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Submitted to

U.S. Department of Energy
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EXECUTIVE SUMMARY

In the deserts of the American Southwest, fire return intervals of centuries to millennia are being replaced by fire return intervals of decades. This increased burn frequency has implications for post-closure management and long-term stewardship for Soils Corrective Action Units (CAUs) and Corrective Action Sites (CASs) on the Nevada National Security Site (NNSS), Tonopah Test Range, and Nevada Test and Training Range for which the U.S. Department of Energy (DOE), Environmental Management-Nevada (EM-NV) is responsible. For many CAUs and CASs, where closure-in-place alternatives were implemented or are being considered, there is a chance that these sites could burn over while they still pose a risk to the environment or human health given the long half-lives of some of the radionuclide contaminants of concern (COCs) (Shafer *et al.*, 2007; Shafer and Gomes, 2009).

Although it would have been ideal to conduct a fire-related study at a radionuclide contaminated site on the NNSS to determine how contaminated soils are transported post-fire, it was more practical to examine fires in environments analogous to where Soils Activity CAUs and CASs exist. Therefore, a series of studies were initiated at three radiologically uncontaminated analog sites to better understand the possible effects of wildfire on erosion and sediment transport by wind and water should vegetation at contaminated sites burn at the NNSS. The studies took place at the sites of the Gleason Fire, a prescribed burn that occurred near Ely, Nevada, which is in the Great Basin ecoregion; the White Rock Fire that occurred near Mesquite, Nevada, which is in the Mojave Desert ecoregion; and the Jacob Fire that occurred near Hiko, Nevada, in the transitional zone between the Great Basin Desert and the Mojave Desert. Each site was representative of one of the three ecoregions found on the NNSS. Erosion and vegetation characteristics were measured multiple times post-fire on both burned and unburned test plots at these study sites to compare and monitor the effects of fire over time. However, at the Mojave Desert site, there was an additional burn from nearly 25 years ago that was also compared with the more recent burned areas from 2013. Also, the time elapsed after the fire and the time when sampling began differed among the ecoregions. For example, at the Great Basin site, samples were collected prior to the fire and less than one month after the fire, whereas data collected at the Mojave Desert site did not occur until 22 months after the fire. The differences between the sample measurements and the time that elapsed after the fire may contribute to the variability in the data and the interpretations of the data.

Runoff and water erosion were quantified through a series of rainfall and runoff simulation tests in which controlled amounts of water were delivered to the soil surface. Runoff data were collected from different microhabitats (e.g., undercanopies, interspace soils, ridges, and drainage areas). The data showed soil hydrophobicity (i.e., water repellency) on the Great Basin site for up to 12 months after the fire. The soil structure remained changed and weaker on the burned areas compared with the unburned areas for the three-year study period. Although runoff data at both the Great Basin site and transition zone site were highly variable, the post-fire runoff potential did not generally increase at either location after three years of monitoring. Mojave runoff did not differ much between fire ages or between burned and unburned areas, possibly because of the amount of vegetation growing on all three surfaces and because the fire intensity was likely lower relative to other ecoregions due to the amount of space between plants. Additionally, the measurements

conducted on the Mojave fire occurred several years after the fire had burned and not within several months of the burn, which may also account for the lack of differences in runoff between burned and unburned locations.

Wind erosion was assessed and quantified with a Portable In-Situ Wind ERosion Lab (PI-SWERL) on both the burned and unburned soils of different microhabitats (e.g., ridges, drainages, interspaces between plants, and plant undercanopies). Estimates of emissions (e.g., particulate matter with aerodynamic diameters less than or equal to 10 micrometers, or PM_{10}) at different wind speeds were collected and filter samples were analyzed for chemical composition. The measurements among the fires were similar and the results of the wind erosion measurements indicate that there were seasonal influences on many parameters, but the potential for PM_{10} windblown dust emissions was higher on burned areas compared with unburned areas. Among the burned areas, drainages produced the most dust emissions, whereas burned ridges were the least emissive. Emissions at the Mojave site and transition zone site were similar between burned and unburned areas approximately three years after the respective fires. Emissions remained much greater on burned soils at the Great Basin site. However, site selection may have also contributed to the differences of emissions among the three study locations.

Post-fire vegetation responses were documented for three years following each fire in different microhabitats (e.g., ridge, drainage, undercanopy, and interspace) at the Great Basin and transition zone sites. In 2013, at the Mojave fire site, random permanent plots were established in newly burned areas, areas that burned approximately 25 years ago, and an unburned control area. At each fire site, the regeneration of native plant species dominated the burned areas, but invasive annual grasses were present at each fire site. During the course of each three-year study, the invasive annual grasses increased in density or percentage at both the Great Basin and transition fire sites but not at the Mojave site, where annual invasive grasses remained high throughout the study period. However, the chronosequence data from the Mojave site indicate a transition to different invasive grass species with time. The post-burn vegetation at all three ecoregions is still in the early stages of succession, and shrub size should increase over time and shrub density should decrease over time. However, the recovery of the plant communities to pre-burn compositions may not occur for decades or centuries, or may not occur because of changes in climate and the presence of invasive annual grasses.

Results from the three study locations suggested that contaminated soils on the NNSS that experienced wildfire had the potential for soil erosion and transport of contaminated soils. Both wind and soil runoff studies indicated that locations in the Great Basin and transition zone ecoregions were more susceptible to erosion than the Mojave Desert ecoregion, but the highest levels of erosion occurred during the early portions of the study (the first 24 months) and the Mojave Desert was not measured until 22 months after the burn. Additionally, data from the Great Basin indicated deeper soils may increase susceptibility to wind erosion. Historically, the Great Basin and transition zone ecoregions burned more frequently than the Mojave Desert because of fuel abundance and connectivity—and they continue to burn more frequently, particularly after the introduction of invasive grasses (Davies and Nafus, 2013). However, both ecoregions and the transition zone have burned with extensive, high-severity fires when certain weather and fuel conditions are met. For example, the high abundance and productivity of annual grasses and forbs—which are usually produced by above average winter precipitation—combined with early spring and summer warm, dry conditions. Future climate scenarios predict increased temperatures, variable to reduced precipitation, and larger, more extreme wildfires throughout western North America. These predictions suggest that future fires on the NNSS could increase in frequency, intensity, and severity similar to their analog ecosystems throughout the Great Basin and Mojave Desert (Dennison *et al.*, 2014; Abatzoglou and Williams, 2016; Westerling, 2016).

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LIST OF ACRONYMS

ac	acres
ºC	degree Celsius
CAS	Corrective Action Site
CAU	Corrective Action Unit
cm	centimeters
CN	curve number
COC	contaminant of concern
DOE	Department of Energy
EC	elemental carbon
ºF	degree Fahrenheit
ft	feet
ha	hectare
in	inches

km	kilometers
kph	kilometers per hour
K_s	saturated hydraulic conductivity
lpm	liter per minute
m	meter
MAB	months after burn
MDTI	mini-disk tension infiltrometer
mi	miles
mph	miles per hour
NB	new burn
NFO	Nevada Field Office
NNSA	National Nuclear Security Administration
NNSS	Nevada National Security Site
NTTR	Nevada Test and Training Range
OB	old burn
PI-SWERL	Portable In-Situ Wind ERosion Lab
PM_{10}	airborne particulate matter with aerodynamic diameters less than 10 micrometers
RPM	revolution per minute
SCS	Soil Conservation Service
TTR	Tonopah Test Range
XRF	X-ray fluorescence spectrometry

INTRODUCTION

Background

Since the mid-1980s, the fire regime in the southwestern United States has changed and larger, more frequent fires have occurred. The increasing occurrence and size of fires in this region have been associated with both changes in climate and changes in the ecosystem structure and composition. The climate has generally been warmer and drier, and ecosystems have increased fuel loads and fuel dynamics caused by the spread of invasive annual grasses and past land-use practices (e.g., Westerling *et al*. [2006] and Crockett and Westerling [2018]). 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 biomass. Regionally, the maximum and minimum temperatures have increased since the 1960s, which have altered fuel moisture and fire behavior (Westerling *et al*., 2006; Diffenbaugh *et al*., 2014; Abatzoglou and Williams, 2016). Additionally, prolonged drought conditions across large areas of the region since 1999 and accumulated fuel loads because of wildfire suppression have led to increases in fire occurrence, large wildfires, and severe wildlife activity (Savage and Swetnam, 1990; Westerling and Swetnam, 2003; Westerling *et al*., 2006; Crockett and Westerling, 2018). Consequently, the historical return intervals of >100 years for large fires in southwestern deserts have been replaced by decadal fire return intervals (Brooks, 2006). Additionally, the introduction of invasive annual grasses, most commonly cheatgrass (*Bromus tectorum*) and red brome (*Bromus rubens*), have substantially altered fire dynamics in shrubland ecosystems. Both grasses were introduced to the region during Euro-American settlement in the nineteenth century (Knapp, 1996). Although both species were found at a few disturbed sites on the Nevada National Security Site (NNSS) as early as the 1960s, the invasive species spread across the region in the 1980s (Hunter, 1991). Today, these grasses rapidly invade disturbed areas and in many plant communities they have colonized interspaces between shrubs, which increases the total fuel load and allows fires to move more easily between shrubs (Knapp, 1996). Besides increasing the chance of a fire occurring and the probability of fires that are larger than historical fires, the invasive plants quickly colonize areas that have burned on the NNSS, which increases the chance that fires will reoccur.

Implications and Objectives

Fires and the changing regional fire regime have important implications for post-closure management and long-term stewardship of Corrective Action Units (CAUs) and Corrective Action Sites (CASs), which are areas for which the U.S. Department of Energy (DOE), Environmental Management-Nevada (EM-NV) has regulatory closure responsibility. For many Soils Activity CAUs and CASs, where closure-in-place alternatives were implemented or are being considered, there is a chance that these sites could burn while they still pose a risk to the environment or human health given the long half-life of some of the radionuclide contaminants of concern (COCs) (Shafer *et al*., 2007; Shafer and Gomes, 2009). Therefore, it is necessary to better understand the possible effects of fire-induced erosion and particle transport by wind and water, the post-fire response of vegetation, and the risks and perceived risks to site workers and public receptors in communities around the NNSS, Tonopah Test Range (TTR), and Nevada Test and Training Range (NTTR). Ideally, these studies would have occurred at a radionuclide-contaminated site on the NNSS, but it was more practical to examine fires in environments analogous to where Soils Activity CAUs exist. Three studies located in similar ecoregions to those found on the NNSS were initiated to examine the effects, and duration of the effects, of fire on the potential for wind and water erosion, as well as the vegetative recovery of burned sites. The first of these studies was conducted at the Jacob Fire site, which is representative of the transition zone between the ecoregions of the Mojave Desert and the Great Basin Desert. The second study was conducted at a prescribed burn near Ely, Nevada. This second site, named the Gleason Fire site, represents the Great Basin Desert ecoregion, which is characteristic of the northern portion of the NNSS and most of the TTR. The second largest number of Soils Activity CASs are found within the Great Basin Desert ecoregion, including those in the higher elevations of the NNSS and at the TTR, where lightning strikes are most common. The third study was conducted at the White Rock Fire near Mesquite, Nevada, in the Mojave Desert ecoregion, which is representative of vegetation found on the southern portions of the NNSS.

The basic approach for characterizing the effects of fire on the potential for wind and water erosion was to conduct measurements on soils in areas that burned and to make parallel measurements (i.e., control measurements) in similar areas of the same landscape on unburned soils. Soil measurements in the unburned control sites provided a benchmark for comparing how fire may have affected the site. Past research (Shafer *et al*., 2010) has shown that vegetation differentially influences the magnitude of wind and water erosion by trapping wind-borne material and concentrating biotic activity (e.g., roots and burrowing fauna). Therefore, site vegetation was also surveyed. The plant species were identified and basic physical measurements were recorded and analyzed. Ultimately, the project objectives were to: 1) determine the source areas and types of particles available in post-fire systems for wind and water erosion, 2) monitor the dynamics of sediment transport over time, and 3) measure fire-related changes in vegetative cover and composition.

Overview of Gleason Fire (Great Basin Ecoregion)

The Gleason Fire was located in east-central Nevada within the boundaries of the Great Basin Desert. This prescribed fire burned an area of 155 hectares (ha) (383 acres [ac]) on August 13, 2009, that was originally vegetated with mixed sagebrush and pinyon-juniper woodland (Table 1). It was estimated that pre-fire fuel loads would result in a low- to medium-intensity fire. Nearly all of the surface plant material was burned, leaving only the occasional singed remains of larger shrubs and trees. Ash and charred soils delineated interspace and plant undercanopy areas after the fire. The nearest weather station to the Gleason Fire was located in Ely, Nevada, and it indicated an average annual temperature of 7.1 °C (45 °F) and annual precipitation ranging from 20 to 40 centimeters (cm) (7.8 to 16 inches [in]) depending on elevation, most of which occurred during winter months.

Post-fire measurements of water erosion, water runoff, wind erodibility, and vegetation recovery were measured for one week prior to the burn and continued intermittently for 34 months after burn (MAB). Additionally, the Gleason Fire was measured for soil properties relating to water and wind erodibility 0.13 months prior to the burn, which was the only site measured pre-burn. Data collection occurred in the burned areas and an

Table 1. General information of the three fire sites.

	Gleason Fire	Jacob Fire	White Rock Fire
Ecoregions	Great Basin ecoregion	Great Basin-Mojave Desert transition zone	Mojave Desert ecoregion
Pre-burn vegetation	Sagebrush, pinyon pine, juniper	Blackbrush, Mormon tea	Creosote bush, Joshua Tree, Yucca, Blackbrush
Elevation range of the fire	2,183-2,397 m (7,162-7,864 ft)	1,200-1,500 m (3,937-4,921 ft)	575-921 m (1,886-3,021 ft)
Location	39°23'43" North, 115°03'57" West	37°42'17" North, 115°12'41" West	36°42'19" North, 114°04'27" West
Fire ignition	Prescribed, low intensity, August 2009	Lightning, intensity unknown, August 2008	Lightning, low to moderate intensity, April 2012
Fire size (ha [ac])	155 (383)	186 (460)	479 (1,184)
Precipitation	200 to 400 mm (7.8-15.7 in), mainly as winter snow with spring and fall thundershowers and rains	Estimated 160-170 mm (6.2-6.7 in), 18 percent occurring in the summer	Estimated 150 mm (5.9 inches) from Mesquite, NV
Soil	Gravelly loams and gravelly clay loams shallow to moderately deep and well drained, with gravels, cobbles, and stones	Alluvial/colluvial materials, with gravelly and sandy loam	Gravelly sandy loam and gravelly loam, with gravels cobbles, and stones
Time of investigation	2009-2012, 0.13 months before the burn to 34 months after the burn	2008-2011, 1-34 months after the burn	2014-2016, 22-47 months after the burn

adjacent control area that did not burn. Generally, measurements were collected under shrubs (termed "undercanopy") or between shrubs (termed "interspace") to better understand the source and dynamics of post-fire wind erosion, differences in water runoff, and vegetation succession.

Overview of Jacob Fire (Transition Zone between Great Basin and Mojave Deserts)

The Jacob Fire was ignited by lightning approximately 16 kilometers (km) (9.9 miles [mi]) north of Hiko, Nevada (Table 1). The fire burned approximately 186 ha (460 ac) of shrublands dominated by blackbrush (*Coleogyne ramosissima*) and Mormon tea (*Ephedra nevadensis*) between August 6 and 8, 2008. Although fire intensity data were not available, fires that completely consume plants in the late-seral stands of blackbrush are generally associated with moderate- to high-intensity fires (Brooks *et al*., 2007). The nearest weather station was located in Hiko, Nevada, and precipitation averaged 17.0 cm (6.2 in) annually between 1999 and 2009. Measurements of water erodibility and runoff, wind erodibility, and post-fire vegetation succession were measured periodically from 1-34 MAB. Data from burned areas were compared with control areas that did not burn.

