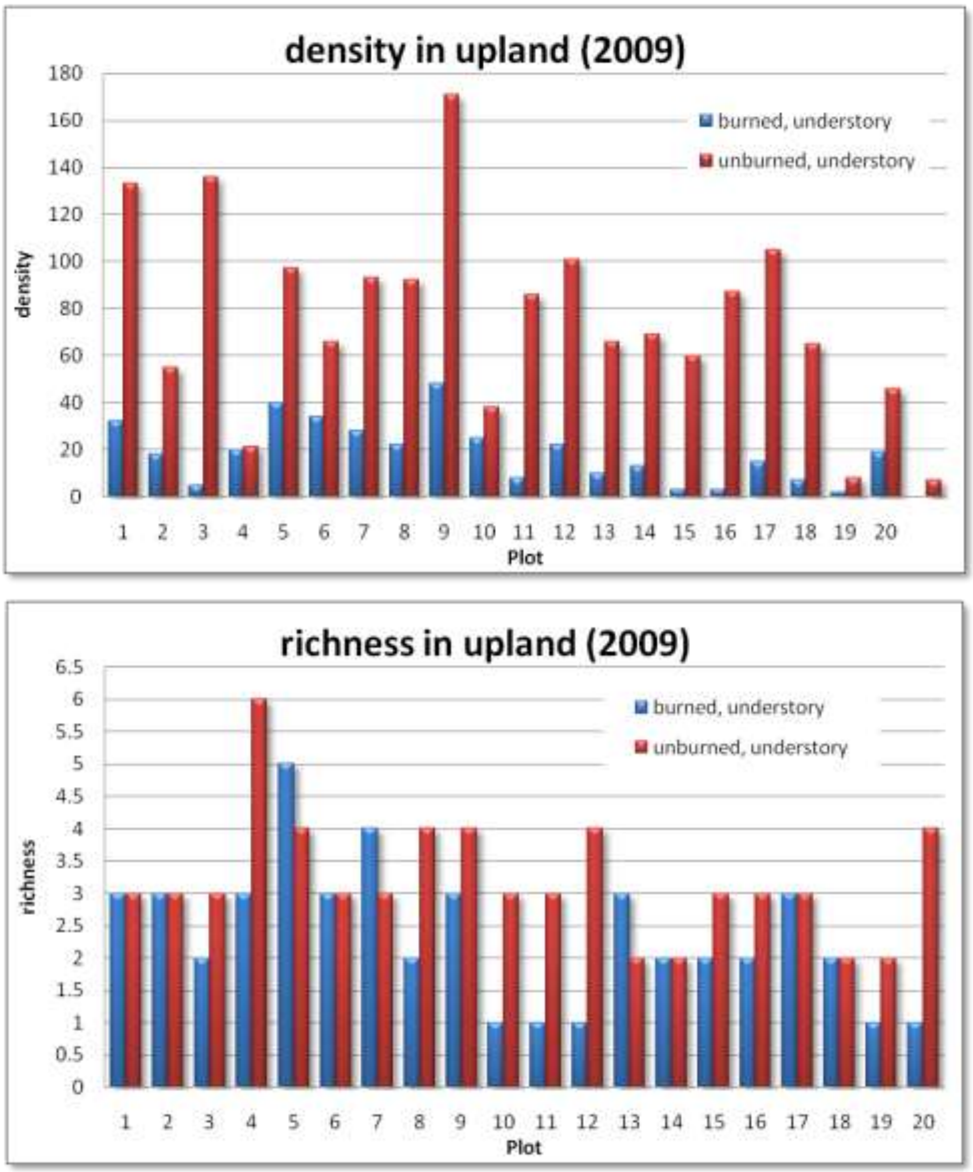


Figure 14. Density and richness data for burned and unburned uplands under the canopy.

Spatial analyses

Results from spatial analyses of plot data for density and richness are presented in Table 4. The pattern of evenness was random in both the unburned plots (I = 0.004, z = 0.369, p = 0.712) and burned plots (I = 0.084, z = 1.60, p = 0.109). For diversity (Simpson's Index) the pattern in the unburned plots was random (I = -0.093, z = -1.32, p = 0.186) while the diversity pattern in burned plots was clustered (I = 0.118, z = 2.185, p = 0.029).

Unburned sites

The unburned sample plots were spatially clustered (z = -14.82, p = 0.000). In the unburned area, drainage sample plots were located in a separate area from upland sample plots. The drainage plots in the unburned area exhibited a dispersed pattern (z = 8.298, p = 0.000). The upland plots exhibited spatial randomness (z = 0.532, p = 0.595). For the unburned plots, density pattern was spatially dispersed (I = -0.128, z = -1.96, p = 0.049) and richness pattern was spatially random (I = -0.014, z = 0.55, p = 0.956).

Neither unburned, upland 'interspace' or unburned, upland 'understory' plots were significantly different than random (z = 0.532, p = 0.595). Because 'interspace' and 'understory' plots were paired within 1 m, respectively, the spatial patterns are exactly the same. Results from Moran's I calculation showed random density pattern for unburned, upland 'interspace' plots (I = -0.06, z = -0.06, p = 0.952) and unburned, upland 'understory' plots (I = -0.090, z = -0.204, p = 0.838). Richness was spatially random for the unburned, upland 'interspace' plots (I = 0.047, z = 0.55, p = 0.583) and unburned, upland 'understory' plots (I = 0.018, z = 0.411, p = 0.681).

Shannon's Index of diversity was analyzed for spatial pattern. Diversity in unburned drainages exhibited a spatially random pattern (I = -0.22, z = -1.26, p = 0.2077) as did unburned uplands (I = -0.087, z = -1.064, p = 0.287). Evenness was spatially random in unburned drainages (I = 0.56, z = 0.808, p = 0.419) and in unburned uplands (I = -0.054, z = -0.496, p = 0.62).

Table 4. Summary of spatial pattern analysis results for sample pattern and for sample plot density and richness. R = Random; C = Clustered; D = Dispersed. Rich = richness, Divers = diversity (D), Even = evenness.

Burned						**Unburned**					
Drainage	*Upland*	*Density*	*Rich*	*Divers*	*Even*	*Drainage*	*Upland*	*Density*	*Rich*	*Divers*	*Even*
C	C	R	C	C	R	D	R	D	R	R	R

	Burned				**Unburned**			
	Drainage		Upland		Drainage		Upland	
	Density	*Richness*	*Density*	*Richness*	*Density*	*Richness*	*Density*	*Richness*
Interspace	R	R	R	R	R	R	R	R
Understory	R	C	C	C	R	C	R	R

The spatial pattern of sample plot distribution in unburned, drainage 'interspace' and 'understory' sites (again, same coordinate location as described above) is dispersed (z = 8.298, p = 0.000). Given the z-score there is less than a 1 percent likelihood this pattern could be the result of random chance. Results from Moran's I shows random spatial patterns for density (I = -0.355, z = -1.021, p = 0.308) and for richness (I = -0.474, z = -1.529, p = 0.126) in unburned, drainage 'interspace' sites. For the paired 'understory' sites density was also not significantly different from random (I = -0.123, z = -0.048, p = 0.961). Richness in the drainage 'understory' plots exhibited a clustered spatial pattern (I = 0.641, z = 3.051, p = 0.002). There is a less than 1 percent likelihood that the clustered pattern could be the result of random chance.

<u>Burned sites</u>

The burned sample plots were spatially clustered (z = -17.11, p = 0.000). Drainage plots in the burned area were spatially clustered (z = -12.10, p = 0.0000). Upland plots in the burned area were spatially clustered (z = 7.351, p = 0.000). The density pattern across the burned plots was spatially random (I = -0.0321, z = -0.339, p = 0.734) whereas the richness pattern was clustered (I = 0.156, z = 2.82, p = 0.005).

The burned, upland 'interspace' and 'understory' sample plot distributions were spatially clustered (z = -2.109, p = 0.035), respectively. These patterns are the same because of the paired plots being within 1 m apart as explained above. The density pattern for burned, upland interspace plots was no different from random (I = 0.076, z = 0.602, p = 0.547) and corresponding richness was also no different from random (I = -0.086, z = -0.153, p = 0.879). Density for burned, upland 'understory' plots was clustered (I = 0.407, z = 2.041, p = 0.0412). Richness for burned, upland 'understory' was clustered (I = 0.471, z = 2.356, p = 0.019).

Shannon's Index of diversity was analyzed for spatial pattern. Diversity in burned drainages exhibited a spatially random pattern (I = -0.045, z = -0.183, p = 0.855) as did burned uplands (I = 0.032, z = 0.605, p = 0.545). Evenness was spatially random in burned drainages (I = 0.53, z = 0.735, p = 0.462) and in burned uplands (I = 0.017, z = 0.443, p = 0.658).

The burned drainage 'interspace' and 'understory' sample plot distributions, also equal due to being sampled in pairs, was random (z = 0.510, p = 0.61). Density for these plots was no different from random (I = -0.060, z = -0.083, p = 0.934). Richness was also random (I = 0.03, z = 0.806, p = 0.42). The burned drainage 'understory' were spatially random for density (I = -1.119, z = 0.739, p = 0.46) and clustered for richness (I = 0.165, z = 2.33, p = 0.02).

Spatial statistics were not computed for 2010 data due to insufficient sample sizes as determined from analyzing the 2009 data.

Cheatgrass (Bromus tectorum) and Blackbrush (Coleogyne ramosissima) dominance

Cheatgrass was identified in 52 of 140 plots surveyed. Eighty-eight plots (63%) had no cheatgrass in 2009. It was identified in 41 (59%) unburned plots and in 11 (16%) burned plots. Mean dominance of cheatgrass in unburned plots where cheatgrass was identified was 0.584 ± 0.329. For burned plots the mean was 0.031 ± 0.029. The difference in cheatgrass dominance between burned and unburned plots was significantly different where unburned plots had predominantly more cheatgrass than burned plots (t(50) = 5.53, $p < 0.0001$). The dominance distribution for *B. tectorum* in burned and unburned plots is shown in Figure 15.

For plots surveyed in drainages cheatgrass was more predominant in unburned plots (0.56 ± 0.25) than in burned plots (0.033 ± 0.033) (t(19) = 0.525, $p < 0.0001$). In total, 39 of 50 drainage plots had no cheatgrass identified, 13 unburned plots had none and eight burned plots had none.

Of the 80 plots surveyed in upland areas, cheatgrass was identified in 28 unburned plots and in only 3 burned plots. Mean dominance in the unburned plots (0.596 ± 0.365) was significantly higher than in burned plots (0.025 ± 0.018) (t(29) = 2.67, $p = 0.0123$).

Blackbrush was identified in 36 of 140 plots surveyed. Mean blackbrush dominance for unburned plots (0.076 ± 0.062) was significantly lower than burned plots (0.466 ± 0.334) (t(34) = -4.31, $p = 0.0001$). Blackbrush was identified in two unburned drainage plots and was not found in any burned drainage plots. Blackbrush was identified in 22 burned upland plots and in 12 unburned upland plots. Blackbrush exhibited significantly higher dominance in burned upland plots (0.466 ± 0.334) than in unburned upland plots (0.064 ± 0.058) (t(32) = -4.11, $p = 0.0003$).

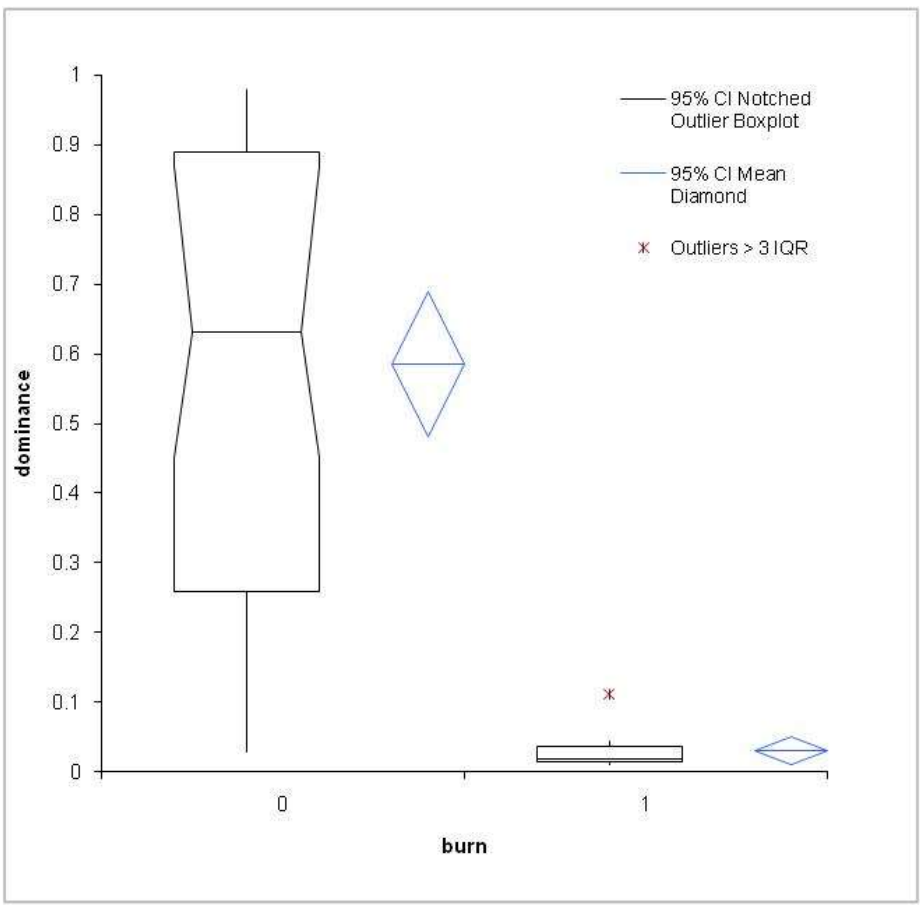


Figure 15. *Bromus tectorum* dominance for 2009 plots comparing unburned (0) and burned (1) conditions.

2009 SUMMARY

The significance of the density and richness of the understory relates to persistence of repeat fire susceptibility. Landscapes with greater plant density provide more fuel. Certain species, such as cheatgrass, have been shown to increase fire risk (Brooks and Matchett 2003; Haubensak et al., 2009). Vegetation community composition and diversity provides a means to assess resilience and susceptibility to wildfires. As the landscape recovers from fire, physical processes affect soil erodiability from wind and water. Erosion inhibits landscape recovery after a fire event. Although there is a correlation between density in terms of biomass, and fuel, greater plant density particularly in terms of grasses and forbs, acts to stabilize soil and reduce erosion. The relationship between vegetation community composition and diversity with fire and the ability to recover post-fire is complex as described below for this study site.

How closely did the post-fire regeneration reflect the unburned plant community diversity?

In 2009 the species composition of burned areas was similar to the unburned areas however vegetation densities, frequency of occurrence, and diversity was different. The relationship between richness and density in both burned and unburned landscapes was positive although the correlation in unburned sites was stronger. The primary differences were found in the upland setting, which had lower density and richness where burned. Within the upland landscape the primary difference was observed where fire had burned shrub canopies. Where shrubs burned, so did the understory vegetation and it did not regenerate to reflect unburned densities within one year post-fire. Nor did the burned uplands return to the unburned richness within one year.

Was there a difference in regeneration in drainages compared with the rest of the landscape?

There was a difference in regeneration in drainages one year post-fire in the burned landscape as compared to unburned areas. Diversity in burned drainages was significantly lower than in unburned drainages, and was almost none. The distribution of the species and abundance in burned drainages was skewed towards a few species unlike unburned drainages, which exhibited a much more even distribution.

Did microsite location afford any advantage towards reflecting the unburned vegetation community?

Plant density was influenced by microsite location within the burned landscape but not in terms of diversity. Under canopy microsites had significantly higher plant densities and richness. There was no difference in diversity or evenness within the burned landscape by microsite.

The vegetation community structure of the unburned landscape, however, had significantly higher plant densities and species richness under the canopy than in interspace microsites. Diversity and evenness between interspace and understory microsites was comparable for both unburned and burned sites.

Where will the landscape be more susceptible to erosion one year post-fire based on these data?

The results reported here indicate that it is more likely erosion would occur in upland areas versus drainages. Upland areas that had burned were found to have nearly half the plant density as in the unburned landscape. The burned upland sites also had significantly lower species richness. However, the difference in cheatgrass predominance between the burned (very low) and unburned (high) landscapes should not be overlooked. Cheatgrass was 35 times denser in the unburned sites than the burned sites overall and 46 times as dense in unburned uplands than burned uplands. The risk of fire recurrence due to cheatgrass regeneration in previously burned sites is relatively low compared to fire risk in unburned areas. However any soil stability benefits from cheatgrass in burned areas would not likely be realized due to the low numbers of plants. Because the overall density in unburned uplands was more than 4.5 times greater than in the burned upland landscape soil stabilizing benefits from sheer number of plants one year post-fire would likely be less until vegetation recovers to a greater extent.

In drainages the plant density in unburned sites was approximately twice that of burned sites. For cheatgrass specifically, the density difference in unburned versus burned sites was 31 times greater. Soil stability benefits from generic plant density would be expected to be greater in drainages than in upland sites, but still would not be comparable to an unburned landscape based solely on plant density.

2010 RESULTS AND DISCUSSION

A total of 80 plots were sampled in the unburned area and a total of 70 plots were sampled in the burned area in 2010 (Table 5). A total of 27 species were identified in plots in 2010 (Table 6).

Basic summary statistics of density and richness are presented in Table 7 for each sampling unit category. Significant differences in density were identified between burned drainage microsites, density in unburned upland microsites, and in richness of unburned drainage microsites.

Unburned plots

Twenty-one species were identified in unburned plots in the 2010 surveys. *Bromus tectorum* (cheatgrass) and *Mentzelia albicaulis* (whitestem blazingstar) were equally abundant. *Chaenactis fremontii* (desert pincushion) was third most abundant. This native forb is found in California, Nevada, Utah and Arizona. The frequency distribution by species in unburned plots is shown in Figure 16.

ANOVA results showed no significant difference in mean plant densities ($F(19, 60) = 0.53$, $p = 0.939$) and richness ($F(19,60) = 1.04$, $p = 0.431$) for plots sampled in unburned areas. Mean plant density in unburned plots was 45.9 ± 45.5 and mean richness was 3.7 ± 1.4. Spearman correlation (Figure 17) showed a positive relationship ($r^2 = 0.19$) between density and richness for unburned plots although it was not significant ($r(78) = 1.73$, $p = 0.088$).

Table 5. Sample distribution for 2010.

	Burned		**Unburned**		
	Drainage	Upland	Drainage	Upland	*total*
Interspace	15	20	20	20	75
Understory	15	20	20	20	75
total	30	40	40	40	*150*

Table 6. Species identified in plots surveyed in 2010. Code = species abbreviation. Treatment = U (found only in unburned plots), B = (found only in burned plots), Both (found in both burned and unburned plots).

