Fire Effects on Wind Erosion in Sagebrush Steppe Rangelands of the Snake River Plain

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ABSTRACT

Wind erosion of soil is an appreciable but unstudied event following fires in cold desert. We examined aeolian transport of sediment for one year following fire in semiarid shrub steppe of southern Idaho. Sediment collectors were used to determine horizontal mass transport of soil ($\int_{height=0}^{2m} mass * area^{-1} * time^{-1}$) and saltation sensors and anemometers were used to determine saltation activity (# seconds saltation is detected per 300 seconds) and threshold wind speed in an unburned control site and an area burned in August. Horizontal mass transport (per 30 d periods) was negligible in the unburned area, but in the burned area was 5.40 kg m^{-1} in October and decreased to 2.80 kg m^{-1} in November and 0.32 kg m^{-1} in December. Mean threshold wind speeds were between 10.0 and 10.6 m s^{-1} at the burned site during fall. A lack of saltation precluded determination of thresholds thereafter at the burned site and for the entire course of the study at the unburned site. Differences in sediment flux and saltation activity at burned and unburned sites became much less pronounced following the emergence of herbaceous vegetation in the spring. Our results indicate that the burned shrub steppe was prone to aeolian sediment transport in the first months following summer wildfire and became less so by early winter, through spring and summer.

KEYWORDS: Aeolian sediment transport, wind erosion, wildfire, sediment flux, saltation

INTRODUCTION

Aeolian sediment transport is a fundamental geomorphic process that has wide-ranging environmental implications for human and environmental health (Whicker et al., 2006), ecological functioning at multiple spatial and temporal scales (Okin et al., 2006), local and global biogeochemical cycling (Okin at al., 2004; Reynolds et al., 2001; Chadwick et al., 1999), and contaminant transport (Whicker et al., 2004). Aeolian sediment transport is a function of the wind's ability (impeded by vegetation and terrain) to entrain soil particles, and the soil's susceptibility to this entrainment (Okin et al., 2006; Bagnold, 1941). Field-based research on aeolian transport in non-agricultural systems has largely focused on cropland and arid landscapes, however semiarid landscapes, and shrublands in particular, exhibit considerable annual fluxes of wind-transported sediment (Breshears et al., 2003). Wind erosion can be especially significant following fire, as shown for prairie and warm desert in the southern United States. Transport via wind can exceed that by water in semiarid shrublands (Vermeire at al., 2005; Whicker et al., 2002; Zobeck et al., 1989).

Increased wind erosion has been reported immediately following fall or spring prescribed fires in areas also affected by grazing (Vermeire et al., 2005), and at various intervals following summer wildfire (Whicker et al., 2002; Zobeck et al., 1989). Whether wind erosion persists through plant re-colonization of burned areas, particularly in more northerly regions in which plant reestablishment is delay during cold winters, is not well known. Moreover, the previous studies of fire effects on wind erosion have all occurred on sandy soils, which were probably impacted by a previous history of soil erosion. We examined the course of wind erosion as it related to changes in plant abundance and hydroclimatology following a fire in cold desert sagebrush-steppe occurring on fine-textured loess soils, which appeared to otherwise have not been affected by recent fire and aeolian losses of soil. This vast habitat type (10 M ha) is uniquely characterized by warm, dry summers, and cold, snowy winters, and spring snowmelt provides the bulk of the annual plant available water at the time of green-up (Smith et al. 1997). Fire frequencies and area burned per year have increased substantially in the recent decade in sagebrush steppe, becoming nearly 500 ha in some years (eg. 2007 Murphy complex fire in Western North America). Fire typically occurs in summer, and charred soils frequently remain un-vegetated until re-sprouting of grasses, as well as some forbs and occasionally shrubs, occurs in the subsequent spring and summer (Harniss 1970). Herb cover, especially grasses, dominate vegetation cover for decades following fire, while species such as sagebrush slowly recover to pre-fire levels. Seeding and planting are commonly used in attempt to bolster cover in the first years after fire (US DOI, BLM), which implies the assumption that natural plant recovery is not adequate to stabilize soil.

Simultaneous observations of saltation activity, the fraction of time in which saltating particles can be detected, and critical aeolian threshold (hereafter "threshold", the minimum windspeed required for saltation) were useful for distinguishing erodibility of geomorphic surfaces with expectedly different susceptibility to erosion by wind (Stout, 2007). We expected that simultaneous observations of saltation activity and critical aeolian threshold in conjunction with sediment flux measurements would similarly be useful for describing differences between the wind erosion potential of burned and unburned rangelands. Saltation activity, threshold, and actual rates of horizontal sediment transport can be determined from simultaneous micrometeorological measurements of wind and other boundary layer atmospheric characteristics, passive sediment sampling systems for measurement of horizontal and vertical sediment mass flux, and direct measurements of soil loss and deposition at the soil surface (Vermeire et al., 2005; Whicker et al., 2002; Zobeck et al., 1989). We used these approaches to determine the following for a burned and unburned control area: 1) differences in wind erosion potential, and 2) the longevity of these differences, in burned vs. unburned semiarid cold desert shrub steppe. We hypothesized that: (i) wind erosion potential would be significantly greater at burned compared to unburned sites during fall, and (ii) significant differences in wind erosion potential between burned and unburned sites would not be detectable following the spring emergence of herbaceous vegetation.