Overview of White Rock Fire (Mojave Desert)

The White Rock Fire occurred 11 km (6.8 mi) south of Mesquite, Nevada, and was ignited by lightning on April 26, 2012 (Table 1), burning approximately 479 ha (1,184 ac). The White Rock Fire partially burned over the 1,420 ha (3,510 ac) Bunker Fire—which burned August 9, 1993—and the Cabin Fire—which burned June 6, 2006. The severity of the White Rock Fire was low to moderate, whereas the severity of the Bunker Fire was moderate to high. Little information exists on the Cabin Fire. The general pre-fire vegetation consisted of creosote brush (*Larrea tridentata*), white bursage (*Ambrosia dumosa*), blackbrush, and several species of yucca (*Yucca* spp.). Between 22 and 47 MAB of the White Rock Fire, measurements of wind and water erodibility were collected and post-fire vegetation succession was recorded for both White Rock Fire and Bunker Fire sites. Because measurements of the White Rock Fire and the Bunker Fire did not occur until 22 MAB, it is difficult to compare early post-fire soil and vegetation dynamics with the other sites.

POST-FIRE COMPARISONS OF SOIL PROPERTIES

At each fire study site the effects of how wildfire altered the physical properties of soil were determined. The alteration of soil properties can affect the water erodibility of radiologically contaminated soils. The three-year study was performed to investigate post-fire soil properties, infiltration, and runoff characteristics conducted with a low intensity rainfall simulation at each site. Table 1 summarizes the general information of the three sites.

Runoff

At each site, field rainfall-runoff simulations were performed over three consecutive years to investigate the effects of fire on runoff generation and erosion in different ecological regions. Runoff results for the three sites were compiled and compared based on two aspects: runoff ratio (or coefficient) and runoff curve number (CN).

<u>Runoff Ratio</u>

Runoff ratio is defined as the ratio of the total runoff (depth or volume) generated from a given rainfall event to the total rainfall input (depth or volume) for the event. The three ecoregions had different the 100-year, one-hour rainfall amounts because of differences in natural precipitation. Therefore, the runoff ratio was chosen to normalize data rather than the runoff volume to overcome the bias of runoff volume because of the different rainfall inputs. Results of the runoff ratio were averaged for each condition (i.e., burned interspace, burned undercanopy, unburned interspace, and unburned undercanopy) over time (Figure 1).

At the Gleason Fire site, tests were conducted to compare runoff on burned and unburned interspace and undercanopy areas. The rainfall simulator test period covered 10-34 MAB. In the first year after the fire (10 and 12 MAB), most tests generated little or no runoff, especially on unburned soil surfaces. Therefore, no data is reported for some of the sub-sites in Figure 1. The runoff ratios of several tests on burned surfaces in undercanopy areas exceeded 10 percent. In the second study year (22 and 25 MAB), most tests generated higher amounts of runoff than the first year. Most of the runoff ratios were larger than

Figure 1. Runoff ratio comparison for the three fire study sites. Note the different scales of the y-axes.

5 percent, and larger than 10 percent in more than half of the tests. On average, runoff ratios were largest in the burned interspace, but the unburned interspace also showed high runoff potential. In the third study year (34 MAB), little or no runoff was generated in the majority of the tests, whereas burned interspace areas showed relatively higher runoff potential, which suggests that the effects of wildfire extended to the third year after the fire. However, the overall runoff ratios were much smaller than in the second year.

At the Jacob Fire site, runoff simulation tests were conducted 1-34 MAB. The results showed that runoff was generated on burned and unburned alluvial surfaces, but little runoff was generated in the drainage channels. Similar to observations at many other sites, the soils in active channels usually had high hydraulic conductivity because their coarse texture and high gravel content do not allow excess runoff to be generated within the channels. Additionally, runoff was most frequently observed in burned or unburned interspace areas, and runoff ratios on these surfaces were fairly large (often approximately 40 percent or larger), indicating relatively high runoff potential regardless of burning. The largest runoff ratio in all cases was observed in unburned interspace, which was higher than 90 percent, meaning that less than 10 percent of the rainfall infiltrated and more than 90 percent of rainfall converted to runoff. At undercanopy areas, burned and unburned runoff ratios were generally similar, but typically lower than interspace areas, although some cases showed comparable values to interspace areas. However, in the early period after the burn (1-7 MAB), runoff ratios for burned interspace showed an increasing trend, whereas runoff ratios for unburned interspace show a decreasing trend over the same period. Comparing the burned and unburned surfaces of the same category (undercanopy and interspace) from the latter part of the second year with the third year (22 MAB to 34 MAB), the results did not show a significant effect of fire on runoff generation. However, the results may indicate a seasonal change in runoff generation for unburned surfaces. For example, the runoff ratio for interspace control conditions was relatively low from fall to spring (November: 3 and 15 MAB; and March: 7 MAB) and high in summer (June to September: 1, 22, 25, and 34 MAB), whereas for undercanopy control conditions, the runoff ratio was measureable during the fall to spring season. These temporal differences seemingly suggest a seasonal trend when the effect of fire is absent. However, validating this trend requires more field test data.

The runoff simulation tests on the White Rock Fire site were conducted at 26, 35, and 47 MAB. Unlike at the Gleason and Jacob Fire sites, an older burn site was selected for study to see if the effects of wildfire endured from 2 to 4 years and 25 years post-fire. These tests were conducted on five different geomorphic surfaces classified by relative age within each fire and control site. Runoff simulations were conducted on interspace and undercanopy spaces in the burned area. In total, five to six tests for each of the interspace and undercanopy areas were conducted for burned surfaces, and two to three tests for each of the interspace and undercanopy areas were conducted for unburned surfaces. For the tests conducted 35 MAB, more than half of the tests generated a significant amount of runoff (runoff ratio larger than five percent), whereas the largest runoff ratio occurred on the unburned undercanopy surface. On average, burned surfaces did not show higher runoff potential. For the tests conducted 47 MAB, runoff was greater on unburned surfaces. The unburned undercanopy area exhibited the largest runoff ratio and it was much higher than the other surfaces. Overall, the effects of fire after that much time post-fire did not exhibit the expected influences (i.e., reducing infiltration and increasing runoff) on rainfall-runoff processes at this site based on the runoff ratio results.

Runoff Curve Number

The Soil Conservation Service (SCS), now the Natural Resource Conservation Service, curve number (CN) approach is commonly used to account for the potential runoff after precipitation losses from evaporation, absorption, storage, and transpiration (USDA-SCS, 1986), and describes a watershed response for a rainfall event. Curve numbers range from 30 to 100. A CN of 100 represents a completely impervious surface, with decreasing CNs for more permeable surfaces. The CN can be estimated using the measured runoff data from the rainfall simulation test. If runoff did not occur in all quadrants of the test plot during the rainfall simulation test, the CN could not be effectively estimated and was displayed as not available.

At the Gleason Fire site, CNs were available for most conditions throughout the sampling period. Similar to the Jacob Fire site, differences between CNs of burned and unburned surfaces were relatively small. The difference between the average CNs for burned and unburned interspace was 5 at the highest but frequently less than 3. The differences between interspace and undercanopy surfaces were also small, except at 25 MAB when the difference was 8, which was larger than that the CNs between burned and unburned conditions. A difference in CN of 5 (e.g., between 90 and 85) could be a substantial amount of runoff in desert environments depending on precipitation levels, and may account for over 20 percent of runoff. For each surface, the CN results show variations over time within the three-year testing period. The largest variation occurred in the burned undercanopy and the difference was over 10. However, there was not a trend of monotonic variation in CN over time during this period. On every surface, CN results showed a decrease followed by an increase over time, and in one case (i.e., undercanopy control) they decreased again in the late testing period. Overall, the effect of fire on runoff was not distinct at this site based on the CN results and the presence of fire did not increase the CN.

The CN results at the Jacob Fire site showed a similar trend to the runoff ratios (Figure 2). The CNs were available for most of the tests on both burned and unburned surfaces for undercanopy as well as interspace areas. However, the differences in CN among various test conditions (e.g., burn undercanopy, burn interspace, and unburned undercanopy) were not constant over time. A temporal change in the CN was observed in the unburned undercanopy surfaces for the test period, which seems to suggest a natural seasonality of runoff potential at this site. In the results for the control interspace, for which CN was available for all tests, CN values during November 2008 and March 2009 (3 to 7 MAB) were generally less than the summer sampling times, which were between June and September. The CNs for interspace control areas were collected between November and March, and not available for most of the other sampling times. Between burned and unburned undercanopy surfaces, the differences in the CN values were not consistent over the three-year testing period. The CN for the burned surface was larger at 3 and 34 MAB, but smaller at 7 and 15 MAB. For burned surfaces of both undercanopy and interspace areas, the results do not reflect a decreasing trend of CN over the three-year period (i.e., a reduction in the effect of fire over time). Overall, the CN results at this site did not clearly show the effect of fire on increasing runoff.

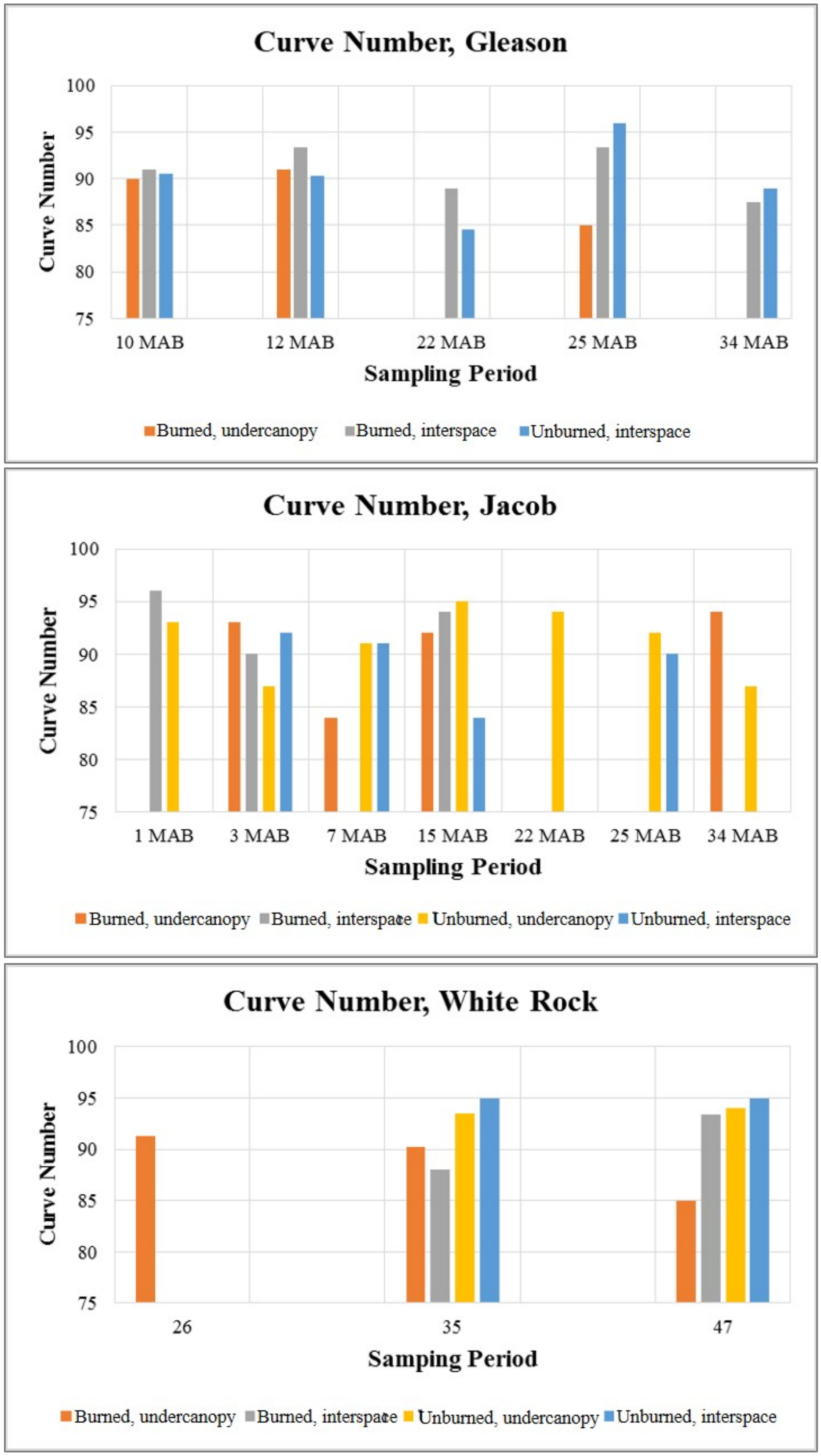

Figure 2. Runoff curve number comparison for the three fire study sites.

At the White Rock Fire site, CNs for unburned sites were not available for the study at 26 MAB. For tests at 35 and 47 MAB, CNs for burned sites were comparable or less than the unburned surfaces, regardless of interspace or undercanopy areas. The CNs for the unburned conditions on both undercanopy and interspace areas remained relatively stable (=93~95) during the third and fourth years, which provides consistent background CNs for this site. For the burned undercanopy areas, CNs showed a decreasing trend over the three-year sampling period, indicating that the surfaces became more permeable over time. The CN values for burned interspace areas were available for the third and fourth years (35 and 47 MAB), and the values showed an increase of CN over time, indicating that the surface became less permeable over time. However, the burned undercanopy areas showed a larger discrepancy from the unburned conditions over time, whereas the burned interspace areas approached the unburned conditions over time. At this point, more data are still needed to better explain the effect of fire on CNs at this site.

In summary, the runoff ratio and CN results at the three fire sites showed complicated patterns. At the Gleason Fire site, runoff ratios on burned interspace were consistently higher than that of the unburned counterpart, indicating increased runoff potential after fire. The CNs for the burned and unburned surfaces were generally similar, whereas interspace areas generally had higher runoff potential than undercanopy areas at most times. At the Jacob Fire site, runoff ratio results indicated ambiguous fire effects that could either increase or decrease runoff potential. Runoff ratios for burned versus unburned interspace areas had larger discrepancies (i.e., effects of fire) than the undercanopy areas. There were some differences between CNs for burned and unburned areas, but the effect of fire on runoff potential was not consistent over time. Both runoff ratio and CN results indicated more prominent fire effects in the early sampling period, and results for the unburned surfaces complicated temporal variation, which suggests possible seasonality at the Jacob Fire site when the effects of fire are absent. At the White Rock Fire site, both runoff ratios and CNs were different between burned and unburned surfaces, with the burned surfaces showing lower runoff potential. Runoff ratios showed more significant temporal changes on unburned surfaces, whereas the CN results showed more temporal changes on burned surfaces.

Soil Properties

Soil properties such as dry bulk density, organic matter, and soil texture were measured at all three fire study sites. Soil bulk density data were collected at all three study locations, but organic material and soil texture were not measured at the Jacob Fire site.

<u>Soil Texture</u>

Soil texture is an important indicator for soil classification and soil properties, such as infiltration and permeability rates as well as moisture holding capacity. Soil texture was determined by analyzing the particle size of soil samples collected at the study sites. At the Gleason Fire site, soil sample analysis showed that the predominant soil texture was sandy loam prior to the burn (-0.13 MAB), with some loamy sand and loam (Figure 3). Soil particle size distribution had large spatial variations on all surfaces. On average, soil texture in the undercanopy and interspace areas were similar before the fire, but undercanopy soil particle size distribution had larger variability, including samples located in the loam soil texture class. During the study period, there was a clear tendency of the soil texture to become finer over time. After three years, approximately half of the samples were classified in the loam or silt loam category. The sandy component in the soil samples decreased over time, whereas the silt component increased. This suggests that as burned soil aggregates broke apart, the finer particles were released and preferentially transported by wind and water, and then accumulated over time on the surface. The clay component was relatively stable during the sampling period. Burned and unburned soil texture was similar, but burned surfaces typically had slightly finer soils. All soils included a significant portion of gravel (14.6 percent on average) and interspace soils had slightly more gravels than undercanopy soils (14.9 percent versus 14.3 percent on average for the whole sampling period). This trend persisted over the entire study period.