Code	Genus	Species	Duration	Growth	Native/exotic	Treatment
ACHY	*Achnatherum*	*hymenoides*	perennial	grass	native	B
BRTE	*Bromus*	*tectorum*	annual	grass	exotic	Both
CABO	*Camissoina*	*boothii*	annual	forb	native	Both
CAWR	*Calycoseris*	*wrightii*	annual	forb	native	U
CHFR	*Chaenactis*	*fremontii*	annual	forb	native	Both
CORA	*Coleogyne*	*ramosissima*	perennial	shrub	native	Both
CRAN	*Cryptantha*	*angustifolia*	annual	forb	native	Both
CRCI	*Cryptantha*	*circumsissa*	annual	forb	native	Both
CRNE	*Cryptantha*	*nevadensis*	annual	forb	native	Both
ERCI	*Erodium*	*cicutarium*	annual	forb	exotic	B
ERDE	*Eriogonum*	*deflexum*	annual	forb	native	U
ERMA	*Eriogonum*	*maculatum*	annual	forb	native	U
ERNI	*Eriogonum*	*nidularum*	annual	forb	native	U
ESGL	*Escholzia*	*glyptosperma*	annual	forb	native	U
GICA	*Gilia*	*cana*	annual	forb	native	Both
LELA	*Lepidium*	*lasiocarpum*	annual	forb	native	B
LILE	*Linum*	*lewisii*	perennial	forb	native	B
MEAL	*Mentzelia*	*albicaulis*	annual	forb	native	Both
MIPA	*Mimulus*	*parryi*	annual	forb	native	Both
NADE	*Nama*	*demissum*	annual	forb	native	U
OXPE	*Oxetheca*	*perfoliata*	annual	forb	native	Both
PHFR	*Phacelia*	*fremontii*	annual	forb	native	Both
POSE	*Poa*	*secunda*	perennial	grass	native	B
PSRA	*Psathyrotes*	*ramosissima*	annual	forb	native	Both
SAOC	*Sanguisorba*	*occidentalis*	perennial	forb	native	B
STEX	*Stephanomeria*	*exiqua*	annual	forb	native	U
VUOC	*Vulpia*	*octoflora*	annual	grass	native	Both

Table 7. Summary statistics (mean ± standard deviation) calculated for each 2010 sample plot category. Values are per 0.25m2 plot.

	Burned				**Unburned**			
	Drainage		Upland		Drainage		Uplan d	
	Density[1]	*Richness*	*Density*	*Richness*	*Density*	*Richness*[1]	*Density*[1]	*Richness*
Interspace	418.5±111.5	3.2±1.3	42.8±80.9	4.1±1.7	192.9±211.3	3.6±1.3	26±59.3	3.7±1.9
Understory	279.5±96.6	2.3±1.4	44.5±67	3.9±1.8	171.8±116.9	2.9±1.3	52.1±57.8	4±1.5

[1] Significant at $\alpha = 0.05$

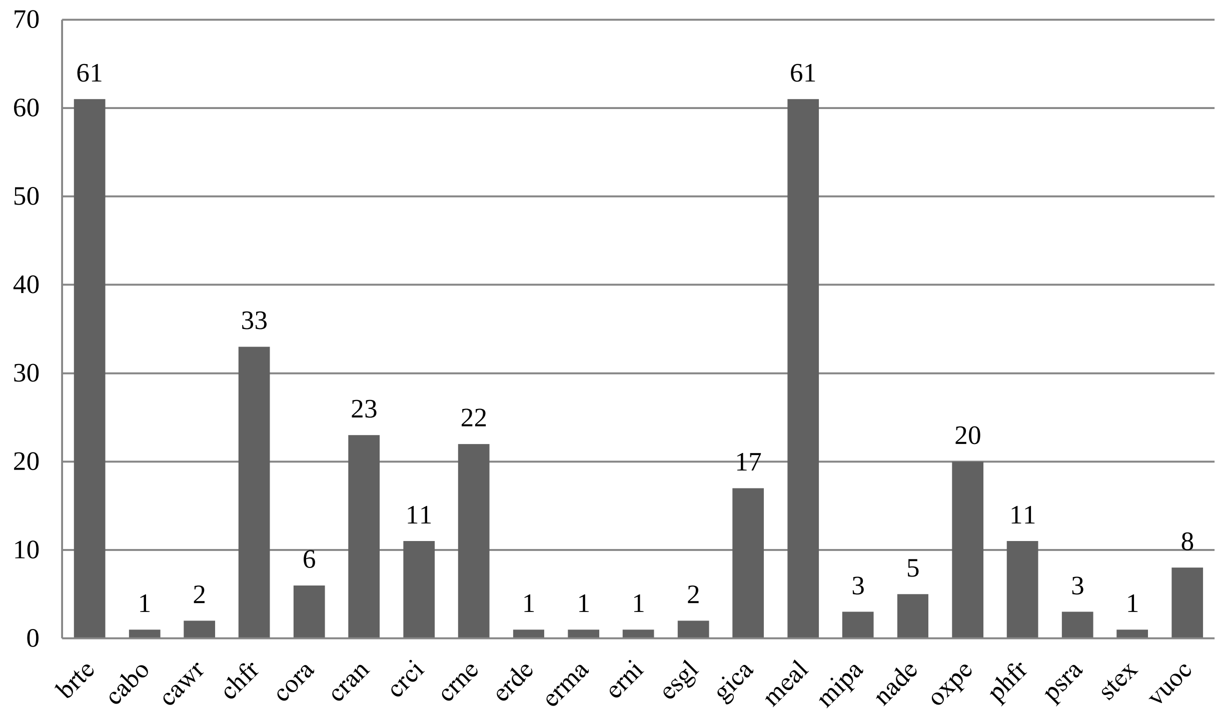


Figure 16. Frequency distribution of abundance of species identified in unburned plots surveyed in 2010. See Table 6 for species code definition.

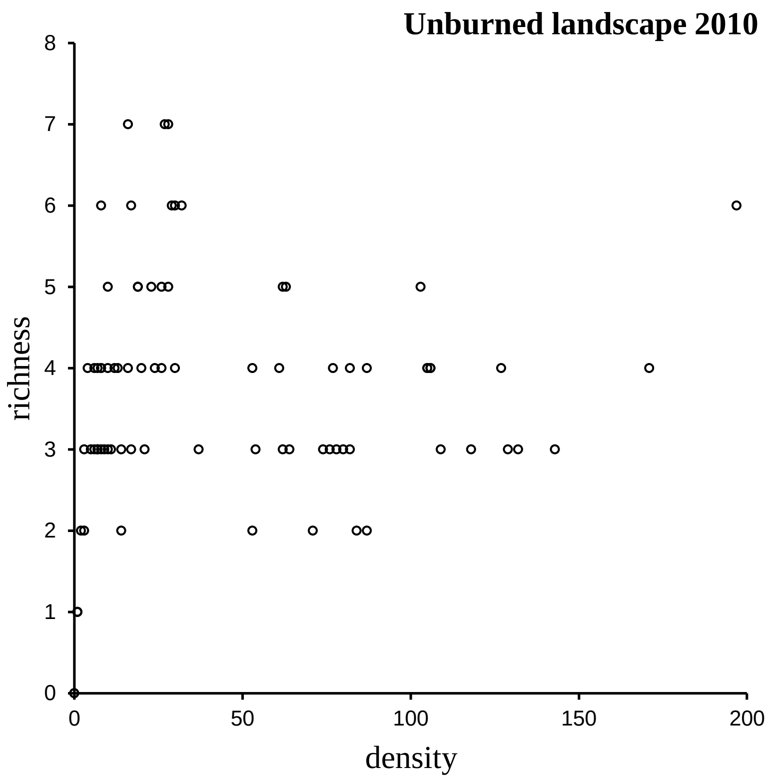


Figure 17. Correlation plot for diversity and richness in unburned plots sampled in 2010.

Plant densities in unburned plots were significantly different between drainages (57.4 ± 46.2/plot) and uplands (34.5 ± 42.3/plot) ($t(78) = 2.32$, $p = 0.023$) but exhibited similar variability ($F(39, 39) = 1.19$, $p = 0.583$). Variability of richness in unburned plots exhibited a significant difference between drainages and upland sites ($F(39, 39) = 0.48$, $p = 0.024$) although richness in drainages (3.7 ± 1.2) was no different than upland sites (3.7 ± 1.7) ($t(78) = -0.08$, $p = 0.939$). In unburned plots results from Shannon-Weiner index showed drainage areas ($\bar{H}' = 0.801$) were significantly less diverse than upland ($\bar{H}' = 1.02$) areas ($t(73) = -2.81$, $p = 0.006$). Simpson's Index results showed that diversity of unburned drainages ($\bar{D} = 0.558$) were significantly different than unburned uplands ($\bar{D} = 0.463$) ($t(74) = 2.31$, $p = 0.024$). Mean evenness for unburned drainages ($J' = 0.638$) and unburned uplands ($J' = 0.759$) was significantly different ($t(73) = -2.50$, $p = 0.015$). The unburned uplands exhibited more equal distribution of plant abundance whereas unburned drainages were skewed towards one or two species.

In the unburned environment sampled, the overall richness of species recorded in unburned uplands and drainages were no different, although drainages exhibited a greater density than uplands. This is reflected in the diversity metrics which account for richness and abundance. In terms of overall diversity however, neither drainages nor uplands would be considered either particularly homogeneous or heterogeneous.

In examining microsite differences for unburned plots, the variability in plant density was significantly different for 'interspace' sites compared to 'understory' sites ($F(39, 39) = 0.14$, $p < 0.0001$). Sites characterized as 'interspace' exhibited significantly more uniformity in density than 'understory' sites although the 95 percent CI included zero (0.00 to 0.23). Mean density was significantly lower in the 'interspace' (16.5 ± 17.1/plot) sites than in the 'understory' sites(75.4 ± 46/plot) (t(78) = -7.59, p < 0.0001). Variability in richness between 'interspace' and 'understory' sites was significantly different (F(39,39) = 2.33, p = 0.014) although mean richness was not significantly different between 'interspace' sites (3.7 ± 1.7/plot) and 'understory' sites (3.7 ± 1.1/plot) (t(78) = -0.08, p = 0.939).

Results from Shannon-Weiner index showed 'interspace' ($\overline{H}'$= 1.039) and 'understory' ($\overline{H}' = 0.791$) microsites had significantly different diversity (t(73) = 3.22, p = 0.002). Simpson's Index results also showed that 'interspace' ($\overline{D}$ = 0.463) and 'understory' ($\overline{D}$ = 0.555) microsites were significantly different (t(74) = -2.22, p = 0.0296). Mean evenness was significantly different between 'interspace' (J'= 0.767) and 'understory' (J'= 0.634) microsites (t(73) = 2.80, p = 0.007). Neither burned nor unburned plots would be considered either particularly diverse or lacking diversity. Although the differences between the two are significant, the practical interpretation is that there is slightly more homogeneity underneath the canopy, whereas the interspace areas presented greater heterogeneity.

Burned plots

Twenty species were identified in burned plots surveyed in 2010 (Figure 18). The native annual forb *Mentzelia albicaulis* (whitestem blazingstar) was the most abundant species identified by threefold over the native shrub blackbrush (*Coleogyne ramosissima*) and the native annual grass *Vulpia octoflora* (sixweeks fescue or sixweeks grass).

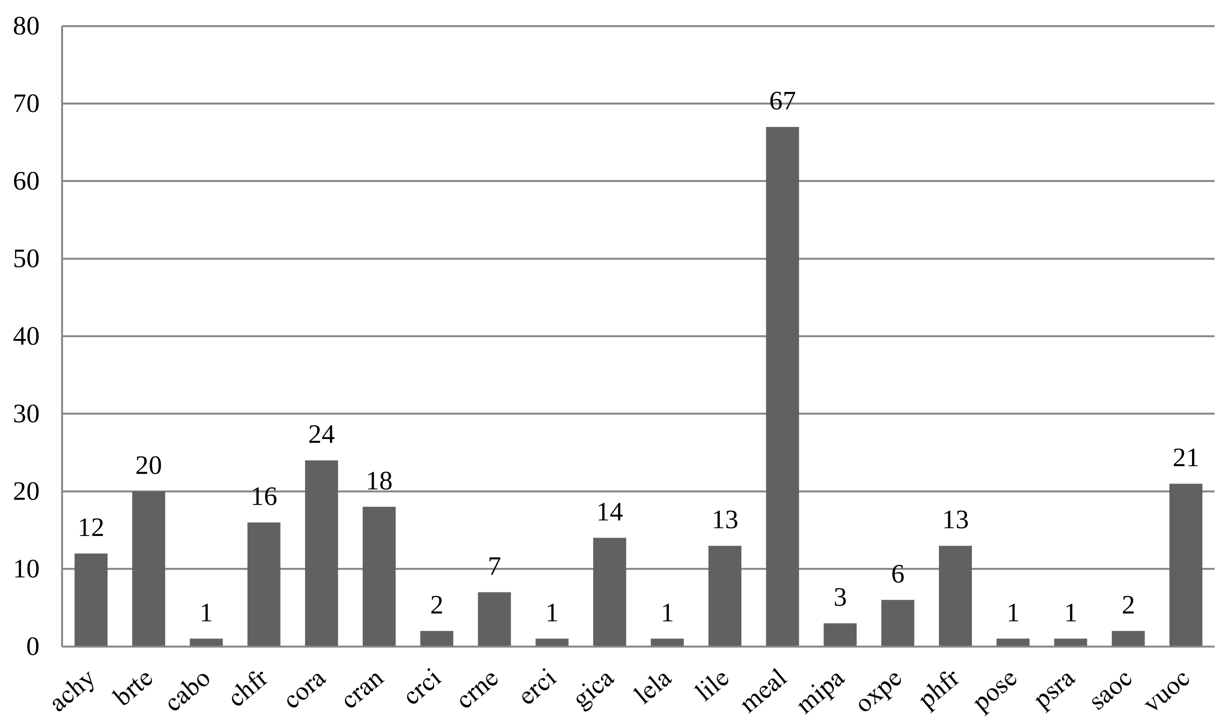


Figure 18. Species identified in burned plots in 2010. Abundance shown above species bar. See Table 6 for species code definition.

ANOVA results showed similar densities (F(19,50) = 0.62, p = 0.875) and richness (F(19,50) = 1.65, p = 0.0796) between plots sampled in unburned areas. Mean plant density in burned plots was 174.5 ± 180.9 and mean richness was 3.5 ± 1.7. Spearman correlation (Figure 19) showed no relationship (r^2 = -0.18) between density and richness for unburned plots (r(68) = -1.51, p = 0.137).

Plant densities in the burned plots were significantly different between drainages (349 ± 124.5/plot) and uplands (43.6 ± 22.1/plot) (t(68) = 12.84, p < 0.0001). The variability of plant density was significantly greater in drainages than upland sites (F(29, 39) = 2.88, p = 0.002). Richness in burned plots was significantly different in drainages (2.8 ± 1.4) than in upland sites (4.0 ± 1.7) (t(68) = -3.16, p = 0.002) There was no difference in variability between drainage and upland sites (F(29, 39) = 0.68, p = 0.277). In burned plots results from Shannon-Weiner index showed drainage areas ($\overline{H}' =$0.07) were significantly less diverse than upland ($\overline{H}' =$0.915) areas (t(60) = -8.37, p < 0.0001). Simpson's Index results showed that diversity of burned drainages ($\overline{D}$ = 0.980) were significantly different than unburned uplands ($\overline{D}$ = 0.539) (t(66) = 9.4, p < 0.0001). Mean evenness for burned drainages (J'= 0.057) and unburned uplands (J'= 0.659) was significantly different (t(60) = -10.17, p < 0.0001). The burned uplands exhibited a more equal distribution of plant abundance whereas unburned drainages were highly skewed towards one or two species.

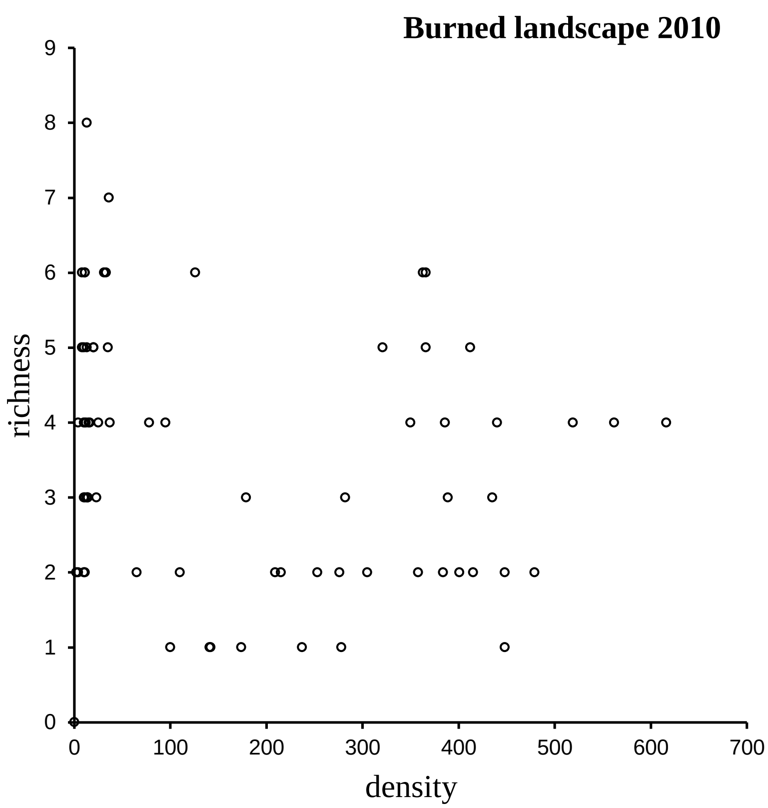


Figure 19. Correlation plot between density and richness for burned plots sampled in 2010.

In examining microsite differences, the variability in plant density was significantly different for 'interspace' sites compared to 'understory' sites ($F(34, 34) = 2.19$, $p = 0.025$). There was less variability under shrub cover than in interspace sites although there was no difference in density of 'interspace' sites (203.8 ± 210.6/plot) than 'understory' sites (145.2 ± 142.4/plot) ($t(68) = 1.36$, $p = 0.177$). There was no significant difference in richness between 'interspace' (3.7 ± 1.6) and 'understory' (3.2 ± 1.8) sites ($t(68) = 1.19$, $p = 0.239$) nor was the variability in plot richness significantly different ($F(34, 34) = 0.81$, $p = 0.5403$). Results from Shannon-Weiner index showed 'interspace' ($\overline{H}'= 0.544$) and 'understory' ($\overline{H}' = 0.638$) microsites in burned areas did not show a significant difference in diversity ($t(60) = -0.65$, $p = 0.516$). Simpson's Index results also showed that 'interspace' ($\overline{D} = 0.728$) and 'understory' ($\overline{D} = 0.726$) microsites were not significantly different ($t(66) = 0.02$, $p = 0.985$). Mean evenness was not significantly different between 'interspace' ($J'= 0.407$) and 'understory' ($J'= 0.447$) microsites ($t(60) = -0.42$, $p = 0.673$).

Comparing burned versus unburned plots

In 2010 approximately two years post fire, the species recorded in burned plots were similar to those in unburned plots with the exception of cheatgrass. Approximately 50 percent of the species reported were shared in both the burned and unburned landscape. Twenty-two percent of species were unique to the burned landscape and just over one-fourth of the reported species were unique to the unburned landscape. Two exotic species were found, cheatgrass and redstem stork's bill (*Erodium cicutarium*). Cheatgrass was identified three times more often in unburned plots than in burned plots. Redstem stork's bill was reported only in one burned plot. The native whitestem blazingstar was highly prevalent across the landscape regardless of having been burned or not. Sixweeks fescue was found nearly three times as often in the burned landscape and blackbrush was recorded four times as often.

By 2010, two years after the fire, the annual native *M. albicaulis* was more than 75 percent dominant in 41 burned plots of which 30 were drainage plots. *M. albicaulis* was 100 percent dominant in seven burned plots. Five different species were more than 50 percent dominant in burned plots and this number was six species in unburned plots. Only blackbrush and whitestem blazingstar were greater than 50 percent dominant in both the burned and unburned landscape. This is important in terms of diversity, which could be considered relevant to fire, soil stability, biomass, and erosion.

Both burned and unburned plots had a minimum of 0 plants recorded. Burned and unburned mean density and richness for drainage and upland plots are presented in Figure 20. Mean plot density by plot type is presented in Figure 21 and Figure 22, respectively. At the landscape level, results showed a significant difference in the mean plant density between burned (174.5 ± 180.9/plot) and unburned (45.9 ± 45.5/plot) plots ($t(148) = -6.14$, $p < 0.0001$) as well as a significant difference in variability of density between burned and unburned plots ($F(79, 69) = 0.06$, $p < 0.0001$). Mean richness of unburned plots was 3.7 ± 1.4 and for burned plots was 3.5 ± 1.7. There was no significant difference in richness or richness variability of unburned and burned plots ($t(148) = 0.74$, $p = 0.46$) and ($F(79, 69) = 0.70$, $p = 0.131$), respectively.

Post-burn drainages exhibited greater density but fewer species than the unburned drainages. The higher density of plants would be expected to have greater soil stability properties. Because the dominance of highly flammable cheatgrass was significantly lower in burned areas and specifically less dominant in drainages (t(44) = 4.48, $p < 0.0001$), the promulgation of fire from cheatgrass would be low until there is a significant change in vegetation characteristics at this site. The upland areas exhibited similar properties regardless of burn status, i.e., greater density but fewer species.

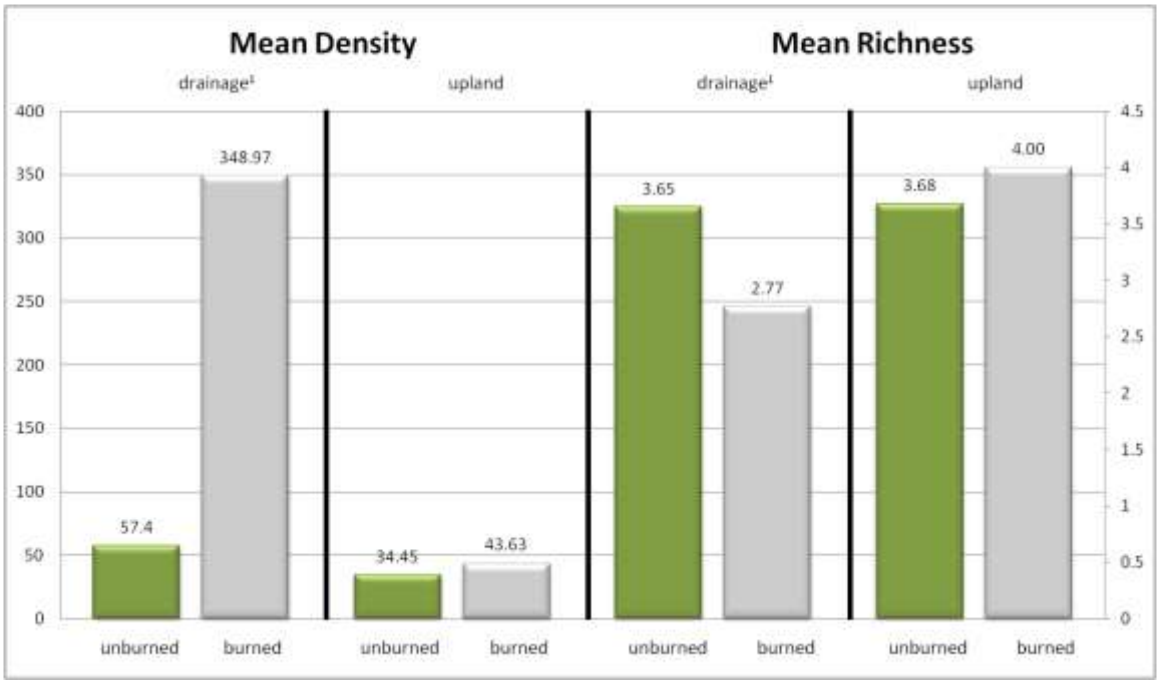


Figure 20. Mean density and mean richness (labeled) for all unburned and burned plots in each geomorphologic setting; drainage or upland (2010). [1] indicates a significant difference.

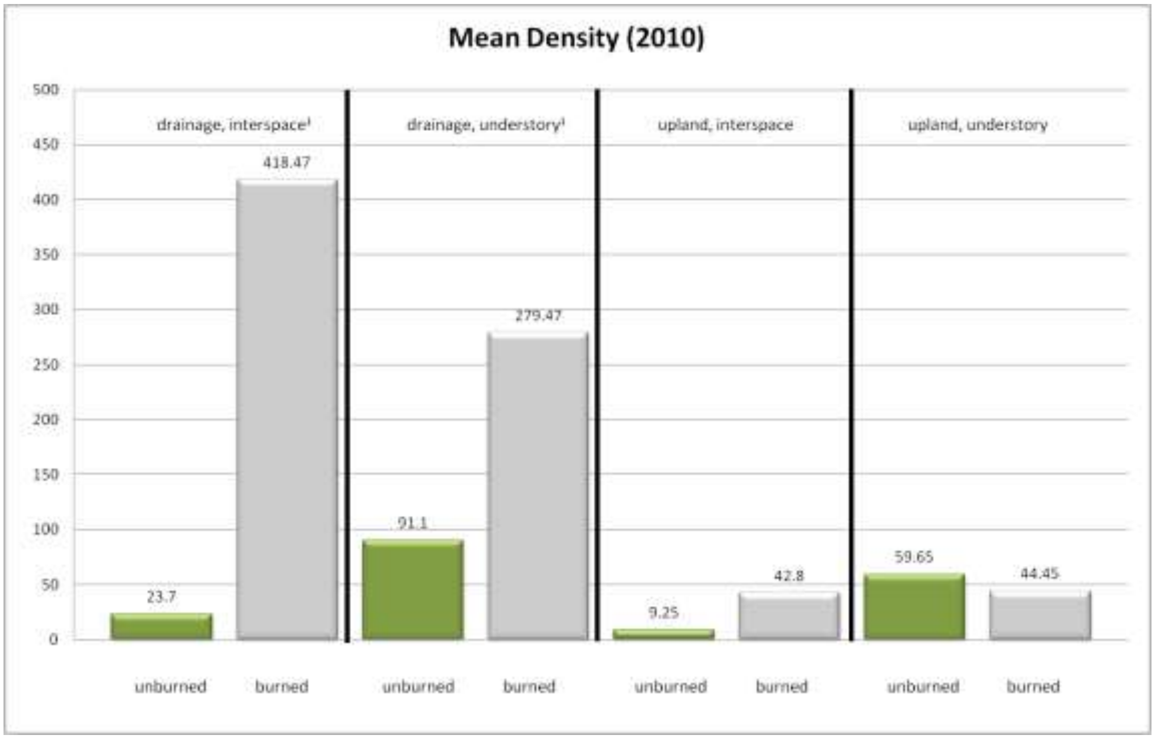


Figure 21. Mean density (labeled) by plot type for 2010. [1] indicates a significant difference.

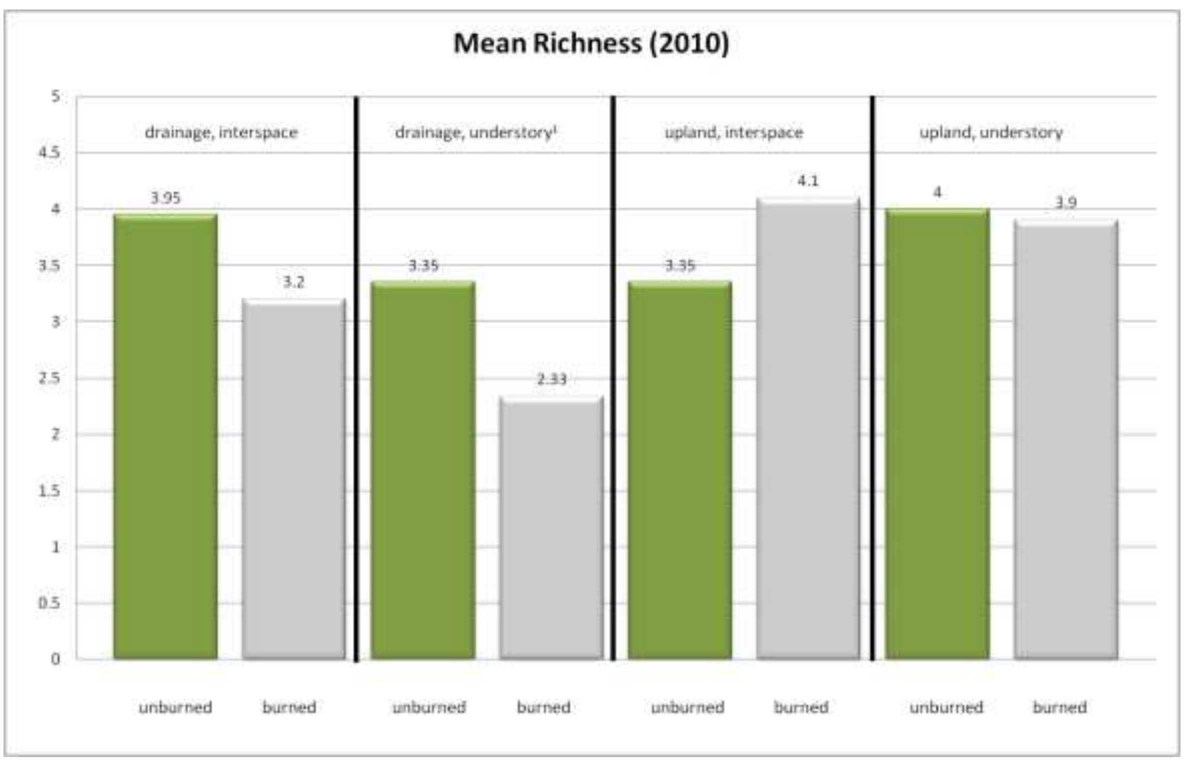


Figure 22. Mean richness (labeled) by plot type for 2010. [1] indicates a significant difference.

Results from Shannon-Weiner index showed unburned areas ($\overline{H}' = 0.906$) were significantly more diverse than burned ($\overline{H}' = 0.588$) areas ($t(135) = 4.02$, $p < 0.0001$). Simpson's Index (D) showed that unburned areas ($\overline{D}$ = 0.511) were significantly more diverse than burned areas ($\overline{D}$ = 0.727) ($t(142) = -5.38$, $p < 0.0001$). Burned area $\overline{D}$ which approaches 1 is indicative of low diversity. Mean evenness for burned (J'= 0.426) and unburned (J'= 0.696) areas was also significantly different ($t(135) = 5.32$, $p < 0.0001$). Diversity and evenness comparisons for burned and unburned, drainage and upland plots, respectively, are presented graphically in Figure 23.

Evaluating 'interspace' microsite plots, there was a significant difference in diversity between unburned ($\overline{D}$ =0.463) and burned ($\overline{D}$ =0.728) plots ($t(68) = -4.87$, $p < 0.0001$) and in evenness ($t(66) = 5.30$, $p < 0.0001$) (unburned J'= 0.767; burned J'= 0.407). Unburned sites between shrubs were more even in species distribution and mean value approached 1. For 'understory' microsite plots, unburned plots ($\overline{D}$ =0.555) were more diverse than burned ($\overline{D}$ =0.726) ($t(72) = -2.93$, $p = 0.005$). Burned (J'= 0.447) and unburned (J'= 0.634) 'understory' microsites showed a difference in evenness ($t(67) = 2.49$, $p = 0.015$) although neither condition represented a distribution that was either even or skewed.

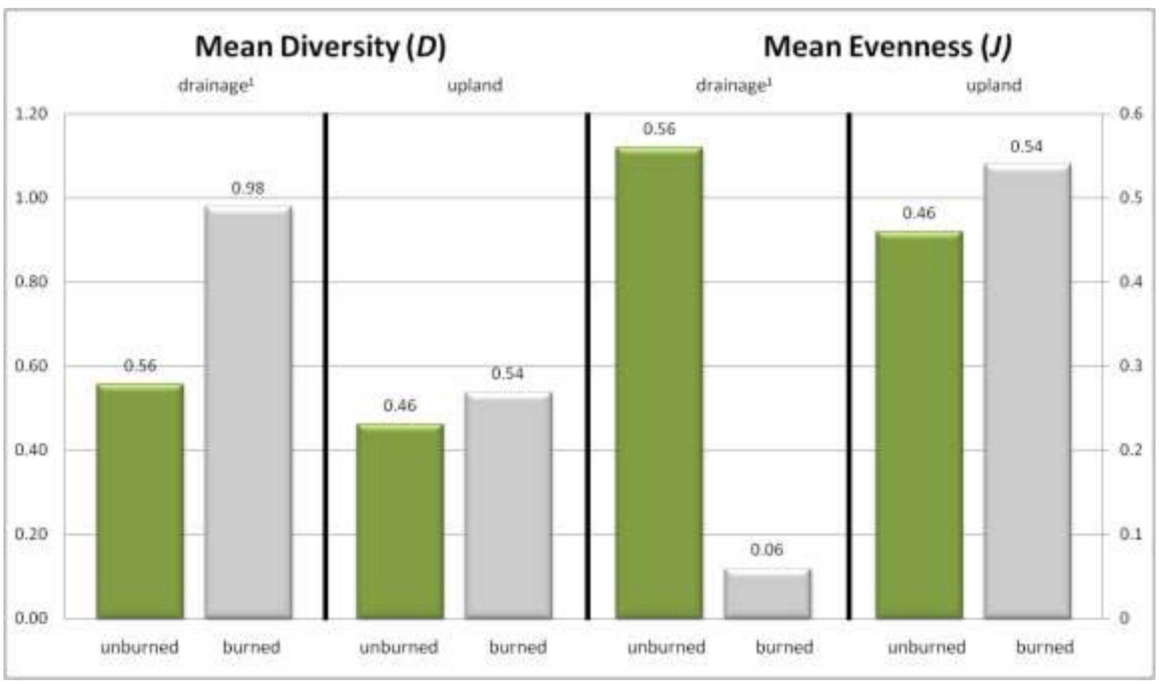


Figure 23. Mean diversity (Simpson's Index) and evenness for drainage and upland locations, comparing burned and unburned plots (2009). [1] indicates a significant difference. Diversity (D) ranges from 0 (infinite diversity) to 1 (no diversity). Evenness (J) ranges from 0 (skewed) to 1 (even).

Drainages

Comparing only those plots in drainages, regardless of microsite, there was a significant difference in densities between burned (349 ± 124.5/plot) and unburned drainage sites (57.4± 46.2/plot) ($t(68) = -13.64$, $p < 0.0001$). In drainages the burned plots and unburned plots returned a significant difference in variability of density ($F(39, 29) = 0.14$, $p < 0.0001$). Mean richness was significantly different ($t(68) = 2.84$, $p = 0.006$) in burned (2.8 ± 1.4) and unburned (3.7 ± 1.2) drainage plots although the variability in richness was not different ($F(39, 29) = 0.67$, $p = 0.233$).

Simpson's Index returned significant differences in diversity between burned ($\overline{D} = 0.98$) and unburned ($\overline{D} = 0.558$) drainages ($t(66) = -13.42$, $p < 0.0001$). Burned drainages ($J' = 0.057$) were skewed and significantly different ($t(61) = 13.11$, $p < 0.0001$) than unburned drainages ($J' = 0.638$). Burned drainages exhibited almost no diversity two years after the fire.

Figure 24 presents the burned and unburned density and richness data for 'interspace' microsite in drainage plots. Mean plant density was significantly lower in unburned, 'interspace' areas of drainages (23.7 ± 20.2/plot) than in burned, 'interspace' areas of drainages (418.5 ± 111.5/plot) ($t(33) = -15.57$, $p < 0.0001$). The variability in plant density was significantly different with burned, 'interspace' sites in drainages exhibiting greater variability than unburned, 'interspace' drainage sites ($F(19, 14) = 0.03$, $p < 0.0001$). Richness between unburned (4.0 ± 1.3/plot) and burned (3.2 ± 1.3/plot) 'interspace' drainage sites was not significantly different ($t(33) = 1.7$, $p = 0.0994$) nor different in variability ($F(19,14) = 0.93$, $p = 0.874$).

Figure 25 presents the burned and unburned density and richness data for 'understory' microsites in drainage plots. There was a significant difference in mean density for plots in 'understory' microsites of drainages between the burned (279.5 ± 96.6) and unburned (91. ± 39.7) plots ($t(33) = -7.91$, $p < 0.0001$). Drainage, 'understory' sites that had burned exhibited a difference in variability of plant density in comparison to unburned, drainage 'understory' sites ($F(19, 14) = 0.17$, $p = 0.0005$). Richness between burned (2.3 ± 1.4/plot) and unburned (3.4 ± 1/plot) 'understory' drainage sites was significantly different ($t(33) = 2.47$, $p = 0.0188$). The variability in richness was not significantly different ($F(19,14) = 0.47$, $p = 0.122$).