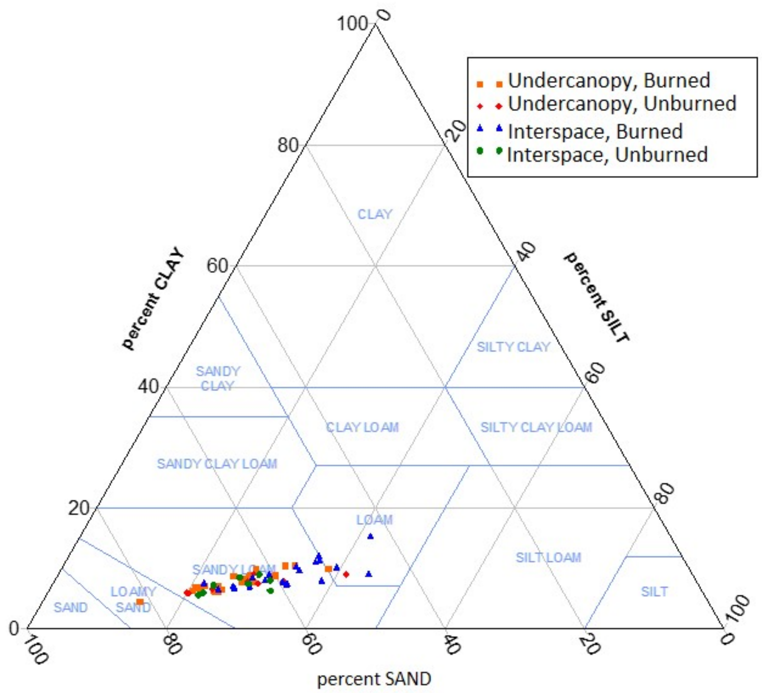

Figure 3a. Gleason soil texture, -0.13 MAB.

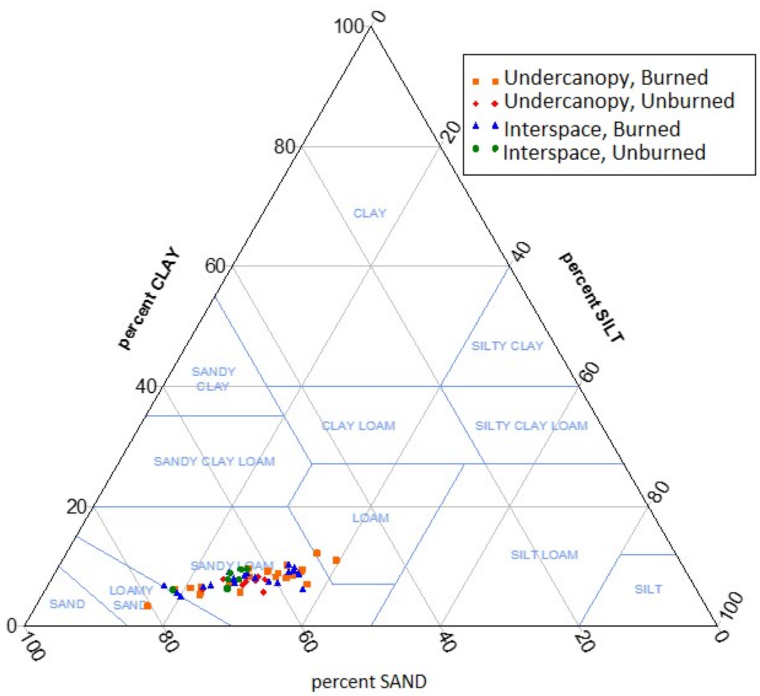

Figure 3b. Gleason soil texture, 0.16 MAB.

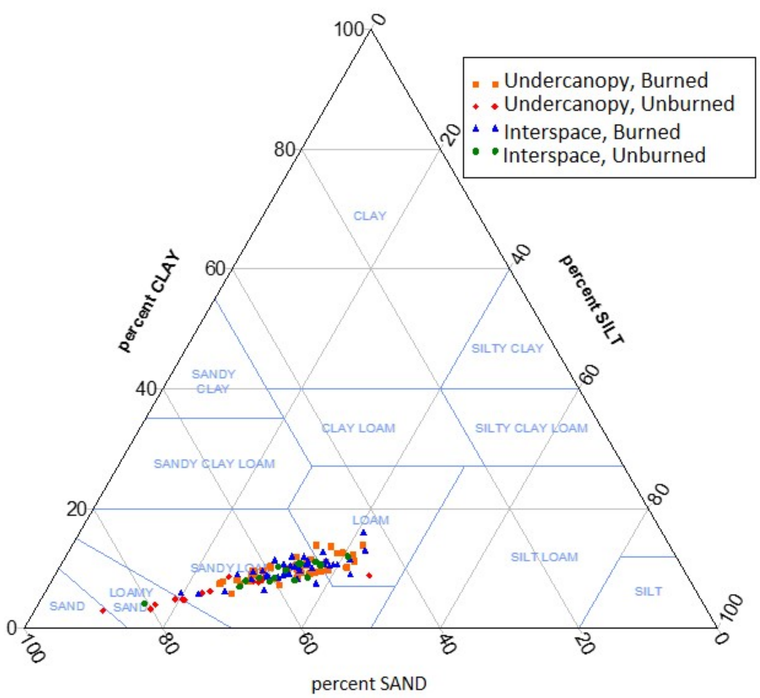

Figure 3c. Gleason soil texture, 10-12 MAB.

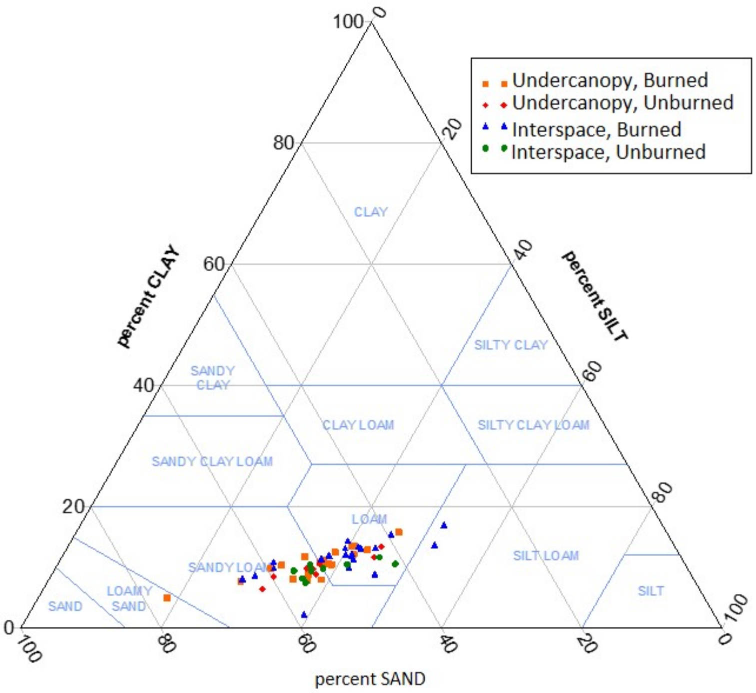

Figure 3d. Gleason soil texture, 22-25 MAB.

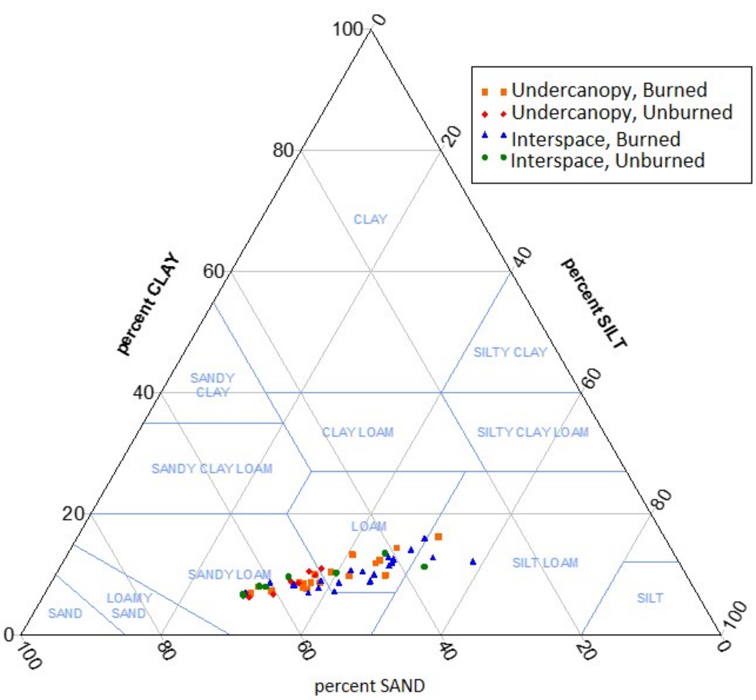

Figure 3e. Gleason soil texture, 34 MAB.

At the White Rock Fire site, soil textures were mainly sandy loam and loam, and they exhibited large spatial variations (Figure 4). Between burned and unburned surfaces, soils on the burned surface were finer, again suggesting that burned soil aggregates were broken apart by the heat. The soil sample analysis did not show a trend of texture change over time and the spatial variation of the soil texture also did not change over time. Given the greater time between the fire and when the White Rock study was performed, it is likely that any prior accumulation of fine particles had been transported away. Soils at the White Rock Fire site contained 24 to 31 percent of gravels, which is more than at the Gleason Fire site. Interspace soils contained 28 to 40 percent gravels, which was more than undercanopy soils (19 to 28 percent). This difference was consistent throughout the three-year sampling period.

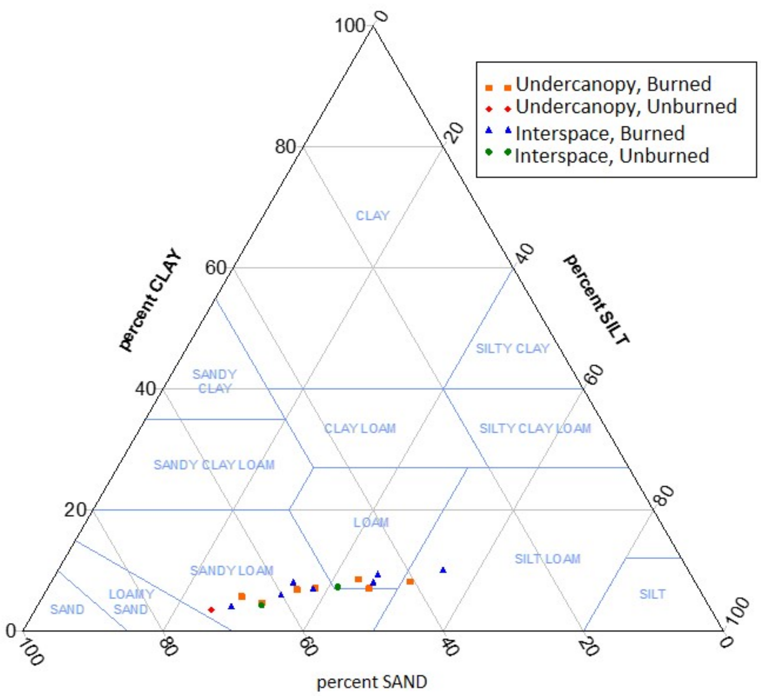

Figure 4a. White Rock soil texture, 26 MAB.

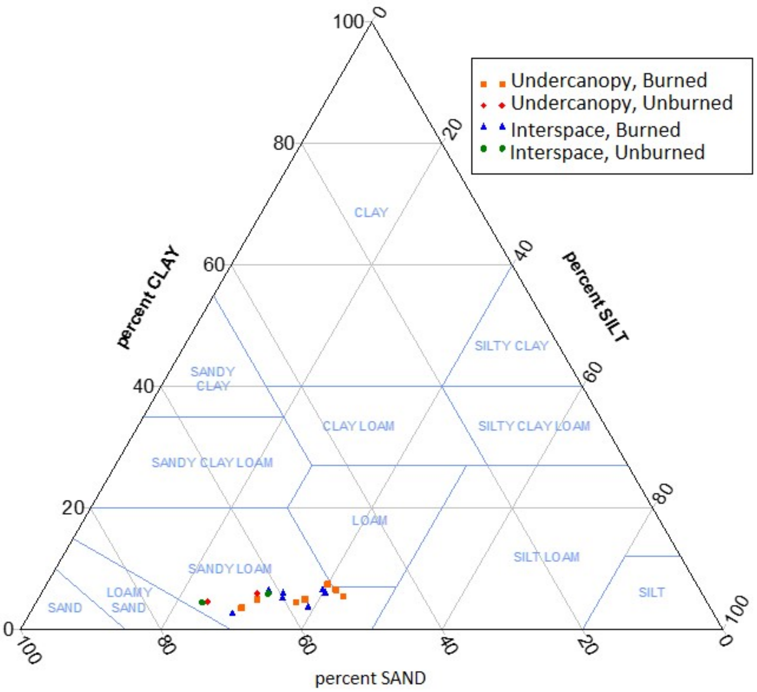

Figure 4b. White Rock soil texture, 35 MAB.

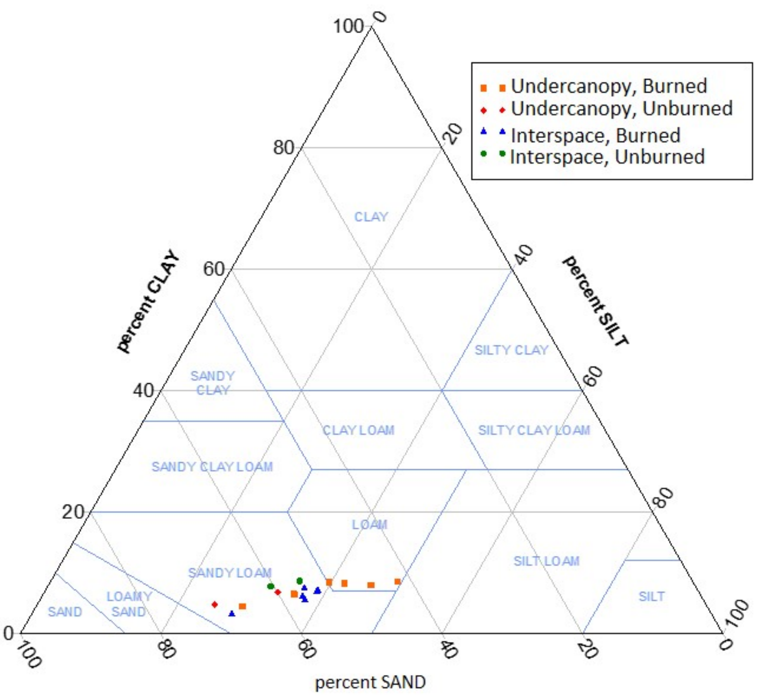

Figure 4c. White Rock soil texture, 47 MAB.

Organic Matter

Soil organic matter content can potentially affect soil hydraulic properties and the related hydrologic processes because organic matter can increase infiltration and reduce water runoff and soil erosion. Soil organic matter was measured in the lab from the soil samples collected during field tests. Soil organic matter was measured on soils from the Gleason Fire and the White Rock Fire, but no measurements were collected at the Jacob Fire.

Soil organic matter content at the Gleason Fire site ranged from two to five percent of the total soil mass, but it was typically less than three percent (Figure 5). There was no clear difference in the organic matter content of burned or unburned soils. Undercanopy soil generally had higher organic matter content. There was a slight increase in organic matter content in the late sampling periods (third year, 25-34 MAB) compared with the early sample periods. The variability across different surfaces was smallest immediately after the fire, most likely because it was burned, but it could return to the pre-fire conditions over time.