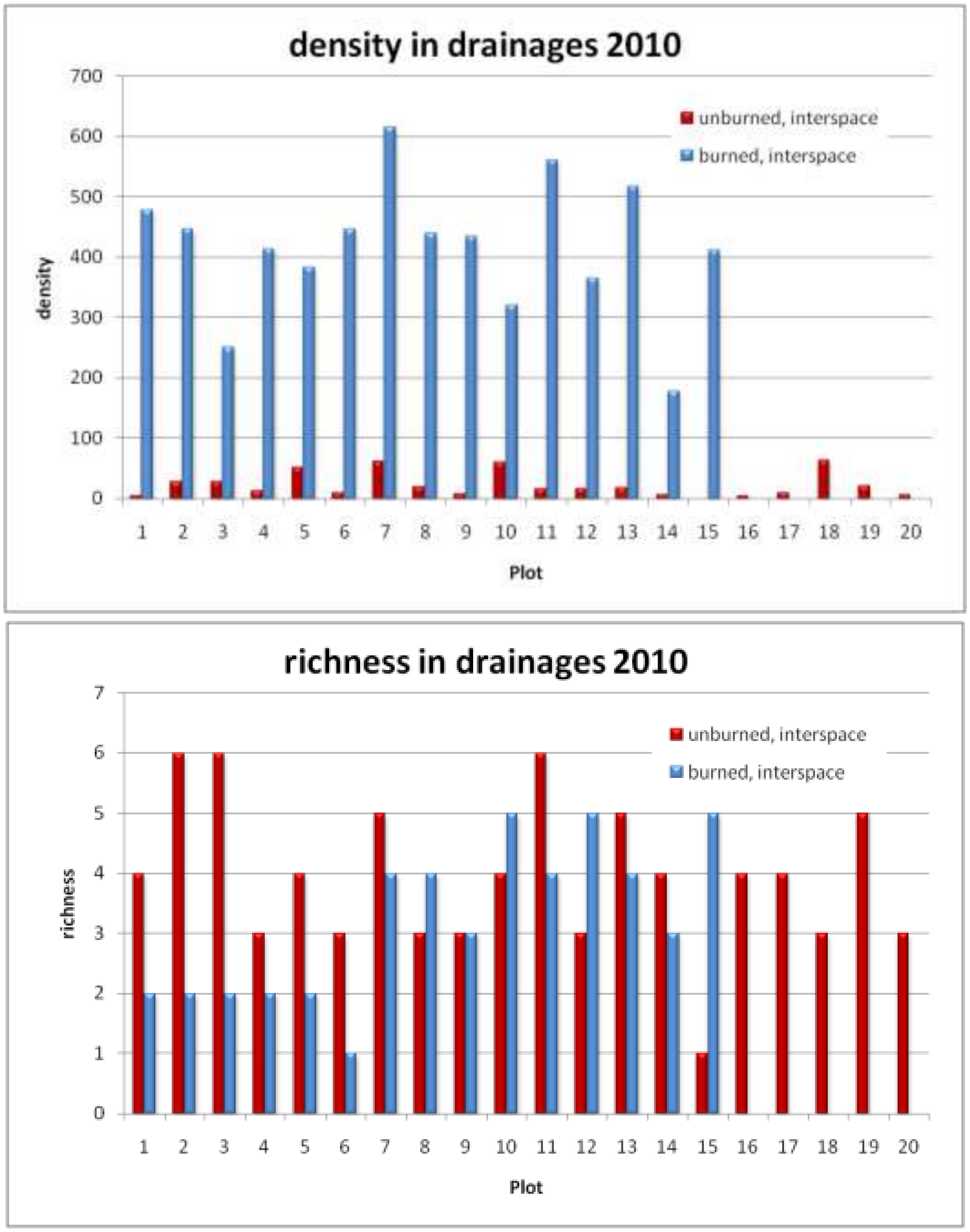


Figure 24. Density and richness for burned and unburned inter-canopy ‘interspace’ drainage sites (2010).

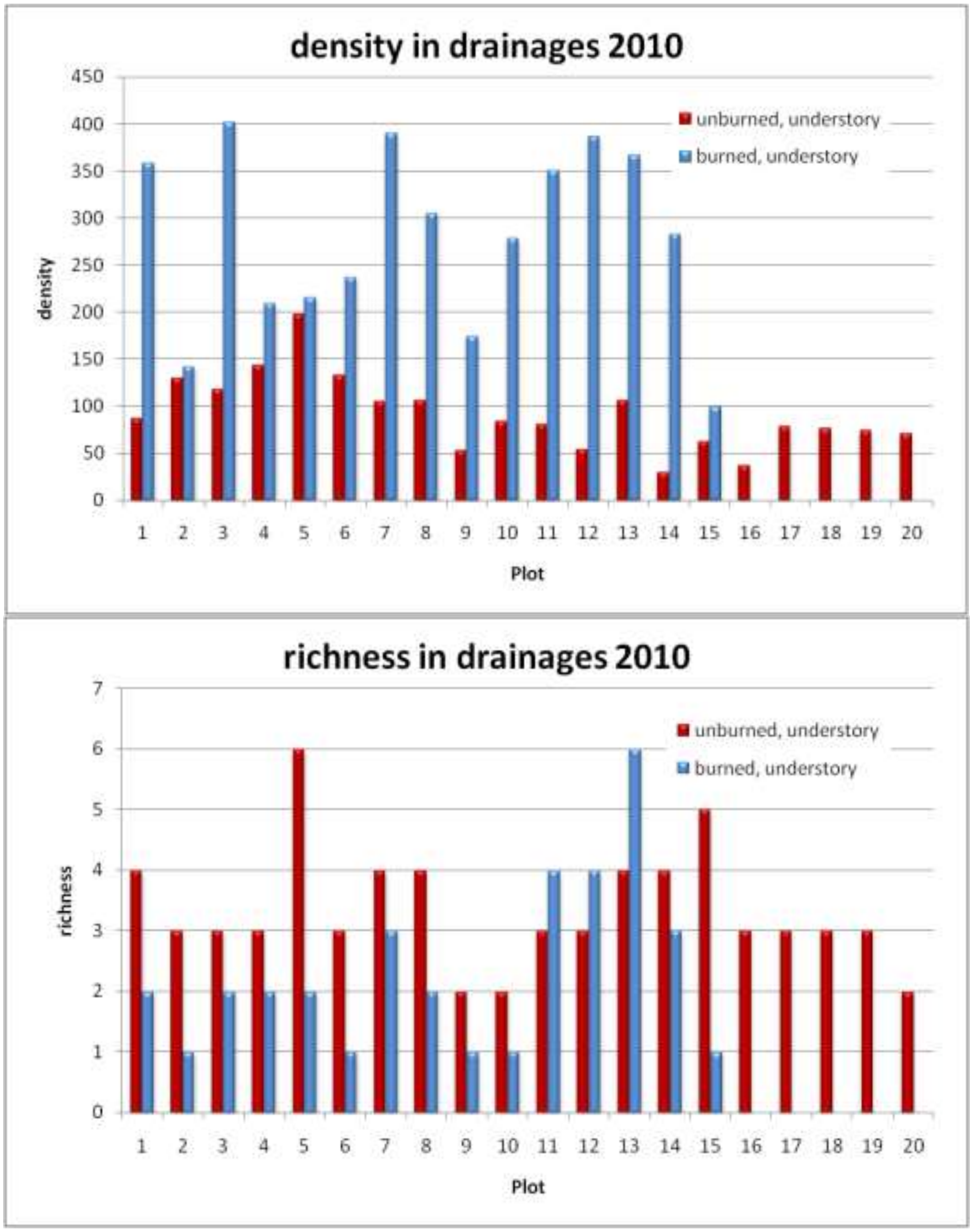


Figure 25. Density and richness data for burned and unburned drainages under the canopy (2010).

Upland sites

Comparing upland plots regardless of microsite, there was no significant difference ($t(78) = -0.69$, $p = 0.495$) in densities between unburned (34.5 ± 42.3/plot) and burned sites (43.6 ± 73.3/plot) and in the variability of density ($F(39,39) = 0.33$, $p = 0.0009$). Mean richness was not significantly different ($t(78) = -0.85$, $p = 0.399$) in burned (4 ± 1.7/plot) and unburned (3.7 ± 1.7/plot) upland plots. Variability in richness was not significantly different ($F(39, 39) = 0.94$, $p = 0.845$).

There was not a significant difference in diversity ($t(74) = -1.49$, $p = 0.1410$) for burned ($\overline{D} = 0.539$) and unburned ($\overline{D} = 0.463$) upland plots. Both unburned ($J' = 0.759$) and burned ($J' = 0.659$) upland plots exhibited similar evenness ($t(72) = 1.73$, $p = 0.0883$).

Figure 26 presents the burned and unburned density and richness data for 'interspace' microsite locations in upland plots. For upland, 'interspace' plots there was no significant difference in density ($t(19.5) = -1.84$, $p = 0.0805$) of burned (42.8 ± 80.9/plot) and unburned (9.3 ± 8.8/plot) plots. There was a difference in variability of density between the two ($F(19, 19) = 0.01$, $p < 0.0001$). Richness between burned (4.1 ± 1.7/plot) and unburned (3.4 ± 2/plot) upland 'interspace' sites was not significantly different ($t(38) = -1.25$, $p = 0.218$) and was not different in variability ($F(19,19) = 1.36$, $p = 0.5102$).

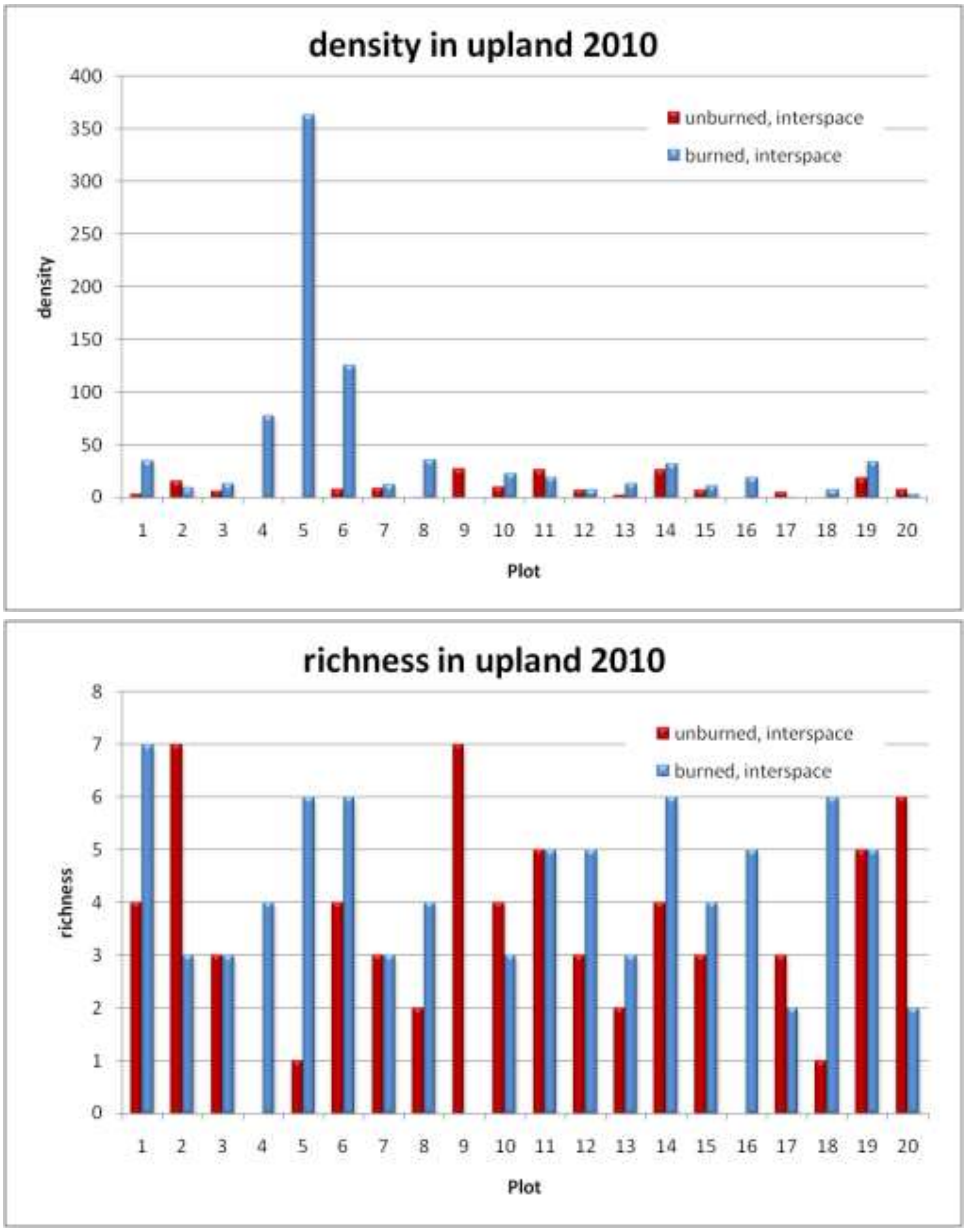


Figure 26. Density and richness for burned and unburned inter-canopy upland sites (2010).

Figure 27 presents the burned and unburned density and richness data for 'upland' microsite plots under the canopy ('under'). Mean density was not significantly different ($t(38) = 0.83$, $p = 0.413$) in unburned (59.7 ± 47.5/plot) than burned (44.5 ± 67/plot) upland 'understory' sites. There was not a significant difference in the variability of density in upland 'understory' plots ($F(19,19) = 0.5$, $p = 0.143$). Richness of unburned upland 'understory' plots (4 ± 1.2/plot) was not significantly different ($t(38) = 0.21$, $p = 0.836$) from burned (3.9 ± 1.8/plot) upland 'understory' sites and the variability between the two was not significantly different ($F(19, 19) = 0.47$, $p = 0.107$).

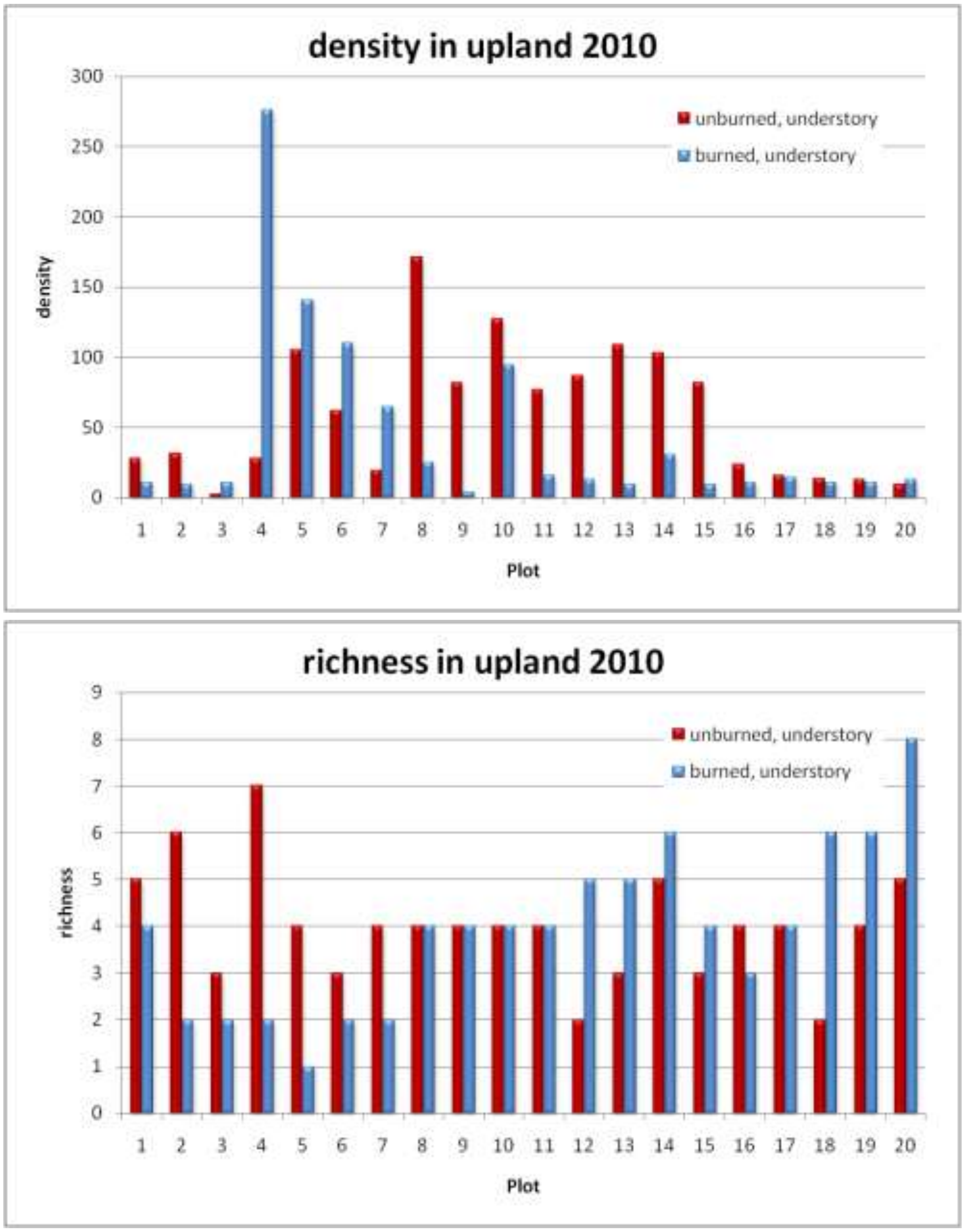


Figure 27. Density and richness data for burned and unburned uplands under the canopy (2010).

Cheatgrass (Bromus tectorum) and Blackbrush (Coleogyne ramosissima) dominance

Two years post-fire, cheatgrass was identified in 81 of 150 plots surveyed. Sixty-nine plots (46%) had no cheatgrass in 2010. It was identified in 61 (76%) unburned plots and in 20 (29%) burned plots. Mean dominance of cheatgrass in unburned plots where identified was 0.377 ± 0.269. For burned plots the mean was 0.032 ± 0.059. The difference in cheatgrass dominance between burned and unburned plots was significantly different where unburned plots had predominantly more cheatgrass than burned plots (t(79) = 5.68, $p < 0.0001$). The dominance distribution for *B. tectorum* in burned and unburned plots is shown in Figure 28.

For plots surveyed in drainages cheatgrass was more predominant in unburned plots (0.26 ± 0.26) than in burned plots (0.002 ± 0.003) (t(68) = 5.52, $p < 0.0001$). In total, 24 of 70 drainage plots had no cheatgrass identified, seven unburned plots had none and 17 burned plots had none. Of the 80 plots surveyed in upland areas, cheatgrass was identified in 28 unburned plots and in only 7 burned plots. Mean dominance in the unburned plots (0.32 ± 0.311) was significantly higher than in burned plots (0.015 ± 0.045) (t(78) = 6.05, $p < 0.0001$).

Blackbrush was identified in 30 of 150 plots surveyed. Mean blackbrush dominance in unburned plots where it occurred (0.242 ± 0.38) was not significantly different than burned plots where it was recorded (0.140 ± 0.16) (t(28) = 1.04, $p = 0.309$). Blackbrush was not identified in any unburned drainage plots and was found in three burned drainage plots. It was identified in 24 burned upland plots and in six unburned upland plots.