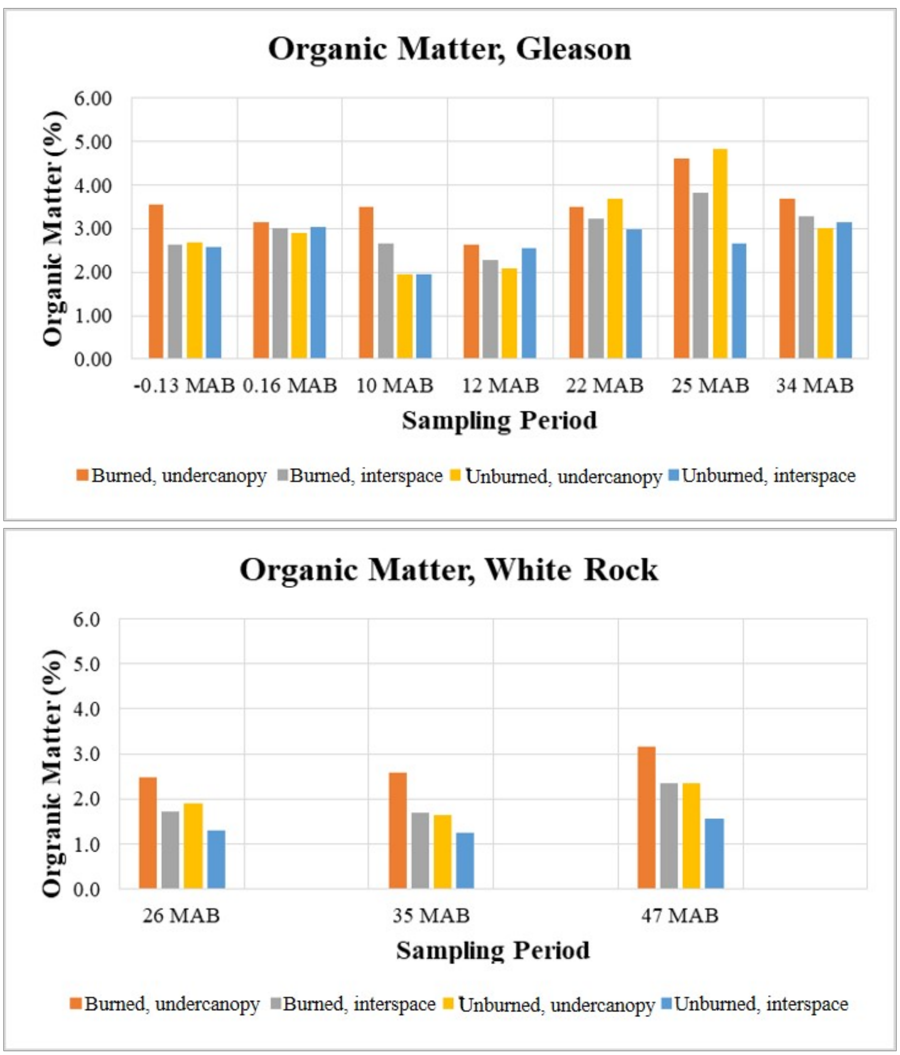

Figure 5. Average soil organic matter comparison. Organic matter was not collected at the Jacob Fire.

Soil organic matter content at the White Rock Fire site ranged from one to three percent, which was slightly lower than the Gleason Fire site. This likely reflects the nature of the Mojave Desert ecoregion where lower total annual precipitation results in less vegetation cover compared with the other two sites located in different ecoregions. Across all soil surfaces, the soil organic matter was similar from year to year. The burned surfaces at both fire sites had a higher organic matter content than unburned surfaces, which was likely a result of the incomplete combustion of plants. The location of burned plants also increased the soil organic matter in undercanopy soils compared with interspace soils. However, although the relative differences in soil organic matter between sample locations (e.g., undercanopy or interspace) were up to 50 percent, the differences were typically less than 1 percent.

Bulk Density

Soil bulk density was measured at each site because soil bulk density can be an important soil property related to infiltration. Generally, soils with lower bulk densities are more absorbent and have lower runoff values than soils with high bulk densities. At the Gleason Fire site, soil bulk density showed a noticeable decrease from the pre-fire conditions to the immediate post-fire (0.16 MAB) conditions. Bulk density increased to the pre-fire level approximately one year after the burn. In the second and third year after the burn, soil bulk density showed a slight decrease. This could be related to the changes in soil organic matter and soil texture. The soil bulk density of both burned and unburned surfaces showed a similar trend of variation with time over a nearly three-year period, which suggests that there may be a natural seasonality in soil bulk density.

Soil bulk density at the Jacob Fire site was measured at 22, 25, and 34 MAB. The values of bulk density ranged from 1.48 (92.4 pound/ft^{-3}) to 1.82 g cm^{-3} (113 pound/ft^{-3}), and the majority of measurements were below 1.6 g cm^{-3} (99.8 pound/ft^{-3}). Overall, no substantial difference in bulk density was observed between burned and unburned surfaces for both undercanopy and interspace areas. The average bulk density in the third year was slightly larger than the second year after the burn for both burned and unburned surfaces (Figure 6).

At the White Rock Fire site, soil bulk density did not show a difference between burned and unburned surfaces. The difference between the second and third years of study was because of the change of sampling strategies. Wet soil samples were collected after the infiltration tests in the second year, whereas dry samples were collected in the third year. The collection of wet samples may result in soil compression, which may change the bulk density.

On average, soil bulk density at the Jacob Fire site was higher than the other two sites, for both burned and unburned surfaces, and both undercanopy and interspace soils. The increased soil bulk density may be one of the reasons that the runoff ratio was higher at the Jacob Fire site than both the Gleason Fire and White Rock Fire sites, because bulk density is usually negatively correlated to soil hydraulic conductivity. Because soil hydraulic conductivity cannot be derived for the Jacob Fire site based on the available data, the bulk density can be used as an indicator to estimate the range of soil hydraulic conductivity at this site compared with the results for the other two sites.

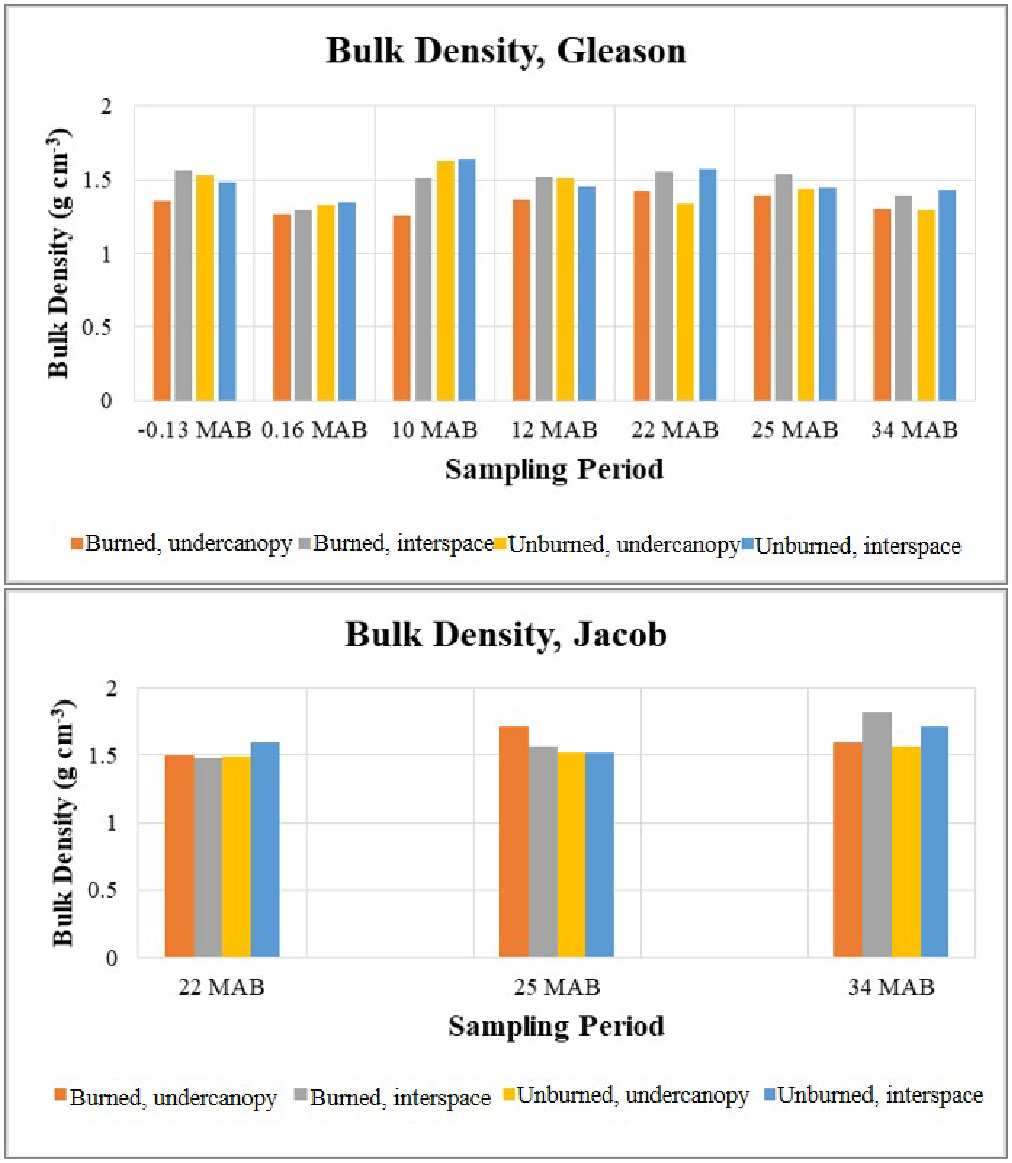

Figure 6. Average soil bulk density comparison for the three fire study sites. Note the different scales of the y-axes.

Soil Hydraulic Properties

The saturated soil hydraulic conductivity (K_s) was used as the representative soil hydraulic property to compare the three fire sites. The K_s describes the capability of soil to transmit water when subject to a hydraulic gradient. It is directly related to infiltration and runoff generation during storm events. The measurement of K_s was performed by using the mini-disk tension infiltrometer (MDTI). At the Gleason Fire site, the test was performed seven times from before the prescribed burn (i.e., -0.13 MAB) to 34 MAB. At the White Rock Fire site, the test was performed at 26, 35, and 47 MAB. This test was not conducted at the Jacob Fire site. Figure 7 shows the K_s results, including the mean, maximum, and minimum for all surfaces (i.e., burned and unburned, and undercanopy and interspace) at the Gleason Fire and White Rock Fire sites.

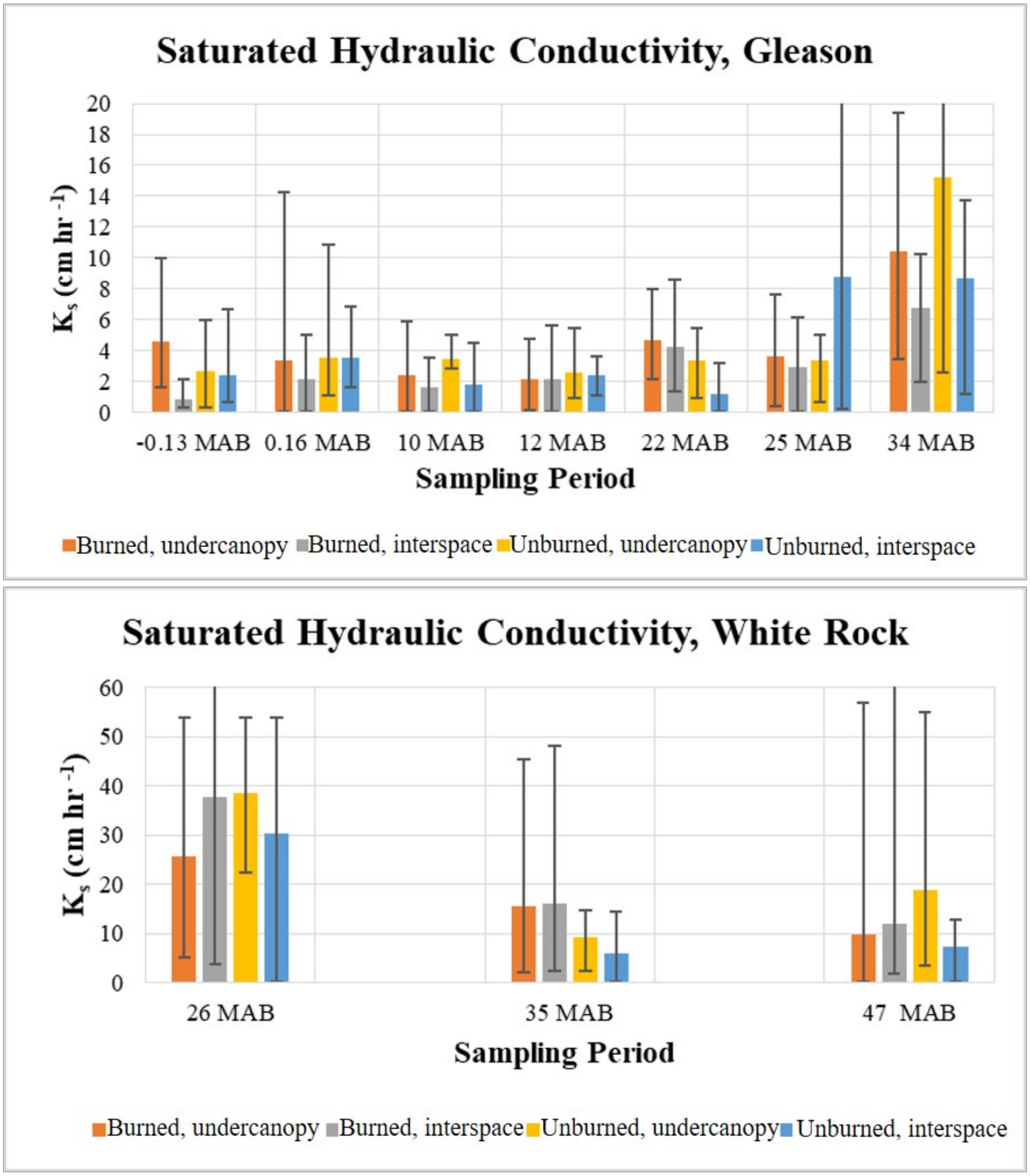

Figure 7. Saturated soil hydraulic conductivity (K_s) for the Gleason Fire and White Rock Fire sites. Error bars are standard deviation. Note the different scales of the y-axes.

At the Gleason Fire site, the K_s of the undercanopy areas on burned surfaces showed a decreasing trend from the pre-fire condition to one year after the fire. The variability also decreased despite the greater variability that occurred immediately after the fire (0.16 MAB). The pre-fire K_s values of burned surfaces were lower with less variability compared with all of the other tests. These values became slightly higher after the fire and remained relatively constant during the first year post-fire. The K_s values of unburned surfaces were relatively constant from pre-fire to one year post-fire and did not vary substantially across undercanopy and interspace areas. In the second and third years of the study, K_s as well as the K_s

variability increased in general regardless of burned or unburned surfaces. The results suggest that fire may have a greater effect on K_s located on undercanopy areas than interspace areas, and the effects of fire may last for approximately one year. Soil hydraulic conductivity may also have a natural seasonal variability.

At the White Rock Fire site, the average K_s showed differences during sampling periods after the fire, with larger K_s values at 26 MAB than 35 MAB and 47 MAB, whereas the latter two measurements had relatively small differences. These results suggest the variation of hydraulic properties may be seasonally influenced. The average K_s values did not show substantial differences between burned and unburned surfaces, but burned surfaces seemed to have larger variabilities in K_s.

Fire did not exhibit a clear influence on runoff generation at these sites. At the Gleason Fire and Jacob Fire sites, fire caused a highly variable response of runoff potential. At the White Rock Fire site, fire generally resulted in reduced runoff potential. For the surfaces where the effect of fire on runoff potential was present, the effect did not necessarily reduce over time. On average, the runoff ratios of both the burned and unburned surfaces at the Jacob Fire site were much higher than at both the Gleason Fire and White Rock Fire sites, which suggests higher runoff potential and possible flooding risks in the Mojave-Great Basin transition ecoregion. Additionally, the results of soil properties tests suggested that fire may potentially affect soil texture, which will affect hydrologic processes such as infiltration, runoff, and evapotranspiration, as well as geomorphic processes such as erosion and deposition.