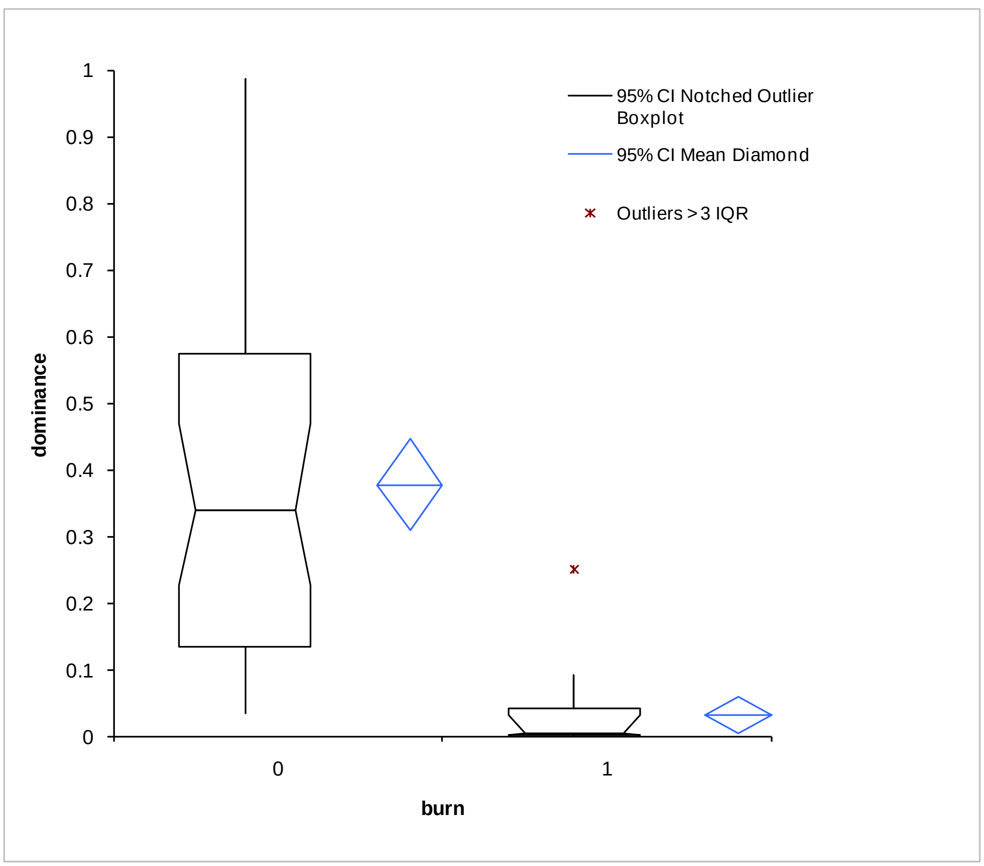


Figure 28. *Bromus tectorum* dominance for 2010 plots comparing unburned (0) and burned (1) conditions.

2010 SUMMARY

Density and richness of the understory has the potential to impact repeat fire susceptibility, which would be anticipated to increase with time assuming the vegetation community regenerates post-fire. Regardless of time interval post-fire, landscapes with greater plant density provide more fuel. Certain species, such as cheatgrass, increase fire risk due to life history traits, i.e., early germination and senescence in advance of summer monsoonal storms that are known to ignite fires via lightning strikes. As discussed in earlier sections, vegetation community diversity is one means to assess resilience and susceptibility to wildfires. As the landscape undergoes revegetation, physical processes affected by fire make the soil more prone to erosion from wind and water. Erosion inhibits the ability of a landscape to heal post-fire. Although there is a correlation between density in terms of biomass, and fuel, greater plant density particularly in terms of grasses and forbs, act to stabilize soil. The relationship between vegetation community composition and diversity with fire and the ability to recover post-fire is complex. This relationship also changes with increasing time, and not necessarily in a linear fashion. The spatial landscape pattern for 2010 was not analyzed due to insufficient sample size.

How closely did the post-fire regeneration reflect the unburned plant community diversity?

In 2010, two years post-fire, the species composition of burned areas was less similar to the unburned areas than observed in 2009. Vegetation densities, species composition, richness, and diversity differed between burned and unburned sites. Within burned and unburned sites neither was found to exhibit a significant relationship between richness and density. The primary differences were found between the geomorphic classes. Burned drainages were six-times denser than unburned drainages while plant densities in burned and unburned uplands did not differ. The density differences in drainages were observed at the microsite level as well. Richness was lower in burned than unburned drainages but was not different in uplands.

Burned drainages exhibited an eight-fold higher vegetation density over burned uplands. However, the upland landscape had greater richness than drainages. From a diversity perspective, drainages exhibited almost no diversity and upland sites were significantly more diverse than drainages. Upland sites also showed more even species distribution.

Was there a difference in regeneration in drainages compared with the rest of the landscape?

Two years post-fire showed large differences in drainages in particular. Density and richness was much greater in burned drainages than unburned drainages whereas there were no differences in these metrics for uplands. Both microsites (interspace and understory) within burned drainages showed consistently greater plant density than unburned drainages. Richness of burned drainages was consistently lower but only significantly so for understory microsites. Diversity in burned drainages was significantly lower than in unburned drainages, and was almost none. The distribution of species and abundance in burned drainages was skewed towards one or two species, namely Indian ricegrass > blackbrush > cheatgrass. The first two species are natives. While significantly different from skewed, unburned drainages were not entirely even in distribution.

Did microsite location afford any advantage towards reflecting the unburned vegetation community?

Microsite did not appear to afford any advantage in terms of vegetation re-establishment or community diversity. Neither plant density nor richness varied with microsite within the burned landscape. There was also no difference in diversity or evenness within the burned landscape by microsite.

These results indicate that two years post-fire the burned landscape has not regained a similar vegetation community structure of the unburned landscape. The unburned landscape had significantly higher plant densities. However as in the burned landscape, species richness within unburned microsites did not differ. Diversity and evenness of unburned microsites were greater in interspace microsites compared with understory microsites.

Where will the two year post-fire landscape be more susceptible to erosion based on these data?

Plant densities were found to be consistently greater in the burned than in the unburned landscape. Within the burned landscape it is more likely erosion would occur in upland areas versus drainages based solely on plant density. Upland areas that had burned were found to have 12 percent of the density of burned drainages. Species richness was no different between the burned and unburned landscape. However, the difference in cheatgrass predominance between the burned (very low) and unburned (high) landscapes should not be overlooked. Cheatgrass was 18 times denser in the unburned sites than the burned sites overall and nearly 30 times as dense in unburned uplands than burned uplands. The risk of fire recurrence due to cheatgrass regeneration in previously burned sites is relatively low compared to fire risk in areas previously unburned. However any soil stability benefits from cheatgrass in burned areas would not likely be realized. Because the overall density in unburned uplands was nearly 3 times greater than in the burned upland landscape, soil stabilizing benefits from sheer number of plants two years post-fire would likely be less until vegetation recovers to a greater extent.

In drainages the plant density in unburned sites more closely approximated that of burned sites and was less than twice as great. For cheatgrass specifically, the density difference in unburned versus burned sites was 15 times greater for the unburned drainages. Soil stability benefits from generic plant density would be expected to be greater in drainages than in upland sites, but still would not be comparable to an unburned landscape based solely on plant density.

2011 RESULTS AND DISCUSSION

A total of 80 plots were sampled in the unburned area and a total of 80 plots were sampled in the burned area in 2011 (Table 8). A total of 26 species were identified in plots in 2011 (Table 9).

Table 8. Sample plot distribution for 2011.

	Burned		**Unburned**		
	Drainage	Upland	Drainage	Upland	*total*
Interspace	20	20	20	20	80
Understory	20	20	20	20	80
total	40	40	40	40	*160*

Table 9. Species identified in plots surveyed in 2011. Code = species abbreviation. Treatment = U (found only in unburned plots), B = (found only in burned plots), Both (found in both burned and unburned plots).

Code	**Genus**	**Species**	**Duration**	**Growth**	**Native/ exotic**	**Treatment**
ACHY	*Achnatherum*	*hymenoides*	perennial	grass	native	B
AMTE	*Amblystegium*	*tenax*		moss	native	U
BRTE	*Bromus*	*tectorum*	annual	grass	exotic	Both
CAWR	*Calycoseris*	*wrightii*	annual	forb	native	B
CHFR	*Chaenactis*	*fremontii*	annual	forb	native	Both
CORA	*Coleogyne*	*ramosissima*	perennial	shrub	native	Both
CRAN	*Cryptantha*	*angustifolia*	annual	forb	native	Both
CRCI	*Cryptantha*	*circumsissa*	annual	forb	native	Both
CRNE	*Cryptantha*	*nevadensis*	annual	forb	native	Both
CRUT	*Cryptantha*	*utahensis*	annual	forb	native	Both
DEPI	*Descurainia*	*pinnata*	annual	forb	native	Both
ERCI	*Erodium*	*cicutarium*	annual	forb	exotic	Both
ERDE	*Eriogonum*	*deflexum*	annual	forb	native	B
ERMA	*Eriogonum*	*maculatum*	annual	forb	native	Both
GICA	*Gilia*	*cana*	annual	forb	native	Both
IPPO	*Ipomopsis*	*polycladon*	annual	forb	native	U
LELA	*Lepidium*	*lasiocarpum*	annual	forb	native	Both
LILE	*Linum*	*lewisii*	perennial	forb	native	B
MEAL	*Mentzelia*	*albicaulis*	annual	forb	native	Both
MIPA	*Mimulus*	*parryi*	annual	forb	native	Both
OXPE	*Oxetheca*	*perfoliata*	annual	forb	native	Both
PHFR	*Phacelia*	*fremontii*	annual	forb	native	Both
POSE	*Poa*	*secunda*	perennial	grass	native	U
PSRA	*Psathyrotes*	*ramosissima*	annual	forb	native	B
SAOC	*Sanguisorba*	*occidentalis*	perennial	forb	native	B
VUOC	*Vulpia*	*octoflora*	annual	grass	native	Both

Basic summary statistics of density and richness are presented in Table 10 for each sampling unit category. Significant differences in density were identified between burned upland microsites, density and richness in unburned upland and drainage microsites, respectively.

Unburned plots

Twenty species were identified in unburned plots in the 2011 surveys. *Bromus tectorum* (cheatgrass) was the most abundant followed by *Vulpia octoflora* (sixweeks fescue or grass). *Chaenactis fremontii* (desert pincushion) and *Gilia cana* (showy gilia) were third most abundant. Showy gilia is native to California and Nevada and like desert pincushion grows in the open on gravelly and sandy soils. Almost as frequently recorded was *Cryptantha angustifolia* (Panamint catseye) which is native in desert scrubland. The frequency distribution by species in unburned plots is shown in Figure 29.

Table 10. Summary statistics (mean ± standard deviation) calculated for each 2011 sample plot category. Values are per 0.25m^2 plot.

	Burned				**Unburned**			
	Drainage		Upland		Drainage		Upland	
	Density	*Richness*	*Density*[1]	*Richness*	*Density*[2]	*Richness*[2]	*Density*[2]	*Richness*[2]
Interspace	30.6±23.7	4.5±1.4	64.1±65	5.7±1.8	20.6±24.4	2.0±1.1	7.2±10	1.9±1.7
Understory	29.6±28.1	3.8±1.3	29.1±26.9	4.8±1.9	188.1±102.6	3.3±1.7	66.4±50.1	5.4±1.8

[1] Significant at α=0.05

[2] Significant at α=0.01

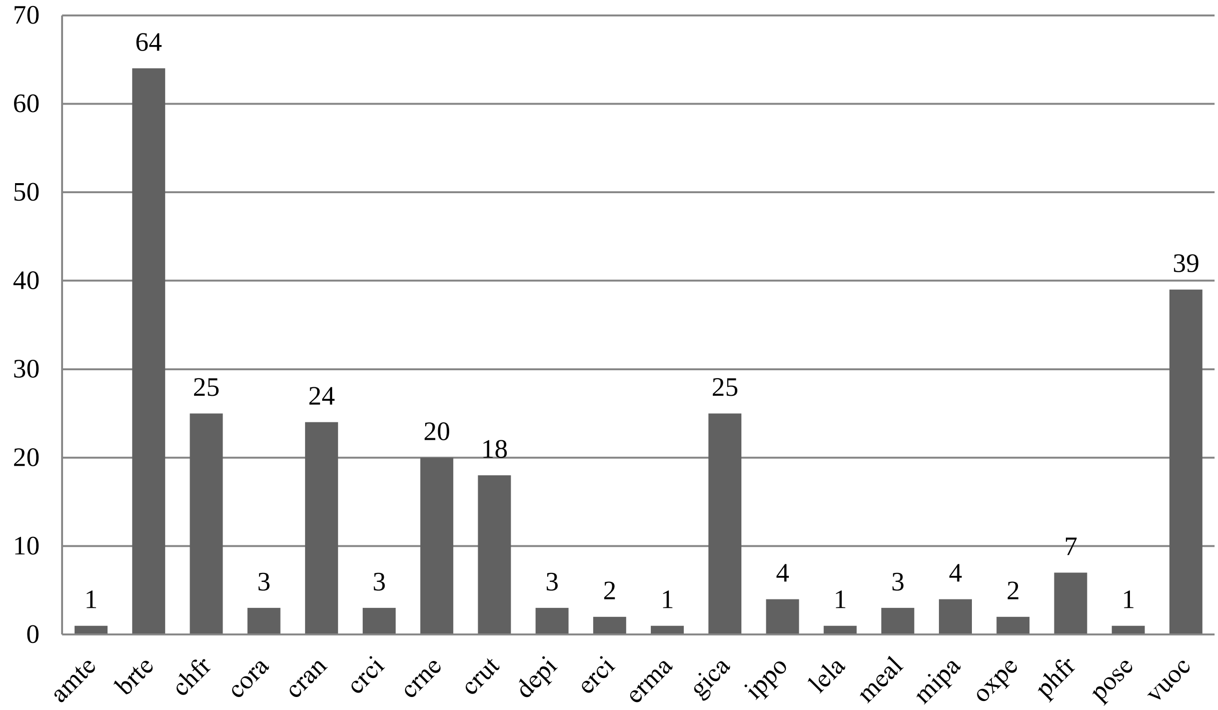


Figure 29. Frequency distribution of abundance of species identified in unburned plots surveyed in 2011. See Table 9 for species code definition.

ANOVA results showed no significant difference in mean plant densities (F(19, 60) = 0.42, p = 0.981) and richness (F(19,60) = 0.67, p = 0.8351) for plots sampled in unburned areas. Mean plant density in unburned plots was 70.6 ± 91.92 and mean richness was 3.1 ± 2.13. Spearman correlation (Figure 30) showed a significant positive relationship (r^2 = 0.53) between density and richness for unburned plots (r(78) = 5.51, p < 0.0001).

Plant densities in unburned plots were significantly different (t(78) = 3.51, p = 0.0007) between drainages (104.3 ± 112.3/plot) and uplands (36.8 ± 46.6/plot) and exhibited similar variability (F(39, 39) = 5.8, p < 0.0001). Variability of richness in unburned plots exhibited a significant difference between drainages and upland sites (F(39, 39) = 0.39, p = 0.0039). Drainages (2.6 ± 1.5) were less rich (t(78) = -2.27, p = 0.0262) than upland sites (3.7 ± 2.5). In unburned plots results from Shannon-Weiner index showed drainage areas ($\overline{H}'$ =0.456) were significantly less diverse (t(56) = -6.67, p < 0.0001) than upland ($\overline{H}'$ =1.09) areas. Simpson's Index results showed that diversity of unburned drainages ($\overline{D}$ = 0. 828) were significantly different than unburned uplands ($\overline{D}$ = 0.508) (t(72) = 5.89, p < 0.0001). Mean evenness for unburned drainages (J'= 0.418) and unburned uplands (J'= 0.746) was significantly different (t(56) = -5.2, p < 0.0001). The unburned uplands exhibited more equal distribution of plant abundance whereas unburned drainages were skewed towards one or two species.

In examining microsite differences in unburned plots, the variability in plant density was significantly different for 'interspace' sites compared to 'understory' sites (F(39, 39) = 0.04, p < 0.0001). Mean density was significantly lower (t(78) = -6.98, p < 0.0001) in the

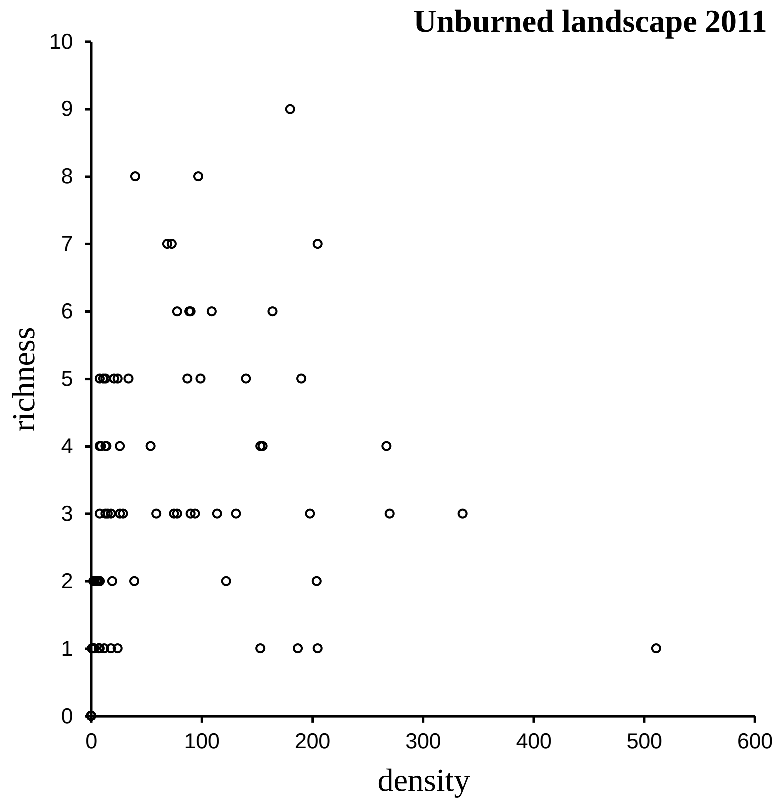


Figure 30. Correlation plot for diversity and richness in unburned plots sampled in 2011.

'interspace' (13.9 ± 3.11/plot) sites than in the 'understory' sites (127.2 ± 100.7/plot). Variability in richness between 'interspace' and 'understory' sites was significantly different (F(39,39) = 0.48, p = 0.0259). Mean richness was significantly different (t(78) = -6.10, p < 0.0001) between 'interspace' sites (1.9 ± 1.4/plot) and 'understory' sites (4.3 ± 2.0/plot).

Results from Shannon-Weiner index showed 'interspace' ($\overline{H}'$ = 0.711) and 'understory' ($\overline{H}' =$ 0.829) microsites were not significantly different in diversity (t(56) = -0.91, p = 0.34). Simpson's Index results also showed that 'interspace' ($\overline{D}$ = 0.734) and 'understory' ($\overline{D}$ = 0.622) microsites were not significantly different (t(72) = 1.75, p = 0.084). Mean evenness was significantly different (t(56) = 2.15, p = 0.036) between 'interspace' (J'= 0.689) and 'understory' (J'= 0.526) microsites.

Burned plots

Twenty-three species were identified in burned plots surveyed in 2011 (Figure 31). The native annual grass *Vulpia octoflora* (sixweeks fescue or sixweeks grass) was the most abundant species identified followed by the exotic species *Bromus tectorum* (cheatgrass) and *Chaenactis fremontii* (desert pincushion), a native species. All other species were less frequently observed.

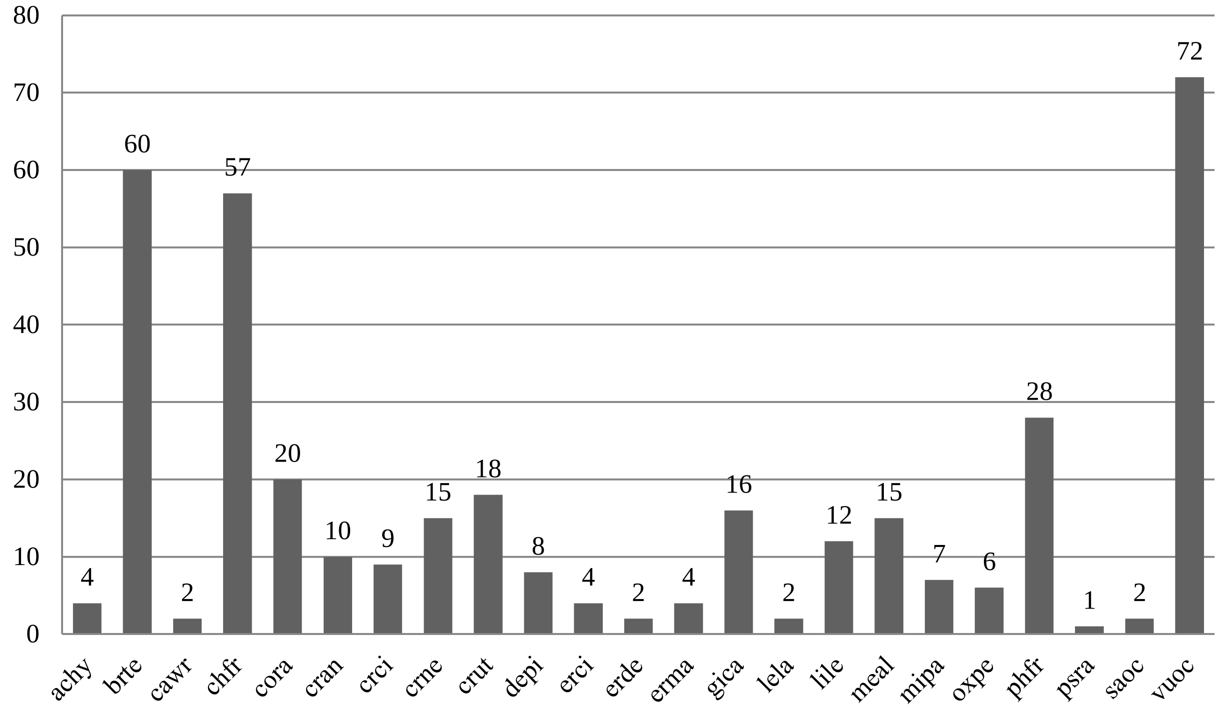


Figure 31. Species identified in burned plots in 2011. Abundance shown above species bar. See Table 9 for species code definition.

ANOVA results showed similar densities (F(19,60) = 1.41, p = 0.159) and richness (F(19,60) = 1.19, p = 0.298) between plots sampled in burned areas. Mean plant density in burned plots was 38.3 ± 41.7 and mean richness was 4.7 ± 1.76. Spearman correlation (Figure 32) showed a significant and positive relationship (r^2 = 0.43) between density and richness for burned plots (r(78) = 4.25, p < 0.0001).

Plant densities in the burned plots were not significantly different (t(78) = -1.79, p = 0.077) between drainages (30.1 ± 25.7/plot) and uplands (46.6 ± 52.2/plot). The variability of plant density was significantly greater in upland sites than drainage (F(39, 39) = 0.24, p < 0.0001). Richness in burned plots was significantly different (t(78) = -2.79, p = 0.007) in drainages (4.2 ± 1.4) than in upland sites (5.2 ± 1.9). There was no difference in variability between drainage and upland sites (F(39, 39) = 0.53, p = 0.053). In burned plots results from Shannon-Weiner index showed drainage areas ($\bar{H}' =$0.998) were no more or less diverse (t(76) = -0.32, p = 0.753) than upland ($\bar{H}' =$1.024) areas. Simpson's Index results showed that diversity of burned drainages ($\bar{D}$ = 0.478) was not significantly different (t(78) = -0.65, p = 0.52) than burned uplands ($\bar{D}$ = 0. 506). Mean evenness for burned drainages (J'= 0.726) and unburned uplands (J'= 0. 396) was not significantly different (t(76) = 1.91, p = 0.06). The burned uplands exhibited a similar distribution of plant abundance to burned drainages.

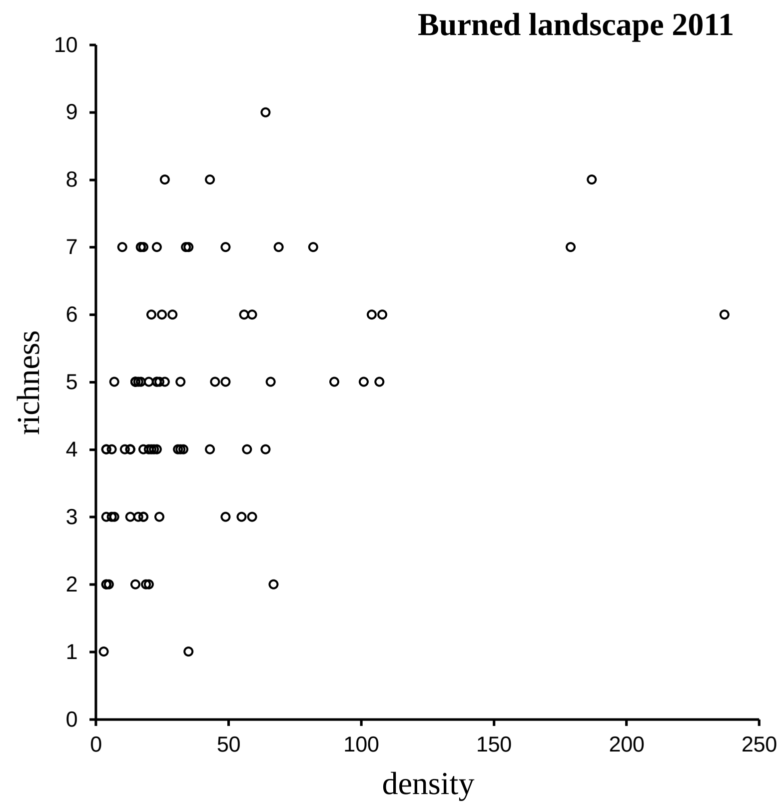


Figure 32. Correlation plot between density and richness for burned plots sampled in 2011.

In examining microsite differences, the variability in plant density was significantly different for ‘interspace’ sites compared to ‘understory’ sites (F(39, 39) = 3.55, p = 0.0001). There was less variability in understory than in interspace sites although there was no difference (t(78) = 1.96, p = 0.053) in density of ‘interspace’ sites (47.3 ± 51.2/plot) than ‘understory’ sites (29.4 ± 27.2/plot). There was a significant difference (t(78) = 2.08, p = 0.041) in richness between ‘interspace’ (5.1 ± 1.7) and ‘understory’ (4.3 ± 1.7) sites although the variability in plot richness was not significantly different (F(39, 39) = 1.02, p = 0.94). Results from Shannon-Weiner index showed ‘interspace’ ($\overline{H}'$= 0.954) and ‘understory’ ($\overline{H}' = 1.07$) microsites in burned areas did not show a significant difference in diversity (t(76) = -1.41, p = 0.162). Simpson’s Index results also showed that ‘interspace’ ($\overline{D}$ = 0.518) and ‘understory’ ($\overline{D}$ = 0.465) microsites were not significantly different (t(78) = 1.23, p = 0.224). Mean evenness was significantly different (t(76) = -2.93, p = 0.0045) between ‘interspace’ (J'= 0.62) and ‘understory’ (J'= 0.748) microsites. ‘Interspace’ microsites exhibited a more even distribution of species than ‘understory’ sites although the absolute counts along the 0-1 scale would only be considered more even than not, and certainly not truly an even distribution.

Comparing burned versus unburned plots

Sixty-five percent of the species recorded in 2011, approximately three years post-fire, were reported in both burned and unburned areas. Twenty-three percent of the species recorded were found only in the burned landscape and approximately 11 percent were reported only in unburned areas. Two exotic species, cheatgrass and redstem stork's bill, were reported in both burned and unburned plots. Sixweeks fescue and desert pincushion were both reported nearly twice as often in unburned areas than burned. The prevalence of cheatgrass was nearly equal in both burned and unburned landscapes.

Burned drainages were much less dense but exhibited higher species richness than unburned drainages. Although the upland area plant density was no different regardless of burn status, the burned uplands exhibited higher richness than unburned areas. The understory microsites in unburned drainages exhibited six times the density of unburned drainages and more than twice the density in unburned uplands. The reverse held true for interspace microsites, where burned areas had much higher recorded plant densities and richness. In terms of overall metrics of diversity, unburned drainages were less diverse than burned, and unburned drainages exhibited almost no diversity. Burned drainages also exhibited a more even distribution of species richness than unburned drainages. Upland areas were equally homogeneous in terms of diversity and both burned and unburned uplands exhibited a more even distribution of species, although unburned areas significantly more so.

Unburned plots had a minimum of zero plants recorded while burned plots had a minimum of three. Burned and unburned mean density and richness results for drainages and uplands are presented in Figure 33. Mean plot density by plot type is presented in Figure 34 and Figure 35, respectively. At the landscape level, results showed a significant difference ($t(158) = 2.86$, $p = 0.0049$) in the mean plant density between burned (70.6 ± 91.9/plot) and unburned (38.3 ± 41.7/plot) plots as well as a significant difference in variability of density between burned and unburned plots ($F(79, 79) = 4.86$, $p < 0.0001$). Mean richness of unburned plots was 3.1 ± 2.1 and for burned plots was 4.7 ± 1.8. There was a significant difference in richness of unburned and burned plots ($t(158) = -5.03$, $p < 0.0001$). Richness variability within plots was not different ($F(79, 79) = 1.46$, $p = 0.092$).

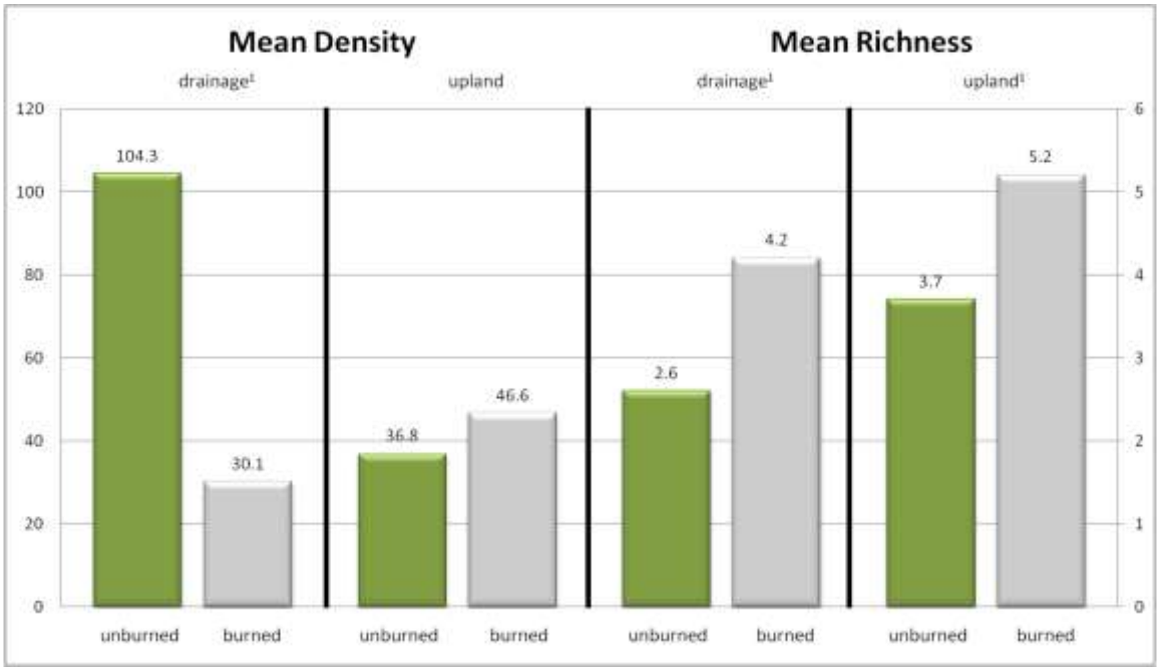


Figure 33. Mean density and mean richness (labeled) for all unburned and burned plots in each geomorphic setting, drainage or upland (2011). [1] indicates a significant difference.

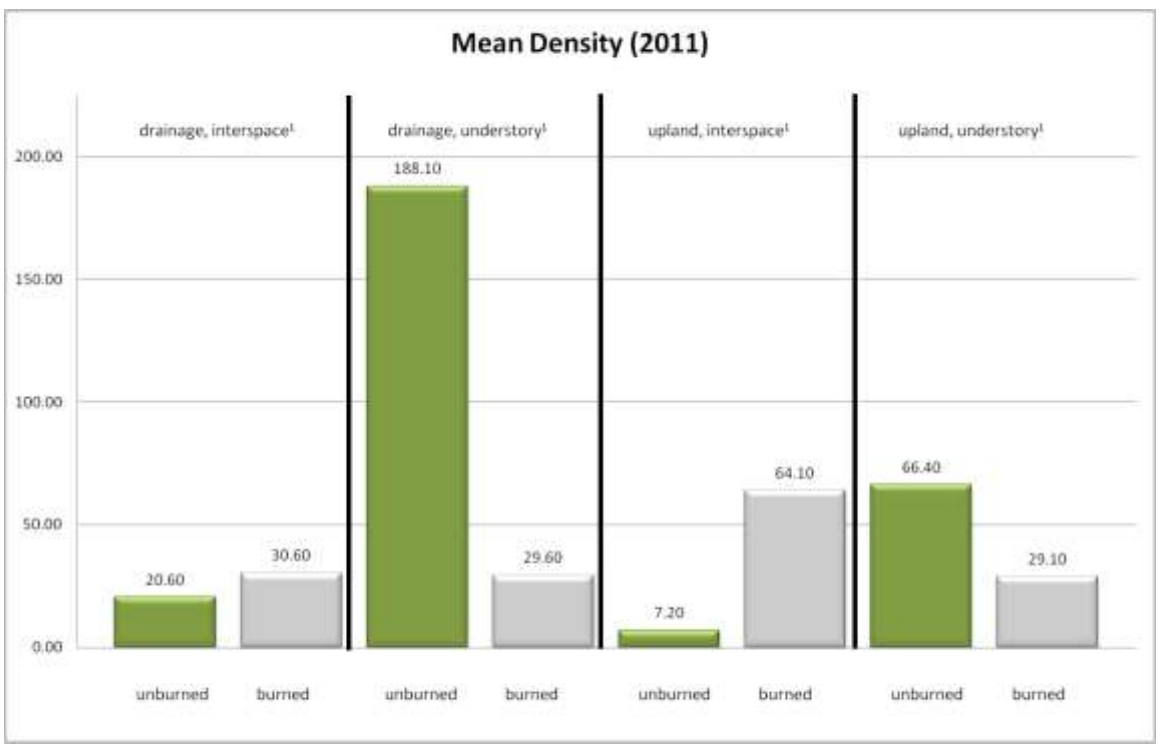


Figure 34. Mean density (labeled) by plot type for 2011. [1] indicates a significant difference.

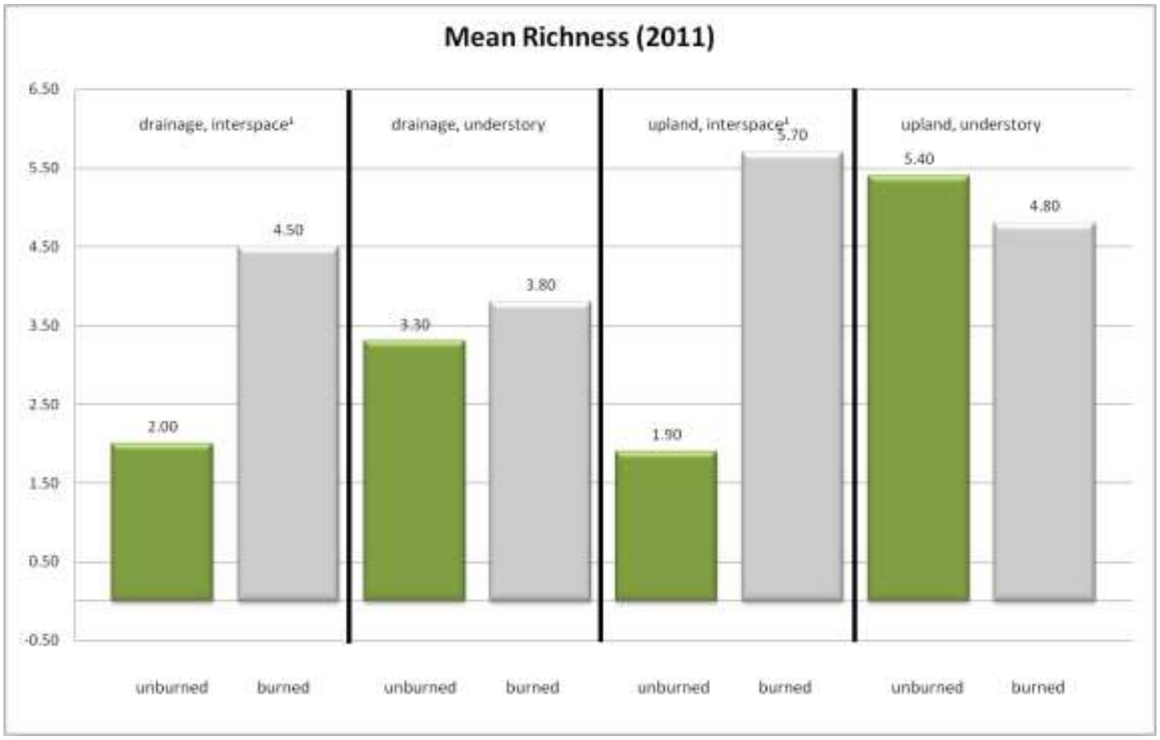


Figure 35. Mean richness (labeled) by plot type for 2011. [1] indicates a significant difference.

Results from Shannon-Weiner index showed a difference in diversity ($t(134) = -3.11$, $p = 0.002$) between unburned areas ($\bar{H}' = 0.784$) and burned areas ($\bar{H}' = 1.011$) areas. Simpson's Index (D) likewise showed that burned areas ($\bar{D} = 0.492$) were significantly more diverse ($t(152) = 4.74$, $p < 0.0001$) than unburned areas ($\bar{D} = 0.674$). As $\bar{D}$ approaching 1 is indicative of low diversity and although there was a difference between burned and unburned areas, neither would be considered relatively high in diversity by this metric. Mean evenness for burned ($J' = 0.682$) and unburned ($J' = 0.588$) areas was also significantly different ($t(134) = -2.25$, $p = 0.026$). Diversity and evenness comparisons for burned and unburned, drainage and upland plots, respectively, are presented graphically in Figure 36.

Evaluating 'interspace' microsite plots, there was a significant difference ($t(72) = 4.22$, $p < 0.0001$) in diversity between unburned ($\bar{D} = 0.734$) and burned ($\bar{D} = 0.518$) plots but not a difference in evenness ($t(60) = 1.15$, $p = 0.255$) (unburned $J' = 0.689$; burned $J' = 0.620$). For 'understory' microsite plots, unburned plots ($\bar{D} = 0.622$) were less diverse ($t(78) = 2.82$, $p = 0.006$) than burned ($\bar{D} = 0.465$). Burned ($J' = 0.748$) and unburned ($J' = 0.526$) 'understory' microsites showed a difference in evenness ($t(72) = -3.97$, $p = 0.0002$).