POST-FIRE DUST EMISSION AND THE POTENTIAL FOR SOIL EROSION BY WIND

Methods

Similar measurement protocols were used at each of the three fire sites, but changes in the methods were warranted in some cases because of environmental conditions. Moreover, because the three fire sites were all slightly different, there were some variations in the selection of the specific measurement locations. Table 2 provides an overview of the wind erosion measurements and sampling similarities and differences among the fire sites. The PI-SWERL was used to measure wind erosion potential at all of the sites, and material suspended by the PI-SWERL was sampled onto filter media.

Across all sites and years, the actual PI-SWERL measurement cycle that was used was held constant (Miller *et al.*, 2013a,b). The main differences among the sites pertain to how the measurement locations were selected, the frequency of the measurements, and the methods for collecting the filter media.

Table 2. Summary of wind erosion measurement methods across fire study sites.

Fire Name, Ecoregion	**PI-SWERL Test Protocol**	**Filter Collection** Instruments, flow (lpm), filters used	**Sampling Locations**	**Dates Spanned by Testing & Test Timing in Months After Burn (MAB)**
Gleason, Great Basin Desert	**Hybrid 4,000** (ramp in between steps held at 2,000 RPM for 60 s, 3,000 RPM for 90 s, and 4,000 RPM for 90 s).	Dual MiniVols (Airmetrics), 5 lpm, simultaneous collection of 47 mm (1.85 in) Teflon and 47 mm (1.85 in) quartz fiber	Unburned: Interspaces between shrubs Burned: Interspaces between shrubs and plant mounds/undercanopy areas	August 2009-June 2012 0, 1, 9 , 10, 12, 21, 24, and 34 MAB
Jacob, Mojave-Great Basin Transition	**Hybrid 4,000**	Dual MiniVols (Airmetrics), 5 lpm, simultaneous collection of 47 mm (1.85 in) Teflon and 47 mm (1.85 in)	Unburned: Interspaces between shrubs Burned: Interspaces between shrubs segregated by location on ridge or wash Plant mounds/undercanopies, separated by ridge or wash	September 2008-December 2011 1, 3, 6, 13, 21, 24, 34, 36 MAB
White Rock, Mojave Desert	**Hybrid 4,000**	Custom filter collection with PM_{10} cyclone size separator, 16.7 lpm, either 47 mm (1.85 in) Teflon and 47 mm (1.85 in) at any one time.	Unburned: Interspaces between shrubs Burned: "old burn" or "new burn" AND interspace or mound/undercanopy For both burned and unburned areas, an effort was made to include a variety of soil unit ages	June 2014-April 2016 26, 35, and 48 MAB

<u>Measurement Frequency and Location Selection</u>

At each fire site, the intent was to quantify the difference in wind erosion potential between the burned and unburned landscape. On burned sections of the landscape, two types of measurement locations were identified: interspaces and undercanopy. On unburned sections of the landscape, the undercanopies were not accessible because the trees/shrubs were still intact. Therefore, measurements on unburned sampling locations were restricted to interspace regions in all cases.

At the Jacob Fire site, burned sampling locations (whether interspace or undercanopy) were further separated by their proximity to the local ridgeline ("ridge") or the drainage ("wash") to determine if topographic locations on the landscape contributed to the different levels of emissions. At the White Rock site, there were two designations: old burn (OB) and new burn (NB). Old burn refers to the Cabin Fire that burned approximately 25 years prior to the new burn, which refers to the region that burned in the 2012 White Rock Fire. Additionally, sampling at the White Rock Fire was constrained because the relative age of the geomorphic surface was estimated and that information was associated with the sampling locations. However, after the sampling scheme was determined, later information suggested that the ages of the geomorphic surfaces were different than were originally determined, and therefore of the most information on the surface ages was not used.

The Jacob Fire site was measured with the greatest frequency, with a total of eight post-fire measurements over 45 months and measurements were more frequent at the beginning of the study. The Gleason Fire site was not measured with the same frequency (i.e., six post-fire measurements over 34 months) because of snowfall during the winter months and generally wetter ground conditions at the higher elevation of the Gleason Fire site, which made it difficult to access the site during periods that were dry enough for wind erodibility measurements. Wind erodibility measurements at the White Rock Fire site were conducted once per year.

Results and Discussion

The Gleason Fire had substantially higher post-fire emissions compared with both the Jacob Fire and the White Rock Fire. Figure 8 shows an example of emissions during the initial post-fire sample period at each fire site. The Gleason Fire site was more prone to wind erosion immediately after the fire event and wind erodibility decreased over time. The PM_{10} emissions from burned plant undercanopies were higher at all values of revolution per minute (RPM) than either the White Rock Fire or the Jacob Fire sites, by up to two orders of magnitude in some cases. The Gleason Fire affected the potential for wind erodibility in at least two ways. First, the burning of large shrubs and small trees resulted in the exposure of relatively large plant undercanopy areas that were previously protected from wind by plant canopies. Second, the burning of large numbers of plants and trees changed the overall aerodynamic roughness of the landscape. The PI-SWERL provides an estimate of emissions if the wind immediately above the test surface is inducing the shear stress, as calculated in Etyemezian *et al.* (2014). Therefore, the PI-SWERL tests could indicate a surface that is highly wind erodible at some level of shear stress. However, it also could be possible that those levels of shear stress are never experienced near the ground because of the large-scale landscape roughness, which decreases the momentum of the wind, and therefore causes little friction over the ground. In such cases, the soil surface under burned plant canopies would become inherently more wind erodible after a fire, but they still might not experience wind erosion because the large-scale roughness protects the landscape as a whole. In the case of the Gleason Fire site, and other sites with relatively thick vegetative cover, when the undercanopy becomes exposed because of the burning of a tree or shrub, the large roughness elements on the landscape (e.g., trees and shrubs) also disappear or diminish in surface area because of the fire, and therefore vegetation is unable to offer protection from the winds aloft. A reduction in surface roughness is likely much higher at the Gleason Fire site

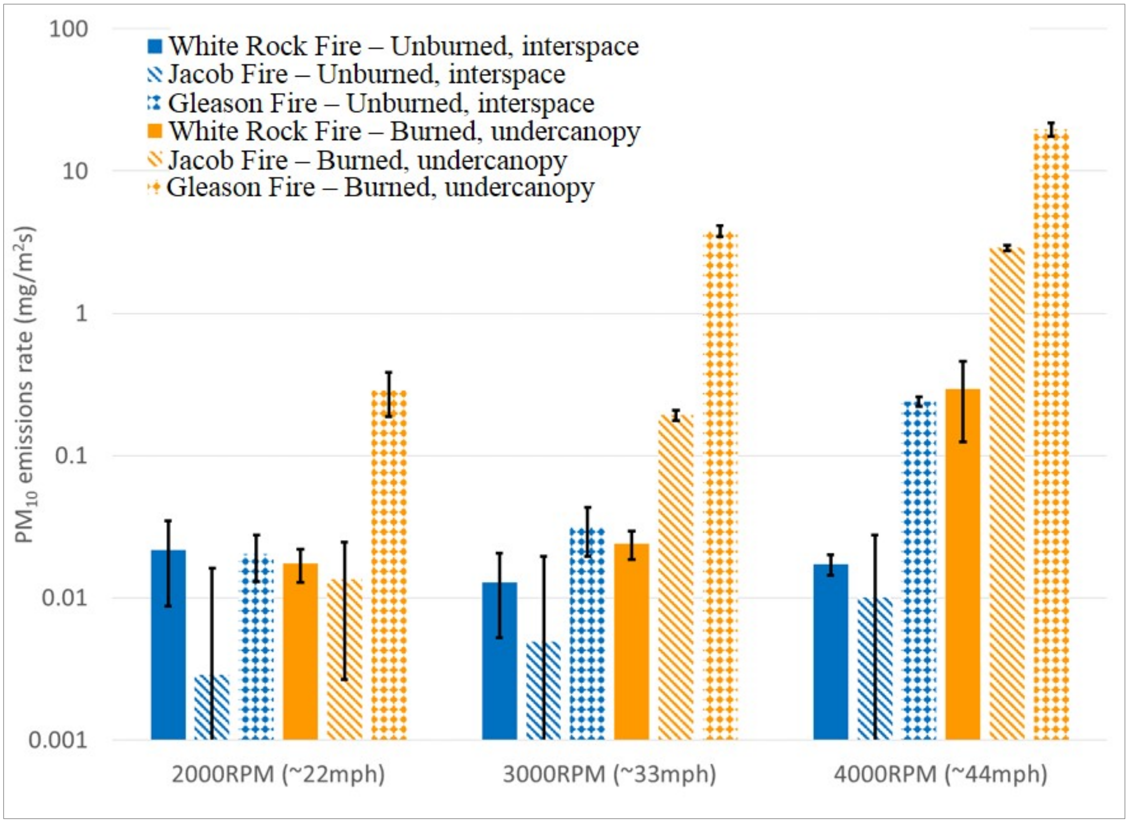

Figure 8. Example of PM_{10} emissions under burned canopies and unburned interspaces after the fire event. Vertical bars represent standard errors of mean.

compared with the White Rock Fire site, which has low-stature shrubs and typically less overall vegetative plant cover. The loss of vegetation because of fire at the White Rock Fire site was not as significant as at the Gleason Fire site in terms of additional exposure of the ground to wind shear.

Wind erosion after the Jacob Fire was variable. For example, emissions from interspace regions were quite low at all values of equivalent wind shear (scales with PI-SWERL RPM) that were applied by the PI-SWERL. Emissions at the Jacob Fire site from burned undercanopy areas at the 3,000 RPM level were higher than the burned undercanopy areas at the White Rock Fire site, but much lower than at the Gleason Fire site. At the Jacob Fire site, the 4,000 RPM level emissions were approximately an order of magnitude greater than the White Rock site, but an order of magnitude less than at the Gleason Fire site. However, the 4,000 RPM emissions reported for the Gleason Fire burned undercanopy areas were likely underestimated because for several PI-SWERL tests, the instrument had to be stopped before completing the 4,000 RPM measurement because of exceedingly high PM_{10} concentrations that exceeded the upper limit of the DustTrak instrument used by the PI-SWERL. Tests during which the emissions were especially high could therefore not be included in the average emissions estimated for the 4,000 RPM level for the Gleason Fire measurements.

The White Rock Fire site, which receives the least amount of annual rainfall, appeared to be least affected by the fire in terms of wind erodibility. The unburned interspace, which was gravelly and sloped, exhibited little potential for wind. However, even the undercanopy areas of plants in burned areas were resistant to wind erosion at the equivalent of 3,000 RPM shear stress (roughly 33 miles per hour [mph] [53 kilometers per hour (kph)] wind speed at 32 ft [10 meters] above ground surface). At the 4,000 RPM level (approximately 44 mph [70 kph]), wind erodibility is elevated in the burned plant undercanopies compared with the interspace areas, which indicates that fire makes these regions more susceptible to wind erosion. Conclusions about the effect of fire drawn from the observation of relatively low wind erosion potential in the interspace regions are tempered somewhat by the fact that there was a delay between when the fire occurred and when measurements at the White Rock Fire site began (approximately 26 months).

An important question that led in part to the series of measurements conducted at these NNSS-relevant fire study sites is: "Following a fire event, how long is the burned surface prone to enhanced wind erosion?" One of the lessons learned during the studies is that in addition to the effect of fire, there are environmental parameters and conditions that have a profound effect on the landscape's potential for wind erosion. Some of these parameters, such as soil moisture, are easier to quantify. However, others such as vegetative cover, soil microphysics, and the degree of biological disturbance (e.g., from rodents, insects, and grazing animals) are difficult to quantify. Figure 9 summarizes the data from the Jacob Fire site by surface type. Even the unburned surfaces exhibited variations of approximately an order of magnitude in the PM_{10} emission potential. In some cases, a causal relationship between precipitation (presumably increased soil moisture content) and reduced wind erosion potential can be inferred. For example, the reduction in PM_{10} emissions at 6 MAB could be inferred to be the result of substantial precipitation (as recorded in nearby Rachel, Nevada, at 5 MAB). However, the precipitation does not explain why emissions at 24 MAB were also comparatively low. Moreover, the relationships between parameters, such as rainfall, and the potential for emissions are qualitative and cannot be used as *a priori* prognostic indicators. That is, environmental conditions have a significant effect on the temporal trends in PM_{10} emissions and care should be taken to avoid drawing conclusions based on apparent short-term trends.

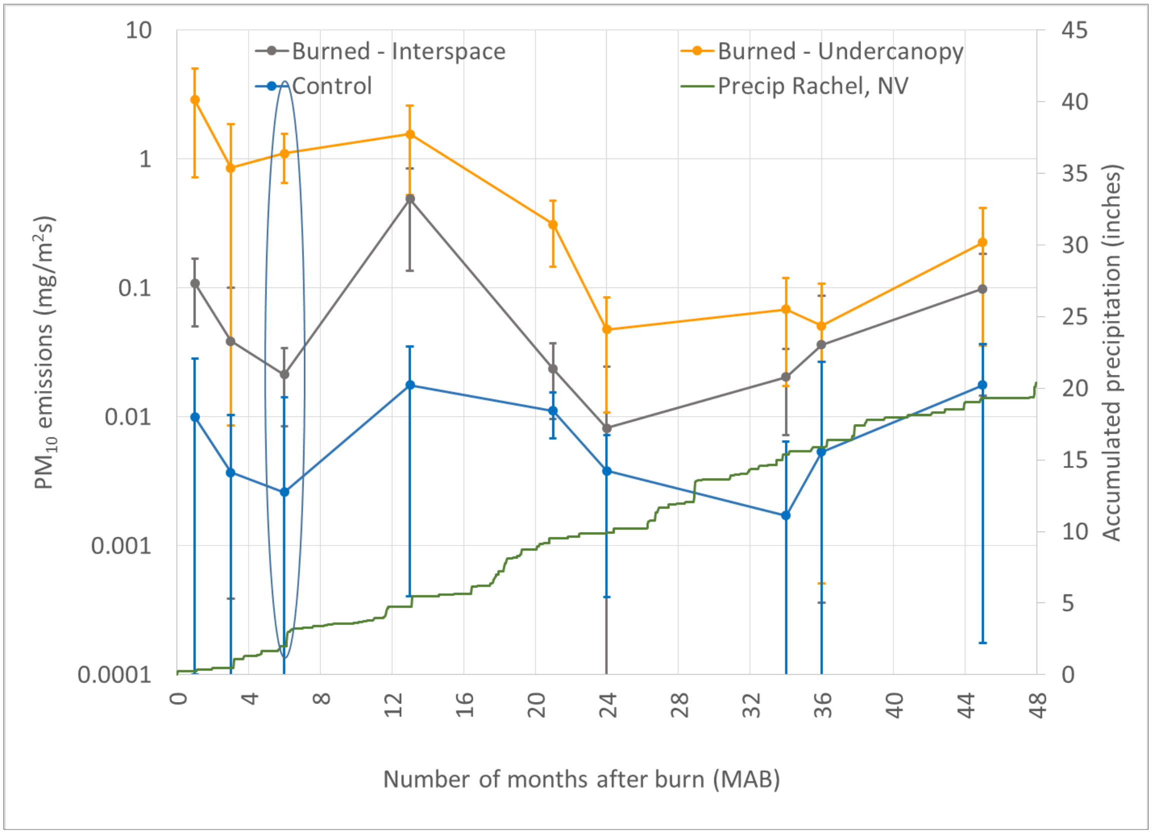

Figure 9. PM_{10} emissions at 3,000 RPM (33 mph [53 kph]) at the Jacob Fire site by burn surface type. The ellipse shows the extreme emissive differences 6 MAB as emissions then begin to attenuate. The green line shows the accumulated precipitation (right axis).