Drainages

Comparing only those plots in drainages regardless of microsite, there was a significant difference ($t(78) = -4.08$, $p = 0.0001$) in densities between burned (30.1 ± 25.7/plot) and unburned drainage sites (104.3 ± 112.3/plot). In drainages the burned plots and unburned plots returned a significant difference in variability of density

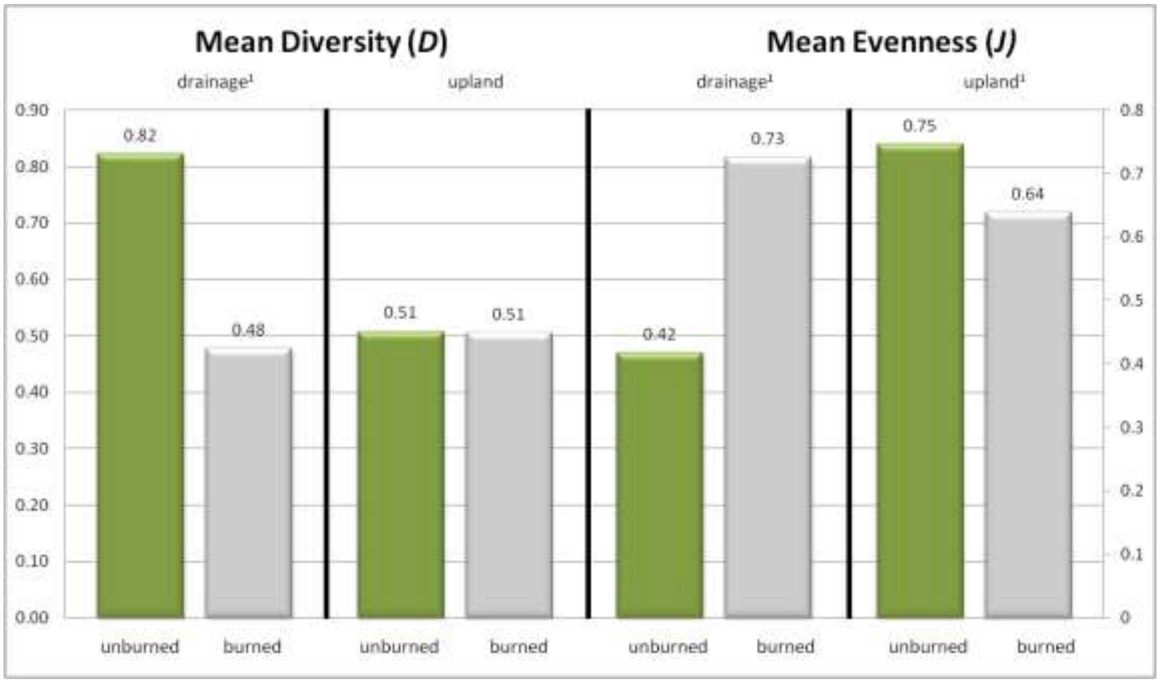


Figure 36. Mean diversity (Simpson's Index) and evenness for drainage and upland locations, comparing burned and unburned plots (2011). [1] indicates a significant difference. Diversity (*D*) ranges from 0 (infinite diversity) to 1 (no diversity). Evenness (*J*) ranges from 0 (skewed) to 1 (even).

($F(39, 29) = 19.14$, $p < 0.0001$). Mean richness ($t(68) = 2.84$, $p < 0.0001$) was significantly different in burned (4.2 ± 1.4) and unburned (2.6 ± 1.5) drainage plots although the variability in richness was not different ($F(39, 39) = 1.21$, $p = 0.548$).

Simpson's Index returned significant differences ($t(77) = 8.37$, $p < 0.0001$) in diversity between burned ($\overline{D} = 0.476$) and unburned ($\overline{D} = 0.823$) drainages. Unburned drainages had almost no diversity while burned drainages were fairly diverse. Burned drainages ($J' = 0.726$) were significantly different($t(65) = -5.48$, $p < 0.0001$) than unburned drainages ($J' = 0.418$). Burned drainages exhibited relatively more even distribution of species two years after the fire than unburned drainages, which were skewed towards a few species in a plot.

Figure 37 presents the density and richness data for drainage plots by 'interspace' microsite. Mean plant density in drainages was not significantly different ($t(38) = -1.31$, $p = 0.197$) in unburned, 'interspace' areas of drainages (20.6 ± 24.4/plot) than in burned, 'interspace' areas in drainages (30.6 ± 23.7/plot). The variability in plant density was not significantly different; burned, 'interspace' sites in drainages exhibited greater variability than unburned, 'interspace' drainage sites ($F(19, 19) = 1.07$, $p = 0.890$). Richness between unburned (4.5 ± 1.4/plot) and burned (2.0 ± 1.1/plot) 'interspace' drainage sites was significantly different ($t(38) = -6.42$, $p < 0.0001$) although the variability in richness was not ($F(19,19) = 0.54$, $p = 0.185$).

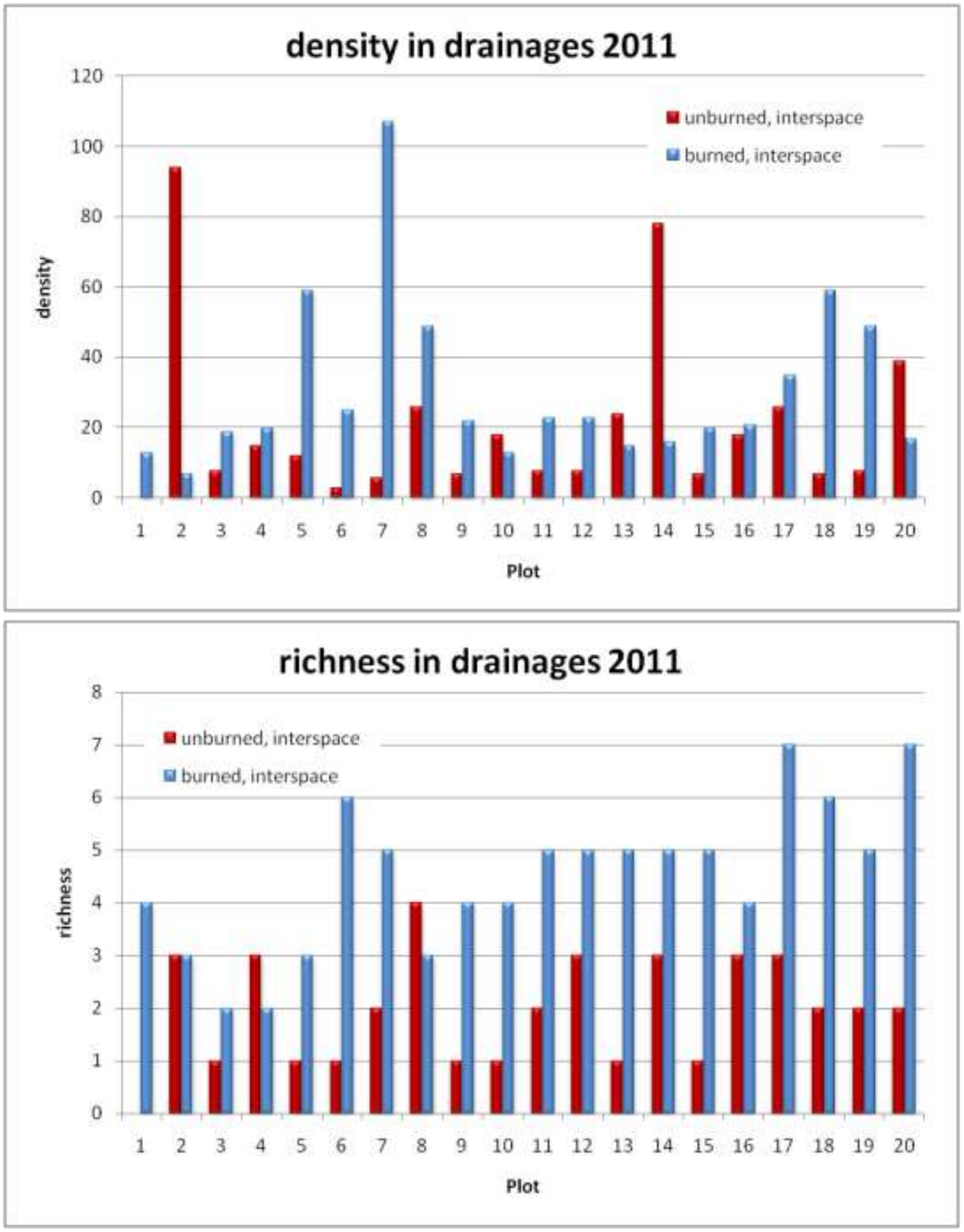


Figure 37. Density and richness for burned and unburned interspace drainage sites (2011).

Figure 38 presents the density and richness data for drainage plots by 'understory' microsite. There was a significant difference ($t(78) = 4.08$, $p = 0.0001$) in mean density for plots in 'understory' microsites in drainages between the burned (30.1 ± 25.7) and unburned (104.3 ± 112.3) plots. Drainage, 'understory' sites that had burned exhibited a difference in variability of plant density than unburned, drainage 'understory' sites ($F(39,39) = 19.14$, $p < 0.0001$). Richness between burned (4.2 ± 1.4/plot) and unburned (2.6 ± 1.5/plot) 'understory' drainage sites was significantly different ($t(78) = -4.69$, $p < 0.0001$). The variability in richness was not significantly different ($F(39,39) = 1.21$, $p = 0.548$).

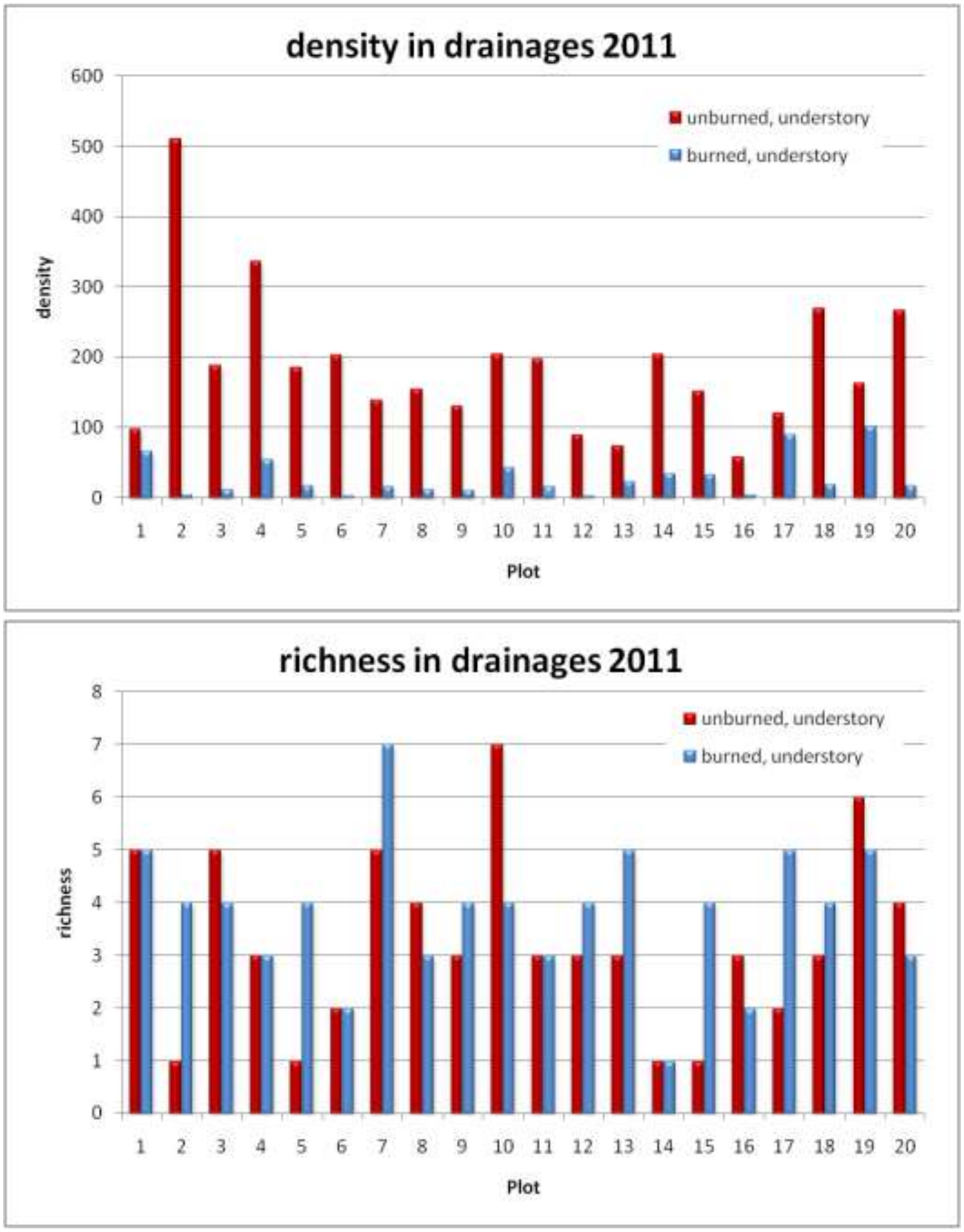


Figure 38. Density and richness data for understory plots in drainages (2011).

Upland sites

Comparing only upland plots regardless of microsite, there was no significant difference ($t(78) = -0.88$, $p = 0.3797$) in densities between unburned (36.8 ± 46.6/plot) and burned sites (46.6 ± 52.2/plot) nor in the variability of density ($F(39,39) = 0.80$, $p = 0.483$). Mean richness was not significantly different ($t(78) = -3.12$, $p = 0.0025$) in burned (5.2 ± 1.9) and unburned (3.7 ± 2.5) upland plots. Variability in richness was not significantly different ($F(39, 39) = 1.67$, $p = 0.114$).

There was not a significant difference in diversity ($t(73) = 0.04$, $p = 0.968$) for burned ($\overline{D} = 0.509$) and unburned ($\overline{D} = 0.508$) upland plots. Both unburned ($J' = 0.746$) and burned ($J' = 0.639$) upland plots exhibited similar evenness ($t(67) = 2.11$, $p = 0.039$).

Figure 39 presents the density and richness data for 'interspace' microsite location in upland plots. For upland, 'interspace' plots there was a difference in density ($t(38) = -3.87$, $p = 0.0004$) between burned (64.1 ± 65) and unburned (7.2 ± 10) plots. There was a difference in variability of density between the two ($F(19, 19) = 0.02$, $p < 0.0001$). Richness between burned (5.7 ± 1.8/plot) and unburned (1.9 ± 1.7/plot) upland 'interspace' sites was significantly different ($t(38) = -6.61$, $p < 0.001$) and was not different in variability ($F(19,19) = 0.90$, $p = 0.812$).

Figure 40 presents the density and richness data for 'upland' microsite plots under the canopy ('understory'). Mean density was significantly different ($t(38) = 2.93$, $p = 0.0057$) in unburned (66.4 ± 50.1/plot) than burned (29.1 ± 26.9/plot) upland 'understory' sites. There was a significant difference in the variability of density in upland 'understory' plots ($F(19,19) = 3.48$, $p = 0.009$). Richness of unburned upland 'understory' plots (5.4 ± 1.8) was not significantly different ($t(38) = 1.10$, $p = 0.278$) from burned (4.8 ± 1.9) upland 'understory' sites and the variability between the two was not significantly different ($F(19, 19) = 0.85$, $p = 0.713$). Spatial statistics were not computed for 2011 data due to insufficient sample sizes as determined from analyzing the 2009 data.

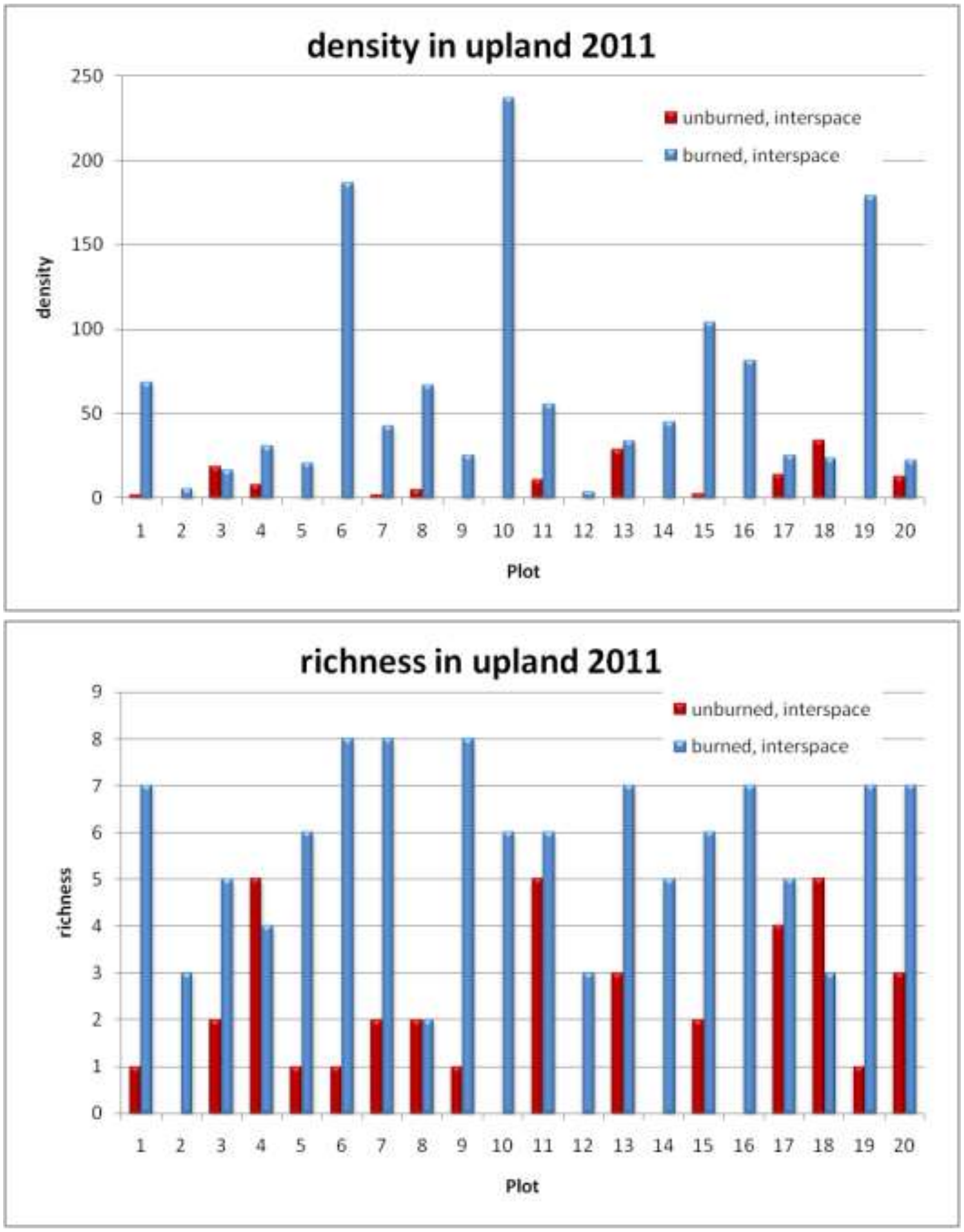


Figure 39. Density and richness for burned and unburned inter-canopy upland sites (2011).