To mitigate the effects of varying environmental parameters on the temporal variations in the emissions, the data were normalized from burned areas by the emissions from unburned areas. An underlying assumption of normalization is that whatever environmental factor affects emissions from the burned areas of a fire site, that same factor should have a similar effect on emissions from unburned areas. Figure 10 shows how some of the temporal noise is removed by normalization, which allows for a clearer understanding of how emissions from burned areas change over time when compared with Figure 9. In Figure 10, PM_{10} emissions from burned plant undercanopy areas attenuate over the study period, with most of the attenuation occurring over the first two years. Burned interspaces exhibited a similar trend of attenuation. Within the uncertainty of the data, PM_{10} emissions from burned interspaces were not greatly different from PM_{10} emissions from unburned interspace areas two years after the burn, which is evidenced by similar normalized emission values.

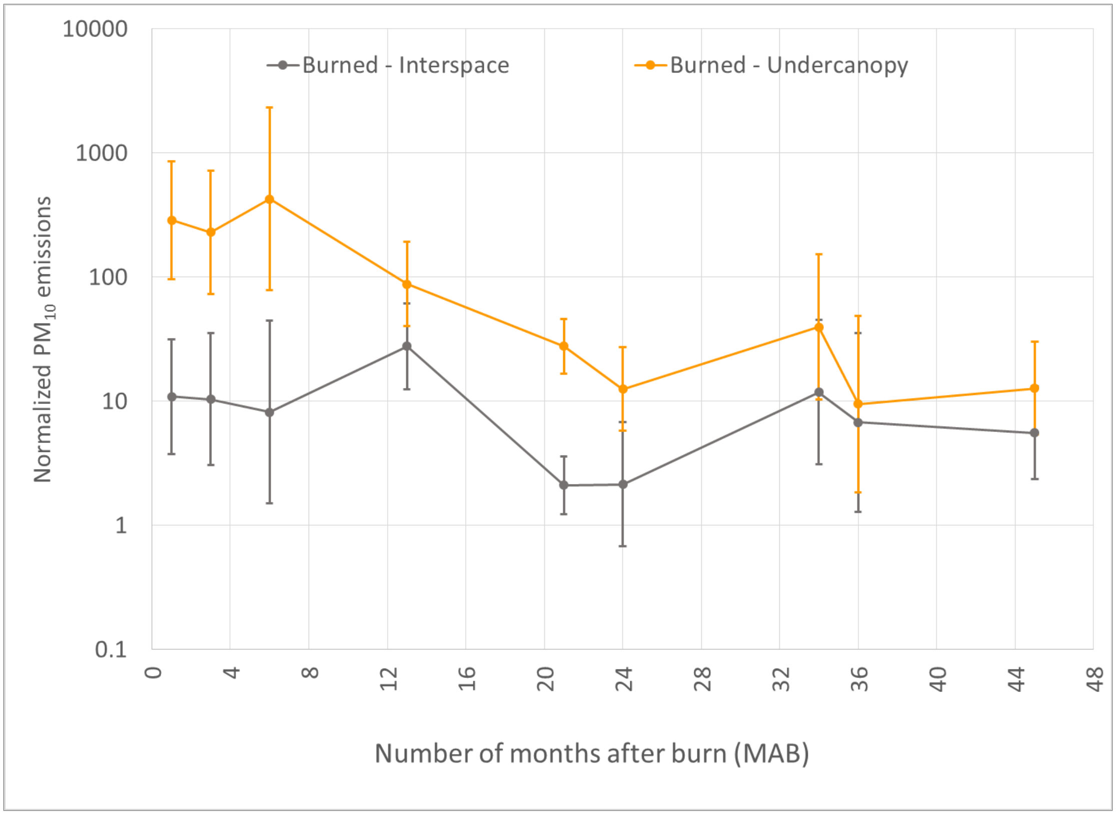

Figure 10. Normalized PM_{10} emissions at 3,000 RPM (33 mph [53 kph]) from burned areas at the Jacob Fire site. Emissions from burned areas have been normalized by emissions from unburned areas. The vertical bars represent the standard error of mean.

The baseline emissions from plant undercanopies are not known because the presence of unburned plants prohibits access with the PI-SWERL instrument. Baseline emissions from interspaces were used to normalize the undercanopy data, but this may introduce uncertainty into the quantitative analyses and the magnitude of the uncertainty is difficult to estimate. However, the temporal trend of emissions from the unburned areas was also shown in the emissions of the burned undercanopies (Figure 9). The normalized emissions from the burned undercanopies trend toward the normalized emissions of the burned interspaces, which indicates that the emission from the burned undercanopies were reduced at a lower, longer-term level. Overall, data from the Jacob Fire site indicated within two years of the fire event, PM_{10} emissions in both the interspace regions and plant undercanopy areas in the burned areas have attenuated to within a factor of 10 of the unburned areas, which are very low emitters of dust by arid landscape standards. Sweeney *et al.* (2011) report PM_{10} PI-SWERL-based emissions from various Mojave Desert landforms that range between approximately 0.01 mg/m^2/s ([0.2 × 10^{-3} pounds/ft^2/s] salt crusted playa) to 0.4 mg/m^2/s ([0.9 × 10^{-2} pounds/ft^2/s] sand dunes and washes), with individual measurements as high as 29 mg/m^2/s (0.69 pounds/ft^2/s).

Normalized PM_{10} emissions from the Gleason Fire measurements were more variable (Figure 11). The data for the first 24 MAB showed no indication that normalized emissions from burned plant undercanopies or burned interspaces were trending toward reduced emissions. However, emissions attenuate by the third year at 34 MAB, but the variability in the temporal data and the uncertainties of the measurements did not preclude the reduced 34 MAB normalized emissions as being measurement uncertainty. Data for the PI-SWERL 4,000 RPM (nominally 44 mph [70 kph] winds) step (Figure 12) and 2,000 RPM step (not shown) did not indicate attenuation had occurred over the three-year measurement time period.

At the White Rock Fire site, there was little difference between burned interspace areas and unburned interspace areas for the PI-SWERL 2,000 and 3,000 RPM steps. It should be noted that the study was started two years after the fire event and that in the case of the Jacob Fire, this was sufficient time for the burned interspace areas to return to near pre-fire emission levels. Figure 13 shows the normalized emissions for burned plant undercanopy areas at the White Rock Fire site. The PM_{10} emissions were elevated by over one order of magnitude in the undercanopy areas compared with interspaces during all three sampling campaigns. Normalized emissions were higher at 35 MAB than 26 MAB and 48 MAB, which suggests the presence of natural variation in emission strength driven by uncharacterized parameters. At 48 MAB, emissions were reduced compared with emissions at 35 MAB because of the widespread presence of short annual grasses. Differences between 35 MAB and 26 MAB may be because of differences in environmental conditions.

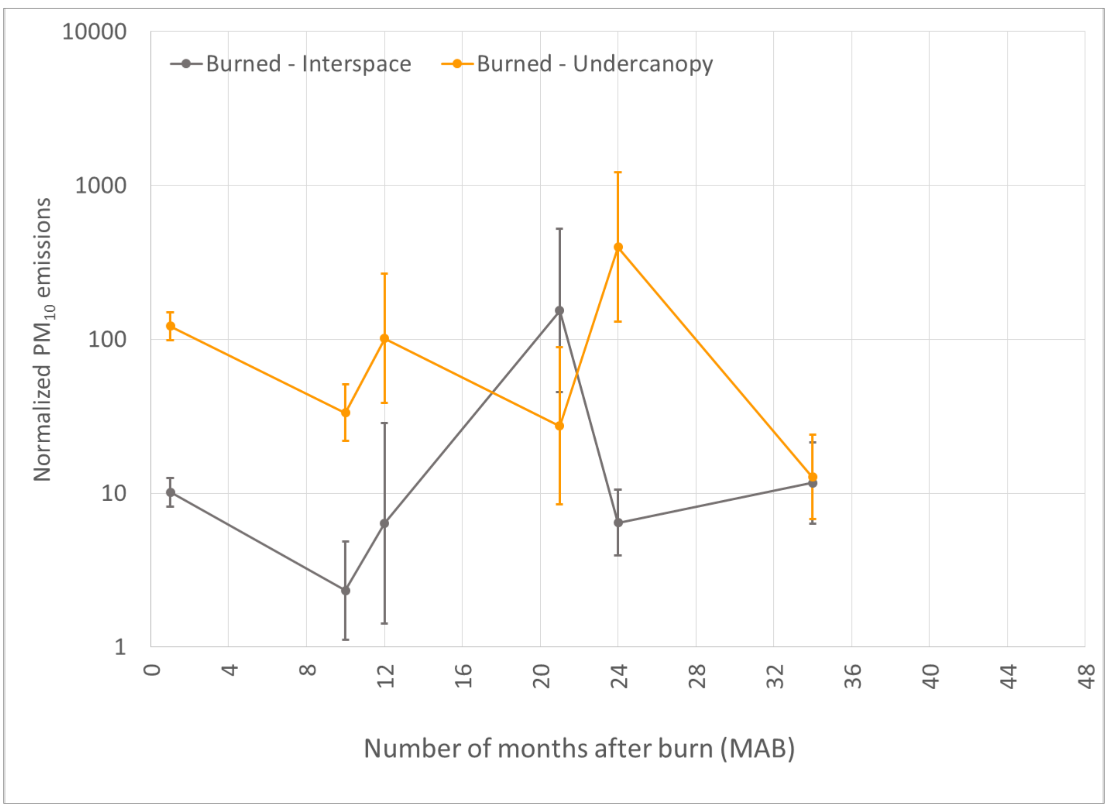

Figure 11. Normalized PM_{10} emissions at 3,000 RPM (33 mph [53 kph]) from burned areas at the Gleason Fire site. Emissions from burned areas have been normalized by emissions from unburned areas. The vertical bars represent propagated standard errors.

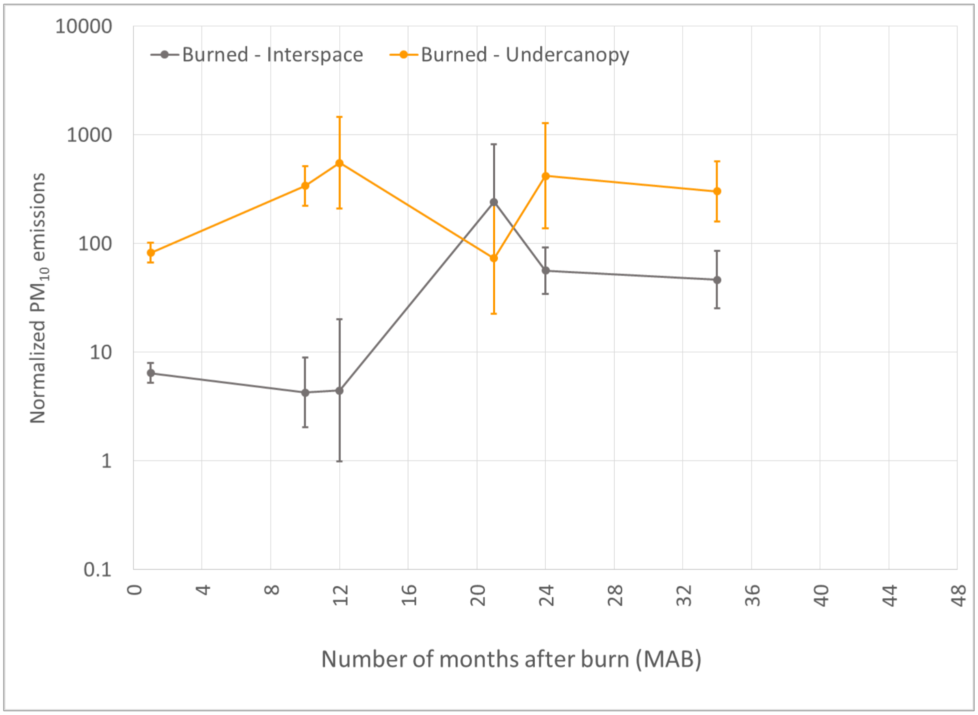

Figure 12. Normalized PM_{10} emissions at 4,000 RPM (44 mph [70m kph]) from burned areas at the Gleason Fire site. Emissions from burned areas have been normalized by emissions from unburned areas. The vertical bars represent propagated standard errors.

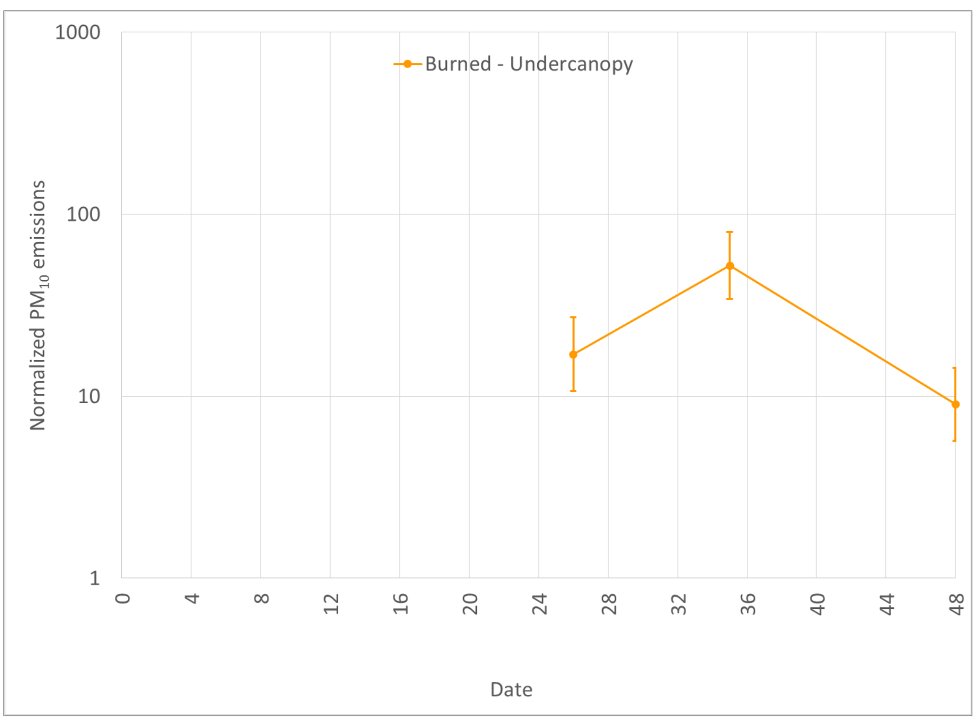

Figure 13. Normalized PM_{10} emissions at 3,000 RPM (33 mph [53 kph]) from burned areas at the White Rock Fire site. Emissions from burned mounds have been normalized by emissions from interspace areas. The vertical bars represent propagated standard errors.

The White Rock fire site data consisted of measurements of PM_{10} emissions from burned areas on soils of varying geomorphic ages, as well as from two different fire events. Miller *et al.* (in draft) examined the differences in emissions between burned plant undercanopy areas from a 2012 fire (NB) and those from a fire that was approximately 25 years old (OB). It was determined that the NB emissions from plant undercanopies were approximately an order of magnitude higher than the OB emissions, indicating PM_{10} emissions from the OB undercanopy areas have attenuated over timescales of two decades. However, Miller *et al.* (in draft) also reported that the age of the geomorphic surface on which the fire occurred appears to affect emissions. Generally, the younger the soil units had higher emissions from the burned plant undercanopy areas because the younger soils on these sites may have a higher concentration of smaller sand and dust particles than older soils. The overall PM_{10} windblown emissions from the White Rock Fire site were low compared with the Great Basin and transition ecoregions.

One important aspect of the PM_{10} emissions measured from burned fire sites is the origin of the material that is suspended. This is especially applicable to the goals of understanding the potential for radionuclide transport from contaminated soils of the NNSS and NTTR in the event of a wildfire. The relevant question is: "When elevated emissions are measured following a fire event, how much of those emissions originate from the soil mineral constituents and how much originate from burned plant material?" Because plants do not uptake and integrate plutonium into their tissue, it is reasonable to expect that PM_{10} emissions from burned plant material is relatively benign and not likely to contribute directly to the wind-borne transport of contamination. However, emissions of PM_{10} associated with soil minerals is more likely to be associated with the preferential binding of radionuclide contamination to fine soil particles. The PI-SWERL does not by itself provide an indication of the origin of wind-suspendible PM_{10}, but rather it provides an estimate of the total.

Differentiating between emissions of burned plant material and underlying soil mineral material was the main objective of collecting filter samples and characterizing the chemical constituents of the particulates that are emitted during PI-SWERL testing. During the earlier Jacob Fire and Gleason Fire studies, two types of filters (one Teflon and one quartz fiber) were simultaneously deployed on the MiniVol, low-volume (five liters per minute [lpm]) samplers to collect particulates from the PI-SWERL exhaust. A common problem encountered was the overloading of filters, which added a large amount of uncertainty to the ensuing chemical analyses. This occurred because there is natural, large variability in the emission potential of PI-SWERL tests. Several tests resulted in the filters being heavily caked in material. This caking becomes a source of error because portions of the collected mass fall off the filter during transport and because analytical techniques such as x-ray fluorescence spectrometry (XRF) are less accurate when the particle deposit is thick. Another problem with the low-volume samplers used was that the size selective inlets on those samplers do not have sharp size cuts. During the later White Rock Fire study, a single filter was collected using a higher volume (16.7 lpm) sampling device with much better size cut characteristics rather than simultaneous two-filter collections. A drawback was that analyses from the two different types of filters did not represent the exact same source material because only one filter could be collected at a time.

Although there were issues with the filter sampling, they still provided useful insights. For example, at the Jacob Fire site (Figure 14), the ratio of non-carbonate, elemental carbon (EC) to soil crustal material (soil, operationally defined as the sum of the oxide minerals of silicon, aluminum, iron, calcium, and titanium) was found to be indicative of the evolution of the relative magnitude of PM_{10} emissions associated with fire products and those associated with the underling soil minerals. High EC/soil ratios immediately after a wildfire are a consequence of particulate matter from the charring of plants and other organic materials being available for windblown emission. Over time, the EC/soil ratio decreased, which showed a depletion of burn by-products in the suspended PM_{10}. Interestingly, it appears that some of the by-products from burned areas migrated to unburned areas over the course of the study, resulting in slightly higher EC/soil ratios on unburned test plots. For the burned areas, the EC/soil ratio had stabilized at a much lower level by 24 MAB than at 1 MAB. This is also coincident with the time that the magnitude of emissions from the Jacob Fire site locations started to approach pre-fire levels. Overall, this supports the idea that within two years after the fire, the Jacob Fire site had largely returned to pre-fire conditions in terms of the types and magnitudes of windblown PM_{10} emissions.

The chemistry data from the Gleason Fire site were much more variable (Figure 15), but also provided some helpful insights. Immediately after the prescribed burn, the EC/soil ratios were elevated at both burned interspace and undercanopy areas, which was a result of burned plant material providing a source of suspendible dust-sized particles. However, those EC/soil ratios were lower than pre-fire values within one year, which suggests that emissions of windblown PM_{10} as measured by the PI-SWERL were almost exclusively because of soil mineral particles at that time.

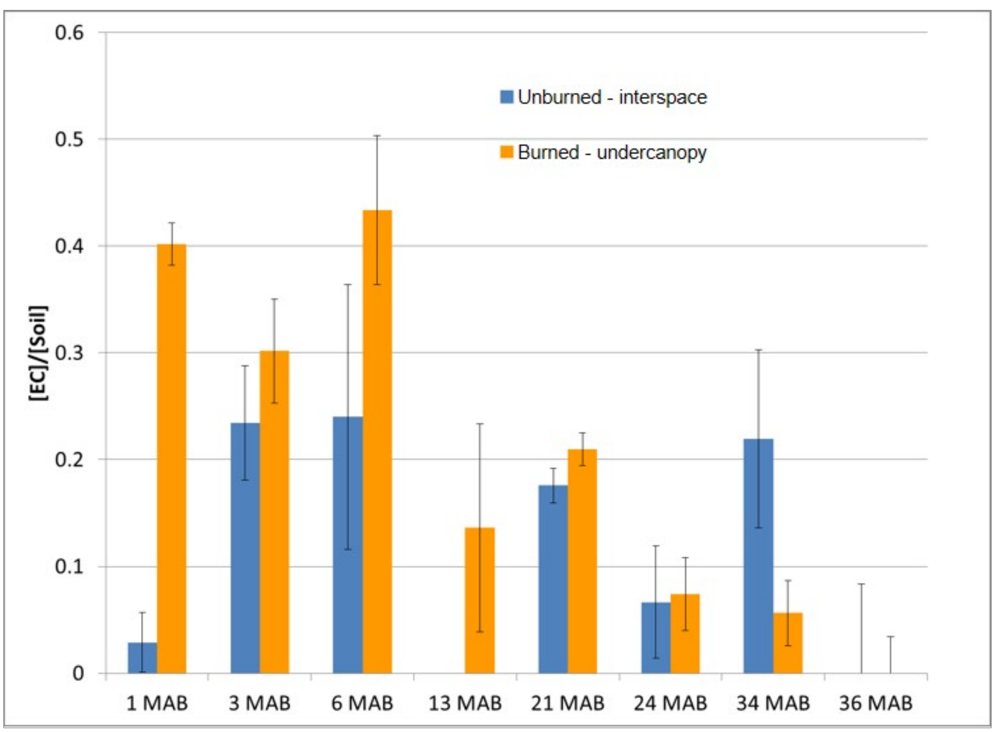

Figure 14. EC/soil ratios at Jacob Fire site.

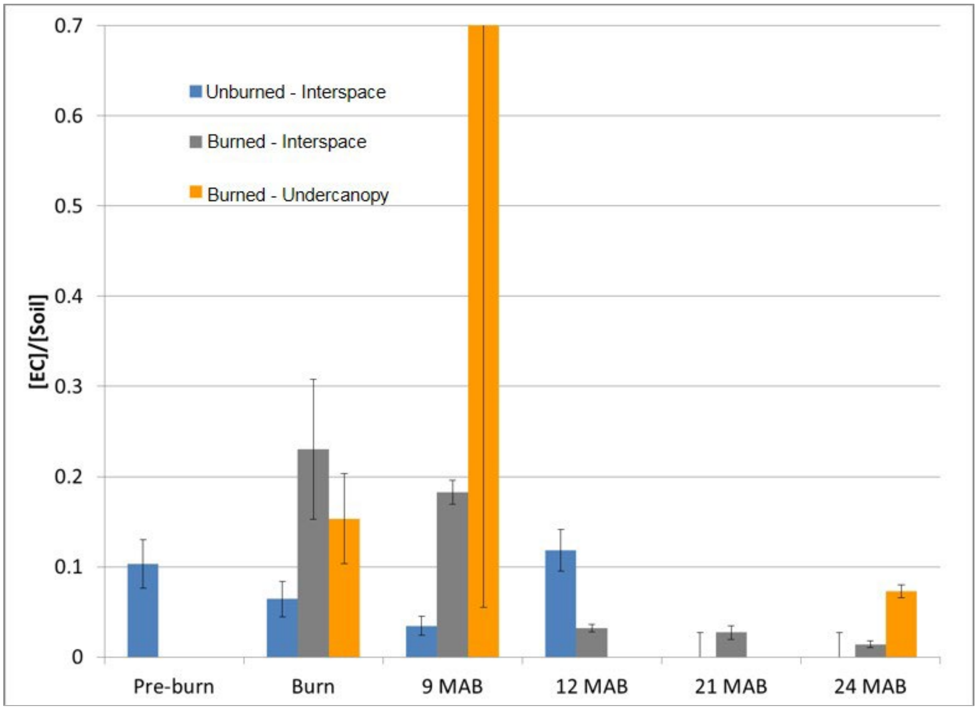

Figure 15. EC/soil ratios at the Gleason Fire site.

Chemistry data from the White Rock Fire study were not amenable for estimating EC/soil ratios because those analytes were from different types of filters and those filter types were not collected simultaneously from the same test surfaces. Therefore, it was not possible to compare EC at the White Rock Fire study.

The three fire study sites represent a gradient of ecoregions that span from the hottest, driest, most sparsely vegetated, and lowest elevation of the White Rock Fire site (Mojave Desert) to the coolest, wettest, most vegetated, and highest elevation of the Gleason Fire site (Great Basin Desert). The Jacob Fire site (transition zone) characteristics fall between those two end-members (Table 1).

The PI-SWERL measurements at these sites also indicated that there was a range of wind erodibilities in response to the occurrence of fire. Wind erodibility at the White Rock Fire site appeared to be least affected by fire, but measurements of the White Rock Fire site may have been affected by the two-year delay between when the fire occurred and when the measurements began. Although at burned plant microsites, emissions were elevated by approximately a factor of 10. At the Jacob Fire site, emissions after the fire from burned plant undercanopies were two or more orders of magnitude higher than emissions from unburned interspace areas. Two years following the fire, emissions in burned areas were within an order of magnitude of the unburned areas, indicating that emissions were attenuating over time. At the Gleason Fire site, emissions were significantly elevated after the fire at both burned interspace and burned plant undercanopy areas compared with the unburned interspace areas. Moreover, it was not clear that any attenuation of these elevated emissions had been achieved by the end of the study period, which was roughly three years after the fire event.

These results indicated that from the perspective of contaminated soils sites, the greatest post-fire wind-erosion risk posed by wildfires was likely to be associated with the landscapes and ecoregions higher in plant density and biomass, which are typified by the Gleason Fire site of the Great Basin (i.e., northern Yucca Flat and Pahute and Rainier Mesas). The landscapes and ecoregions that more closely resembled the White Rock Fire site of the Mojave (i.e., Frenchman Flat) represented a lower risk of windblown contaminated soils transport. However, there are a few caveats. First, the sites were on terrain with substantial relief, and therefore soils at these sites were well armored with gravel cover. Although these types of terrains are more likely to be the sites of fires in desert environments precisely because the terrains are vegetated and sloped, they are also inherently less dusty than flatter terrain, where soils typically have a higher fine sand, silt, and clay content (i.e., finer soil particles). It is noteworthy that the slope where wind erosion was measured on the burned portions of the Gleason Fire site was relatively gentle compared with the slope of the rest of the terrain of the study area, which was steeper. It is possible that the localized flat area was the site of finer-grained deposition and it had more erodible material compared with the steeper areas of the study site, which added to the comparatively higher and more prolonged susceptibility to wind erosion at the Gleason Fire site.

Second, wind erodibility was often found to be elevated in burned undercanopy areas compared with interspace areas. However, no measurements of wind erodibility on plant undercanopy areas in unburned portions of the landscape were completed because it was not possible to take measurements while plants were still present/alive. Therefore, the PI-SWERL data did not show whether the fire was the cause of increased susceptibility to wind erosion under the burned plants or the soil under the plants was composed of inherently wind-erodible material. There is no straightforward way to infer how different the emissions from these undercanopy areas would be if the plant was simply cut away instead of being burned away. An attempt was made to address this issue by collecting samples for chemical analyses, but the techniques used did not provide conclusive results.

Third, the PI-SWERL data reported here only address changes in the wind erodibility of the soil surface as a result of the fire event. Those changes, even in the cases of burned undercanopy areas, are relatively modest (up to a few orders of magnitude greater than unburned interspace). In the cases of these landscapes, a paramount consideration is what happens to the surface roughness of the terrain as vegetation is burned during a fire event. Through the process of shear stress partitioning, roughness elements such as shrubs and trees offer the soil surface protection from wind shear stress, effectively shutting down the windblown dust emission pathway altogether in many cases. Once these roughness elements are removed by fire, the unmitigated wind shear stress is free to act in its entirety on the surface. Because the amount of dust emitted is nonlinearly related to the wind strength (Figure 8), the combined effect of fire on enhanced wind erodibility of burned plant undercanopy areas over the surrounding interspace areas may be much more than the few orders of magnitude that are indicated by the PI-SWERL data alone. This effect will vary from location to location and has to be quantified separately using shear stress partitioning techniques, but it is generally most pronounced for the more heavily vegetated areas.

SYNTHESIS OF POST-FIRE VEGETATION RESPONSES

Gleason Fire (Great Basin Ecoregion)

Methods

There were two different studies performed to examine the effects of fire on Great Basin plant communities: 1) 12 to 24 MAB, burned plant communities were compared with unburned plant communities under shrubs (undercanopy) and in between shrub crowns in the interspaces; and 2) in 2012, the spatial characteristics of plant communities were examined in unburned sites. The two different techniques provided insights into plant recovery both temporally and spatially.

Twelve to 24 MAB, plants were surveyed using 50 m × 2.5 m (164 ft × 8.2 ft) belt transects. Each plant was measured for height and crown width. In 2012, 1 m^2 (10.7 ft^2) plots were conducted to examine the spatial patterns of the plants, which are important for understanding the movement of water and sediment in the system.

Results

Post-fire vegetation 34 MAB was dominated by annual forbs and grasses and perennial grasses (Figure 16). There was little recruitment of shrub species, especially typical Great Basin species such as big sagebrush (*Artemisia tridentata*), and there was no observed recruitment of tree species, such as Utah juniper (*Juniperus osteosperma*) or pinyon pine (*Pinus monophylla*). However, cheatgrass (*Bromus tectorum*)—an invasive annual grass—was present in post-fire burned plots. Overall, the post-fire vegetative cover was lower in the burned plots than in the unburned plots after three years, which was likely because of the low post-fire recruitment of shrubs. Other studies in sagebrush-dominated ecosystems suggest the recovery of canopy cover depends on the subspecies of sagebrush, but may take at least 25 years and possibly much longer (Lesica *et al*., 2007; Sankey *et al*., 2008; Cooper *et al*., 2011).

Figure 16. Image of the unburned area (left) adjacent to the Gleason Fire and the burned area of the Gleason Fire (right). The trees and shrubs visible in the images are sagebrush (*Artemisia tridentata*) and Utah juniper (*Juniperus osteosperma*).

Although cheatgrass was uncommon in the burned plots, it was also uncommon in the unburned plots. However, post-fire cheatgrass was not found in burned plots during the first year, but cheatgrass was found at similar levels in the burned and unburned plots after two years. *Gilia* spp. and Sandberg bluegrass (*Poa secunda*), which are both common species found in sagebrush shrublands after fires, were the only species that positively responded to the fire in both 2010 and 2011 (Ott *et al*., 2001). Other species showed varied responses or negative responses when densities in unburned and burned plots were compared, especially shrubs and bunch grasses. Additionally, post-fire grazing by cattle on the Gleason Fire site may have altered the natural succession. Heavy post-fire grazing can reduce perennial grass cover (Bates *et al*., 2009; Bates and Davies, 2014), which anecdotally occurred based on field observations but was not tested statistically. Although cattle do not occur on the NNSS, Pronghorn antelope and a small population of feral horses, typically between 30 and 50 (Wills, 2015), do occur on portions of the NNSS within the Great Basin vegetation community and can have effects similar to cattle on vegetation (Beever *et al*., 2008).

Unfortunately, the overstory or canopy cover of pinyon pine and juniper was not measured in either the burned or unburned areas, which would have provided information on the change of canopy cover. Canopy cover alters near-ground solar radiation and affects the ecohydrology of the ecosystem by providing a buffered microhabitat to low-growing or immature plants. Canopy cover is also an important control for fuel and fire dynamics. However, qualitative observations suggested that nearly 100 percent of overstory trees were killed by the fire and much of the shrub cover was removed.

Jacob Fire

<u>Methods</u>

Vegetation surveys were conducted between 2009 and 2011, 12 to 34 MAB. Plot sample size varied annually, but a total of 220 plots were established in unburned locations and 230 plots were located in burned areas. All plots were measured using a 50 cm × 50 cm (19.6 in × 19.6 in; 0.25 m^2, 2.7 ft^2) quadrat, but plots were not revisited annually and were established randomly in new locations each year. Within each plot, all plants were identified to the lowest taxonomic level possible and the number of each species was tallied in each plot. To examine how plants recovered on different geomorphic settings, or microhabitats, plots were established in either “drainages” (if located in a drainage) or “uplands” (if located outside of a drainage channel). Additionally, each plot in a specific geomorphologic setting was located either within the shaded extent of a shrub crown (undercanopy) or the intercanopy space between shrubs (interspace). Burned undercanopy and interspace sites were identified by the charred soil surface or visible shrub remains. 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 is related to the persistence of repeat fire susceptibility. Because of the prevalence of blackbrush (*Coleogyne ramosissima*) on the NNSS and its propensity to combust during fire, there was a focus on blackbrush and cheatgrass, the invasive non-native grass that has been shown to greatly increase fire risk (Brooks and Matchett, 2003).