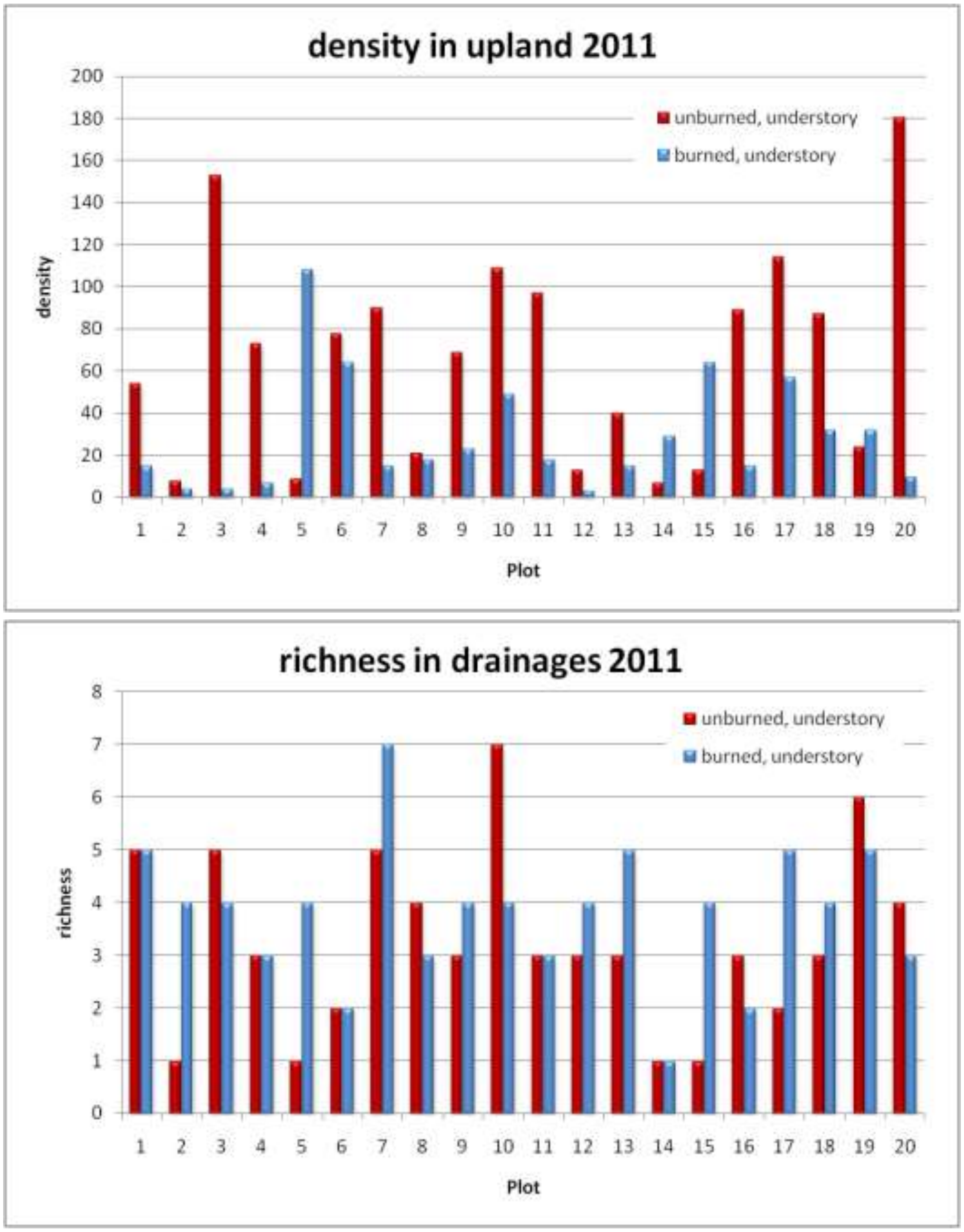


Figure 40. Density and richness data for burned and unburned uplands under the canopy (2011).

Cheatgrass (Bromus tectorum) and Blackbrush (Coleogyne ramosissima) dominance

Three years post-fire, cheatgrass was identified in 124 of 160 plots (78%) surveyed. Fifty-six plots had no cheatgrass in 2011. It was identified in 64 (80%) unburned plots and in 60 (75%) burned plots. Mean density of cheatgrass was higher in unburned plots (68.4 ± 95.96) than in burned plots (5.6 ±12.12) Mean dominance of cheatgrass in unburned plots where identified was 0.638 ± 0.335. For burned plots the mean was 0.182 ± 0.203. The difference in cheatgrass dominance between burned and unburned plots was significantly different where unburned plots had predominantly more cheatgrass than burned plots ($t(122) = 9.08$, $p < 0.0001$). The dominance distribution for *B. tectorum* in burned and unburned plots is shown in Figure 41.

For plots surveyed in drainages cheatgrass was more predominant ($t(67) = 10.95$, $p < 0.0001$) in unburned plots (0.848 ± 0.224) than in burned plots (0.248 ± 0.229). In total, 10 of 80 drainage plots had no cheatgrass identified, one unburned plots had none and nine burned plots had none. Of the 80 plots surveyed in upland areas, cheatgrass was identified in 26 (33%) unburned plots and in 29 (36%) burned plots. Mean dominance in the unburned plots (0.33 ± 0.207) was significantly higher ($t(53) = 4.58$, $p < 0.0001$) than in burned plots (0.1125 ± 0.144).

Blackbrush was identified in 22 of 160 plots (14%) surveyed. Mean blackbrush dominance in unburned plots where it occurred (0.465 ± 0.5) was marginally significantly different ($t(20) = 2.11$, $p = 0.0472$) than burned plots where it was recorded (0.123 ± 0.219). Blackbrush was not identified in any burned drainage plots and was found in six unburned drainage plots. It was identified in 13 burned upland plots and in three unburned upland plots.

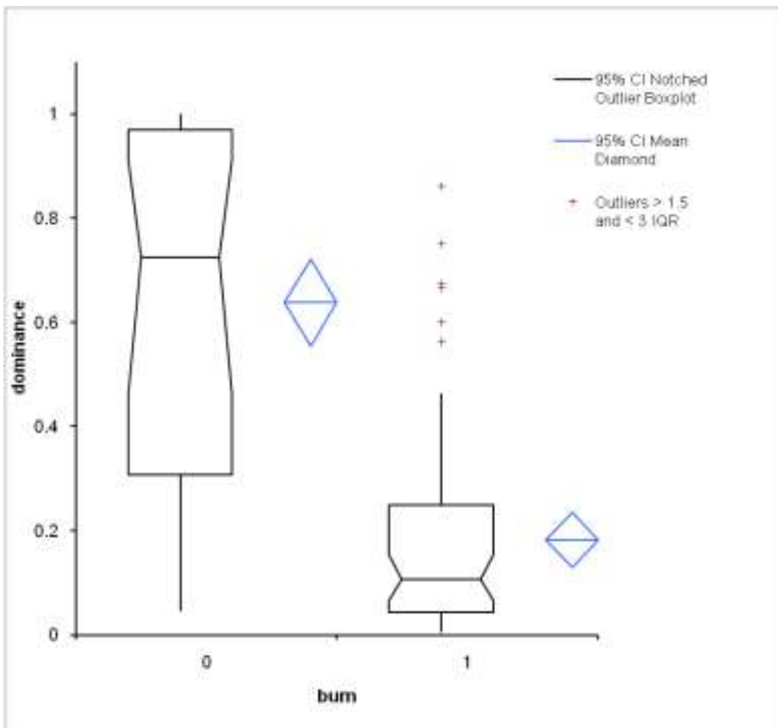


Figure 41. *Bromus tectorum* dominance for 2011 plots comparing burned (1) and unburned (0) conditions.

2011 SUMMARY

The significance of the density and richness of the understory is an important factor for repeat fire susceptibility, which would be anticipated to increase with time assuming the vegetation community regenerates post-fire. Regardless of time interval post-fire, landscapes with greater plant density provide more fuel. Certain species, such as cheatgrass, have been documented to increase fire risk. As discussed in earlier sections, vegetation community diversity is one means to assess resilience and susceptibility to wildfires. As the landscape undergoes fire and revegetation, processes occur that affect both soil chemical and physical properties which enhance or reduce erosion from wind and water. An increase in erosion inhibits the ability of a landscape to heal post-fire. Although there is a correlation between density in terms of biomass, and fuel, greater plant density particularly in terms of grasses and forbs, act to stabilize soil. The relationship between vegetation community composition and diversity with fire and the ability to recover post-fire is complex. This relationship also changes with increasing time step, and not necessarily in a linear fashion.

How closely did the post-fire regeneration reflect the unburned plant community diversity?

In 2011, three years post-fire, the species composition of burned areas more closely resembled the species overlap observed in 2009 than 2010. More species were identified in the burned landscape than the unburned landscape. Cheatgrass and sixweeks fescue were the most commonly identified species found in the unburned landscape and these two species were also highly prevalent in burned sites along with desert pincushion. Cheatgrass frequency in burned plots approached that of unburned plots in 2011. Looking within each treatment, burned and unburned sites respectively, a significant relationship was reported between richness and density and, as reported for 2009 data, was stronger in the unburned landscape. The primary differences were found between the geomorphic classes. Burned drainages were three times denser than unburned drainages while plant densities in burned and unburned uplands were similar. Density differences were observed at the microsite level as well. Richness was lower in burned versus unburned drainages as well as in both microsites in upland burned and unburned locations.

Burned drainages and uplands were not different in density however the upland landscape had greater richness than drainages. From a diversity perspective, there was no difference in diversity or richness across the burned landscape for both drainage and upland geomorphologic locations.

Was there a difference in regeneration in drainages compared with the rest of the landscape?

Three years post-fire showed large differences in drainages in particular and both in density and richness. Vegetation density was much lower in burned drainages than unburned drainages whereas richness was greater in burned than unburned drainages. Density by microsites within burned drainages showed higher densities in interspace microsites and lower densities in understory microsites. Diversity in burned drainages was significantly higher than unburned drainages which were not very diverse. The distribution of the species and abundance in burned drainages was relatively even and significantly more so than unburned drainages. This is due to the fact that although three species were predominant (cheatgrass, sixweeks fescue, whitestem blazingstar) the other species identified were recorded in relatively greater numbers than in previous years.

Did microsite location provide any advantage towards reflecting the unburned vegetation community?

In 2011 the interspace microsites did appear to have provided some advantage in terms of vegetation re-establishment in both density and richness. All microsite combinations were significantly different in the burned landscape compared with the unburned landscape and interspace microsites within the burned area had greater plant density than in the equivalent unburned sites. The understory microsites showed much lower densities in burned areas compared to unburned areas. The burned interspace microsites were also consistently richer. In terms of biodiversity, drainages were approximately half as diverse in the burned compared to the unburned landscape although that diversity was fairly evenly distributed. Uplands were no different between burned and unburned sites and both exhibited an evenness metric that was closer to even than skewed. These results indicate that three years post-fire the burned landscape still does not reflect the vegetation community structure of the unburned landscape and that microsite has an effect, albeit counterintuitive. Because burning of canopies release nutrients for subsequent vegetation germination and growth, understory canopy microsites would be expected to have a higher plant density than interspace microsites, which was not the case for this particular fire location.

Where will the landscape be more susceptible to erosion three years post-fire based on these data?

The data suggest that drainages and where shrub canopies had burned would be more prone to erosion due to lower plant densities, but interspace microsites are not as likely to promote increased erosion. This is because plant densities were found to be less in the burned area in understory microsites whether in drainages or uplands. The understory microsites in drainages are the primary source of the difference between the burned and unburned landscape. Species richness was consistently higher in the burned landscape. However, the difference in cheatgrass predominance between the burned (low) and unburned (high) landscapes should not be overlooked. Cheatgrass was 12 times denser in the unburned sites than the burned sites overall and seven times as dense in unburned uplands than burned uplands. In drainages the difference in density was twelve and a half times for unburned compared to burned areas. The risk of fire recurrence due to cheatgrass regeneration in previously burned sites remains relatively low compared to fire risk in areas previously unburned even three years post-fire. With the relatively low overall species densities in drainages, any soil stability benefits in burned areas would not likely be realized. Because the overall density in burned uplands was greater than in the unburned upland landscape, soil stabilizing benefits from sheer number of plants three years post-fire would likely be greater there than on the previously unburned landscape.

In drainages the plant density for burned sites was significantly less than that of unburned sites. For cheatgrass specifically, the density difference in unburned versus burned sites was 12 times greater. Soil stability benefits from generic plant density would be expected to be greater in uplands than in drainages, and specifically burned drainages would not be comparable to an unburned landscape based solely on plant density.

OVERALL SUMMARY

This study examined the regeneration and diversity of a burned and unburned blackbrush community at three discrete time periods post fire. The resilience of a landscape to erosion from natural physical processes (e.g., wind, water) depends in a great part on the stability of the soil and in particular the vegetation cover. Subsurface biomass, namely the root structures of plants, help stabilize soil. Vegetation plays additional roles in maintaining the integrity of the upper soil horizons as well, through direct water capture in rain events, soil water use, among others. Biomass varies with plant species, abundance, and spatial distribution, among other factors as well. There is also a temporal component to biomass and root structure that will vary with community composition. Therefore it is important not only to calculate basic metrics such as plant density and richness, but also abundance, dominance, and diversity metrics. Thus there are several factors that come in to play in terms of maintaining a functional landscape where soil stability is concerned. Regeneration of the plant community can have a significant effect on landscape resilience, and return to ecological functionality after a wildfire.

One year post fire cheatgrass was observed in the burned area. Plant density in burned areas approximated that of the unburned landscape although burned areas lacked equivalent richness. An examination of the landscape in detail showed that although there was no difference at the broadest level, the interspace of burned drainages was 16 times denser than where burned and was equally rich as unburned drainage interspace. Overall the diversity one year post fire was substantially lower where the burn occurred with the exception of the drainage interspace as identified above. In terms of soil stability the density of plants should provide similar benefit in the burned landscape however the benefit of root mass of blackbrush or other shrub species would not be realized as shrubs did not substantially regenerate within one year. From an ecological standpoint the decrease in diversity would also be expected to have an effect on soil stability and the likelihood of fire recurrence. Burned drainages did not reflect the relative evenness of unburned drainages and exhibited almost no diversity. Diversity was equally low in burned and unburned uplands however the burned uplands exhibited a trend towards even distribution of the plant community diversity. Cheatgrass had not recolonized the burned area one year post fire to the extent that it was found in the unburned landscape. The majority of drainages sampled showed no cheatgrass at all and where it was found was usually in the unburned area. Cheatgrass in the burned uplands was rare. Blackbrush on the other hand did show signs of high regeneration within one year post fire in the uplands but was not found in any burned drainages.

Two years after the fire, the burned area exhibited very high plant densities, significantly greater than the adjacent unburned landscape. This was primarily a function of the six-fold higher densities measured in burned drainages over measurements from unburned drainages. Overall the richness of burned and unburned areas was comparable however the unburned landscape remained overall more diverse. Examining drainages specifically, density in unburned drainages was no different between interspace and understory while in the burned drainages the interspace exhibited nearly twice the plant density of the understory. This difference was not observed in the burned uplands, which exhibited equal densities across microsites. Burned drainages showed almost no diversity two years post fire. At this two year post fire time step, the burned uplands did not approximate the microsite density proportion of the unburned landscape, which was nearly twice as dense in the understory as the interspace but did exhibit comparable diversity. Although similarly even, neither burned nor unburned uplands would be considered to exhibit evenness in diversity, and burned drainages showed highly skewed diversity distributions indicating only a few species were dominating the community. Cheatgrass was three times as prevalent in the unburned landscape. Blackbrush occurrence was no different in burned and unburned areas although the size was different, where the shrubs had not yet established to the full size potential as compared with unburned areas. While there was no overall difference in blackbrush occurrence it was recorded four times more often in burned uplands than in unburned uplands. The implications for soil stabilization two years post fire interpreted based solely on the plant density is that a positive effect simply due to biomass in the upper soil horizon could be expected. However the other aspect to biomass is its role as fuel and the likelihood of burning would be expected to increase with additional fuel loading. Shrubs had not regenerated sufficiently two years post fire for soil stabilization benefits to be realized through extensive root development.

The final time step examined was three years post fire. The unburned landscape was found to show substantial differences in plant densities and richness between the two geomorphologic settings (drainages and uplands), respectively. In contrast the burned landscape showed only a density difference between the understory and interspaces in sampled uplands. In the burned landscape densities and richness were fairly uniform. This difference might appear counterintuitive initially, in that the understory would not be expect to have significantly lower plant densities three years into recovery; however due to the fact that the shrubs were completely killed in the fire, whether or not real effects gained from an understory are possible (probably not) must be considered. Within the landscape the unburned drainages were shown to have three times the plant density as unburned drainages although burned drainages exhibited greater species richness. Upland areas were equally dense although burned uplands also showed greater richness. Microsite differences in richness of the interspaces were observed between burned and unburned areas; burned areas had higher richness. The understory microsites exhibited similar richness for burned and unburned areas. From a diversity perspective the uplands were equally diverse regardless of whether or not it had burned. The unburned uplands had a more even distribution of diversity compared to the burned landscape but neither was particularly even. Burned drainages were more diverse and the diversity was fairly evenly distributed while unburned drainages returned a generally low and skewed diversity. Cheatgrass was reported in three-quarters of the burned area sampled which was a close approximation to results from the unburned area, and densities were higher in unburned plots. It was also more dominant in the community composition in unburned plots. Blackbrush was not reported in the burned drainages but was reported in unburned drainages. It was four times more prevalent in the burned uplands than the unburned uplands. Three years post fire the burned landscape continued to revegetate but had yet to approximate the condition of an unburned landscape. The exotic species cheatgrass, notorious for changing fire cycles, had increased in the burned area but so did other native species as well. Due to the uniformity in density and richness across the burned landscape, soil stabilization benefits from vegetation biomass would also be anticipated to perform consistently at a broad level. The lower density of the burned area would not be expected to support stabilization benefits to the same level as the unburned landscape. However there may be some additional value gained from the greater diversity of species in the burned landscape. The lower prevalence of cheatgrass coupled with higher diversity in the burned landscape might reasonably be anticipated to have an effect on fire recurrence, although no desert plant species can be considered fire resistant; any vegetation present whether annual forb, grass or perennial shrub constitutes fuel.

The landscape was shown to be dynamic in terms of trajectory for revegetation. The three discrete time periods sampled showed the breadth of variability that exists when comparing burned and unburned areas. Of particular significance is the apparent opportunity to revegetate, to resist cheatgrass recolonization, and develop the biomass needed to assist in soil stabilization, which is provided by different geomorphologic landforms and by microsites.

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