Results

The fire removed most of the overstory vegetation (Figure 17) at the Jacob Fire site, and dramatic temporal changes in the plant community occurred in the first three years after the fire. The changes can be attributed to annual precipitation and post-fire succession. For example, post-fire plant density after one year was similar between burned and unburned plots, but plant density was higher on post-fire plots after two years. Although burned and unburned communities on the uplands had similar plant densities three years after the fire, burned plots in drainages had significantly higher plant densities than other plots. Plots located in drainages typically had more dense plant communities than those located in uplands because of increased soil moisture. Shrub thinning was also likely as the community matured because of competition among neighboring plants.

Cheatgrass was observed in 75 percent of the burned plots, which was similar to the unburned plots. However, densities were higher in unburned plots. By the end of the study, cheatgrass was also more dominant in the community composition in unburned plots. The dominance of cheatgrass on the unburned plots as well as its presence on the burned plots indicates that the potential for future fire may be similar between the burned and unburned areas. Blackbrush was not observed in the burned drainages, but it was observed in unburned drainages and it was four times more common in the burned uplands than in the unburned uplands. Three years after the fire, the burned plots continued to recruit blackbrush.

In 2011, plots located in the intercanopy showed increased plant density and species richness three years after the fire (Table 3). The density and richness of all the plots was significantly higher in the burned plots compared with the unburned plots. Additionally, intercanopy plots within the burned area had greater plant density than intercanopy plots in unburned areas. The undercanopy plots showed much lower plant densities in burned areas than in unburned areas. The burned intercanopy plots also had consistently higher species

Figure 17. Photograph of the unburned area (left) adjacent to the Jacob Fire site and the burned area of the Jacob Fire site (right). The conspicuous plants observed in the photographs are Nevada ephedra and blackbrush.

richness. Plant densities were consistently greater in the burned plots than in the unburned plots, but were likely a result of many young plants regenerating after the fire. As the plants mature, self-thinning will likely occur and the density will become similar to the unburned areas.

Species richness was similar between burned and unburned plots. However, the difference in cheatgrass dominance between the burned (relatively low) and unburned (relatively high) plots was large. Cheatgrass was 18 times more dense in the unburned plots than in the burned plots overall, and nearly 30 times more dense in unburned upland plots than burned drainage plots. Typically, cheatgrass increases after fire (Keeley, 2006), but in some situations, fire has been shown to temporarily reduce the seedbank, which may explain the differences in cheatgrass abundance (Alexander and D'Antonio, 2003). The risk of fire recurrence because of cheatgrass regeneration in previously burned sites several years after the fire is likely relatively low compared with fire risk in previously unburned areas because of the differences in cheatgrass density. However, cheatgrass provides little soil stability and the reduced post-fire presence of cheatgrass limits any potential stability. Additionally, the overall density of cheatgrass in unburned upland plots was nearly three times greater than in burned upland plots, so soil stabilizing benefits from the number of plants two years after the fire would likely be limited until perennial vegetation recovers to a greater extent or additional annual species colonize the burned uplands.

White Rock Fire

Methods

The White Rock Fire burned in 2011, but parts of the fire re-burned the 1993 Bunker Fire and 2006 Cabin Fire sites. However, no plots were established on the Cabin Fire site. Plots were established to test two responses of the plant community: the differences between NB plant communities (White Rock Fire site), OB plant communities (Bunker Fire site), and the unburned areas; and how plant communities responded across soil ages to fire.

Permanent 1 m^2 (10.7 ft^2) plots were established in June 2014 and were stratified randomly by fire type (OB, NB, and unburned). A total of 60 plots were established, with 20 plots in each of the OB, NB, and unburned areas spread across varying geomorphic surfaces. Data were collected between June 2014 and March 2016 using the same 1 m^2 (10.7 ft^2) plots each year. Measurements at each plot included fire age, plant species, live/dead plant status, primary and secondary axis measurements of plant crown, and plant height. A subplot, 10 × 10 cm (3.9 × 3.9 in), was randomly dropped once from each of the three sides of the larger 1 m^2 (10.7 ft^2) primary plot. The number of plants within each of the three subplots were counted and were identified by species when possible (i.e., when sufficient live or dead plant material was present to identify the species accurately) to estimate the density of annual and herbaceous plant material within each plot.

Results

Patterns of the plant community suggest all three treatments were in different states of succession or community maturity, and some recovery had occurred on the OB sites. The NB sites had a substantially higher amount of native early successional species, such as desert globemallow (*Sphaeralcea ambigua*) and desert marigold (*Baileya multiradiata*). However, the occurrence of invasive species, such as red brome and stork's bill (*Erodium cicutarium)*, were higher on recently burned sites than all other treatments. The OB sites had a mix of early successional species, such as desert marigold and desert globemallow, but also late successional species such as white bursage (*Ambrosia dumosa*) and creosote bush (*Larrea tridentata*). Interestingly, *Schismus* spp. was significantly more abundant on the OB sites than in either of the other treatments, but red brome had similar abundance as on the unburned plots. These results suggested that red brome was an early post-fire colonizer, but *Schismus* spp. may outcompete over time and have higher abundances.

General Findings from the Three Ecoregions

Predicting wildfire potential among three different ecoregions is difficult without quantifying fuel loads, but the potential for a fire to spread is likely much higher in all ecoregions compared with pre-Euro-American settlement because of the increase in introduced annual grasses. Introduced annual grasses increase the fuel connectivity and provide additional fine fuels, especially the early season annuals such as cheatgrass and red brome. The early season annuals use winter precipitation to grow and set seed prior to most native plants, but then they dry up during warm, dry time periods, which provides substantial fuel and increased fuel connectivity. The prevalence of annual grasses at each burned area increases the probability of fire compared with locations without annual grasses (Table 3). The annual grasses dry up early in the season, which increases fuel loads and fuel connectivity and allows fire to carry more easily between shrubs. Although some locations, such as the Jacob Fire site (transition zone), generally had reduced levels of cheatgrass after the fire compared with the unburned areas, each year after the fire showed increased cheatgrass densities, which suggests densities may become similar over time.

Table 3. Common introduced species found on each fire site and general post-fire responses.

Fire Name	Introduced Species	Post-fire Potential for Another Fire
Gleason	Cheatgrass	Increase
Jacob	Cheatgrass	Lower first year post-fire, increase after the third year
White Rock	Red brome, *Schismus* spp.	Increase in all post-fire landscapes

<u>Recovery from Fire among the Three Ecoregions</u>

Understanding post-fire vegetation recovery is increasingly important for land management agencies because fire frequency in the western United States has increased over the past several decades (Brooks, 1999; Brooks and Esque, 2002; Brooks and Matchett, 2006). Since Euro-American settlement, fire potential has increased in all three ecoregions because of: 1) introduced annual grasses, which were found in burned and unburned plant communities among all ecoregions; 2) increasing regional and global temperatures have increased fire season length (Westerling *et al.*, 2006), and persistent regional drought conditions since the year 2000 have led to larger, more intense wildfires; 3) human-ignited fires account for >80 percent of the fires since 1992, which has increased the length of the fire season and the amount of ignitions throughout the region (Balch *et al.*, 2017).

Additionally, in the shrub dominated ecosystems of the West, it has been difficult to determine fire return intervals or accurately date fires, which are vital to properly understanding post-fire succession. The Gleason Fire site and similar locations at the pinyon-juniper woodland/sagebrush shrubland ecotone can provide evidence through fire scars, but other shrub dominated systems similar to those at the Jacob Fire and the White Rock Fire sites cannot provide such evidence (Table 4). Although many state and federal land management agencies have begun to document fire locations and perimeters for the past 30 years, developing a holistic model of post-fire recovery has been limited by the short temporal span (Brooks and Matchett, 2006), which is often significantly shorter than the average lifespan of the shrubs. Fire chronosequence studies, such as those conducted at the White Rock Fire site (i.e., Brooks and Matchett [2003], Abella [2010], and Engel and Abella [2011]), provide insights on ecological succession and fire recovery in semiarid regions. However, paleoecological data suggest disturbed plant communities can take millennia before late-successional species colonize after disturbance (Bowers *et al*., 1997; Cole, 2010).

The data from each study showed the burned plots did not return to a pre-fire plant community and suggest that recovery may require a longer time than this study, perhaps multiple decades to centuries or millennia. Introductions of invasive plant species, especially the annual grasses, suggest that desert shrublands may not return to a pre-burned plant community and future climate projections suggest that regional drying may occur, and therefore it is possible that the burned areas may not return to a pre-fire plant community as the region's climatic niche changes (Bell *et al*. 2014). The increase in density of invasive species after the fire and the slow recolonization of some species suggest that recovery times may be extensive. However, the unburned sites also contained invasive species. Although the invasive species were at lower densities at the White Rock Fire site, higher densities were found in unburned plots at the Jacob Fire site.

Table 4. Dominant vegetation found at each fire site and the general post-fire response.

Fire Name	**Unburned Vegetation**	**Burned Vegetation**	**Recruitment of Late-successional Species**	**Post-fire Introduced Plant Species Present**
Gleason	Pinyon-juniper/sagebrush	Annual grasses/forbs	No	Yes.
Jacob	Blackbrush, Nevada ephedra	Annual grasses/ forbs, blackbrush	Yes	Initially reduced, but similar levels after three years.
White Rock	Creosote-bursage/Joshua tree	Annual grasses/perennial grasses, early succession shrubs	No	Yes. Some species increased on recent burn, whereas others increased on older burn.

Unknowns of Plant Recovery in the Great Basin, Transition Zone, and Mojave Desert

There are still many data gaps regarding fire, fire dynamics, and plant recovery in semiarid shrublands. Data gaps exist for numerous reasons, but the long time periods of post-fire recovery greatly add to the lack of data. To appropriately understand the post-fire dynamics in each ecosystem, long-term data at varying temporal and spatial scales are needed. Unfortunately, there is a generally poor understanding of long-term fire in shrubland ecosystems, such as those assessed in this study, because of the paucity of long-term fire records. Shrublands do not record fires well because plants are consumed by fire and few other paleoecological studies occur in semiarid shrublands because of the lack of proxy records capable of recording fire events in space and time. Therefore, estimates of pre-Euro-American fire return intervals in shrublands vary dramatically. For example, the time spans of fire return intervals in creosote-bursage shrublands—a dominant vegetation type of the Mojave Desert and a common component of the southern portion of the NNSS (e.g., Frenchman Flat), as well as the vegetation type of the White Rock Fire site—range between 100 and 1,000 years. The sagebrush shrublands of the Great Basin Desert (northern portion of Yucca Flat and mesas) have return intervals of 10 to 100 years, and the return intervals of the blackbrush shrublands of the transition zone (southern and middle portions of Yucca Flat) range between 35 and 100 years.

Comprehensive Understanding of Ecological Succession in Shrublands

There is a lack of a comprehensive understanding of ecological succession in the semiarid shrublands of the Great Basin and Mojave Desert. Although it is generally predictable that early successional species, or pioneer species, colonize more rapidly than slow growing, long-lived shrubs, it is not well understood how plant communities change with time after a fire. Additionally, the presence of the invasive annual grasses inhibits native species regeneration after fire and it is unknown how shrublands will respond, or recover, over long time periods in the presence of invasive annual grasses.

Differential Burn Severity

The burn severity of a fire is frequently unknown, is spatially variable, and provides uncertainty into the recovery process or may introduce biases. The goal of the Monitoring Trends in Burn Severity (MTBS; https://www.mtbs.gov) program is to document fire boundaries and severity using satellite imagery, but data are not always available. For example, MTBS has data for the White Rock Fire and the Bunker Fire, which burned prior to the White Rock Fire, but it does not have data for the Gleason Fire or the Jacob Fire. Additionally, prescribed fires rarely have calculated burn severity data, although they may be estimated by field personnel. Burn severity may greatly affect the soil and vegetation recovery, which could potentially alter post-fire sediment transport.

SUMMARY

This report summarizes data from the Great Basin and Mojave Desert ecoregions and the transition zone between them. Three years were spent collecting post-fire data at each ecoregion on soil erosion potential by wind and water, and documenting vegetation response following a fire at a site comparable to areas of the Nevada National Security Site (NNSS) and the Tonopah Test Range (TTR). The White Rock Fire site was first measured at 22 months after burn (MAB), which makes directly comparing post-fire responses between 0 and 22 MAB to the other fire sites difficult, but the later stages were comparable between the three sites.

Fire appeared to have the greatest effect on the soils of the Gleason Fire sites. For example, water repellency was evident on post-burn soils up to 12 MAB, but soil began to return to pre-fire soil properties by the end of the project, which was three years post-fire. Overall, runoff on burned interspace soils was greater than on unburned soils throughout the duration of the project, but burned undercanopy soils were likely not source areas of erosion, and consequently runoff. Runoff data from the Jacob Fire and the White Rock Fire sites showed more similarities between burned and unburned soils, especially by 48 MAB—although the White Rock Fire site was not measured during the first 22 MAB, which suggests that any increases or changes in runoff may not last more than several years after a fire. Similar patterns occurred in the wind erosion study as well. Wind erosion from soils exposed to fire was elevated compared with unburned soils and did not return to a pre-fire level during the duration of the project at the Gleason Fire site. However, wind erosion appeared to return to levels similar to the unburned areas at the Jacob Fire and the White Rock Fire sites. One reason for the persistently high wind erosion at the Gleason Fire site may be the site selection, which was located in an area of deeper soils with possibly different soil properties, whereas sites at the other fire locations were on ridges with shallow soils.

Post-fire vegetation and recovery of vegetation to pre-burn levels did not occur within the three-year study time frame at any site, and even the old fire (Bunker Fire) at the White Rock Fire site had a different vegetation composition than the unburned plots. These data suggest that the shrublands of the Great Basin and Mojave Desert and their transition zones may require decades to return to vegetation states similar to the unburned areas. However, regeneration of native perennial vegetation occurred at each fire site, with the Great Basin site dominated by forbs and grasses, whereas the transition zone and Mojave Desert sites were dominated by shrubs and forbs. Additionally, the burned areas in each of the ecoregions contained substantial amounts (either in density or percentage) of introduced annual grasses. Although the burned areas of the Great Basin and the transition zone sites generally contained reduced annual grasses, the densities increased every year after the fire, which further suggests that the post-fire density may reach or exceed the levels of unburned vegetation over time. The presence of the invasive annual grasses indicate that the probability of future fires may increase, but fuel loads and connectivity will greatly depend on winter precipitation and summer drought to provide the appropriate productivity of annual grasses and drying of fuel loads.

Overall, the data suggest that the hydraulic properties and wind erodibility of the soils returned to pre-fire conditions more quickly after fire than the vegetation, but site differences (e.g., soil depth, soil properties, and plant community composition) strongly influenced runoff and wind erosion potential from burned areas. Based on the measurements made at the Gleason Fire site, it is unclear whether wind or water has the greater erosion potential because more of the landscape surface (i.e., burned undercanopy soils) was exposed to wind or because precipitation will more frequently result in runoff. Although vegetation at all sites did not show a return to pre-burn community structure, post-fire recruitment was generally dominated by native species, even though invasive annual grasses were present at each site.

Future fire potential will greatly depend on the persistence of the invasive annual grasses as well as the precipitation regime because both variables drive the productivity of invasive annual grasses. Soils and vegetation at the NNSS should be expected to behave in a similar manner to those studies here, showing increased risk of soil erosion after fire, particularly in the Great Basin ecoregion, especially from wind. Most erodibility measurements suggest that by three years post-fire, erosion potential has nearly returned to pre-burn conditions.

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