MATERIALS AND METHODS

Study Sites

The study sites are located in the southern portion of the Eastern Snake River Plain, near Aberdeen, Idaho. The area is located in a zone of 200 mm to 280 mm of mean annual precipitation and 7 to 13 degrees C mean annual temperature (NCSS Web Soil Survey). This portion of the Snake River Plain is characterized by near surface winds that trend generally throughout the year from SW to NE (Clawson et al., 1989). On average, the frequency of high wind events is greatest in spring and summer, with fall being calmer, and winter the calmest season (Clawson et al., 1989).

The rangeland vegetation includes Wyoming big sagebrush (*Artemisia tridentata ssp. wyomingensis* Rydb.) and the less abundant shrubs "three-tip" sagebrush (*Artemisia tripartita* Rydb.) and green rabbitbrush (*Chrysothamnus viscidoflorus* [Hook.] Nutt.), bluebunch wheatgrass (*Agropyron spicatum* Pursh.), prairie junegrass (*Koeleria macrantha* Ledeb.), and sandberg bluegrass (*Poa secunda* J. Presl) (NCSS Web Soil Survey). Soils at the study sites have developed from silty loess overlying basalt bedrock (NCSS Web Soil Survey). Depth to bedrock ranges from 20 to 150 cm, and surficial soil textures are silt loam in deeper loess-derived soils and a stony loam in some locations where either fractured bedrock is nearer the surface or small, intermittently active alluvial channels cross the sites (NCSS Web Soil Survey). Soils at the sites are rated with a Wind Erodibility Index value of 1.9×10^5 kg ha^{-1} $year^{-1}$ soil loss, and have a Wind Erodibility Group rating of 4, which is intermediate to values for all USA soils (NRCS Web Soil Survey).

Site Characterization

Sampling was initiated in September, 2006 at a large fire on the Eastern Snake River Plain (Crystal Fire, approximately 80,000 ha, August, 2006). Two study sites in the burn and one study location in the neighboring unburned area were selected, at the easternmost extent of the fire. Sampling intensity was greater at one of the burn sites (hereafter referred to as the primary burn site), compared to the other burn site. Each site was delineated as a 100-m radius circle, and there was at least 300 m between the study sites and the edge of the burn area. Four transects radiating out from the center of the circle in the cardinal directions were used for the characterization of vegetation composition at each site. Percent cover of shrubs, herbaceous plants, and bare ground was measured along each transect using the line intercept method, in which cover beneath all lengths of transects was recorded and reported as a fraction of total cover. Height of individual shrubs along each transect were measured.

Micrometeorological data collection

At the outset of the study the average shrub, herbaceous vegetation, and bare ground cover, as well as shrub height means and standard deviations, were calculated from the eight transects for the two burned sites and from the four unburned site transects. Micrometeorological and saltation monitoring equipment was installed at the center of the unburned site and the primary burned site in the last week of September, 2006. Sensors included two anemometers (Model 014A, MetOne Instruments Inc., Grants Pass, OR, USA) mounted at the top of and below the unburned vegetation canopy (110 and 40 cm), a SENSIT® piezoelectric sensor (Sensit Company, Portland, ND, USA) mounted at 5 cm above the ground surface that records impacts from saltating soil particles, and temperature and relative humidity sensors (Model HMP50L, Vaisala Group, Vantaa, Finland) mounted at 5 cm above ground. The mean, standard deviation, and maximum wind velocity, the number of seconds with particle impacts, and average relative humidity and temperature were recorded at 5-minute intervals, at a 1-second scan rate using dataloggers (model CR10, Campbell Scientific Inc., Logan, UT, USA). Data collection was suspended in mid-January due to winter access problems. Data were simultaneously collected at the burned and unburned sites during approximately 105 days during this period of the study (fall/winter).

The micrometeorological and saltation monitoring equipment was reinstalled following the spring emergence of herbaceous vegetation on May 16, 2007 and data were collected until August 11, 2007. Vegetation cover was resampled on the unburned and primary burn site at the end of this 2007 period.

Data were collected simultaneously at the two sites for 85 days during this period of the study (spring/summer).

Aeolian sediment collectors

Big Springs Number Eight (BSNE®) (Custom Products, Big Spring, TX, USA) omnidirectional passive sediment collectors mounted at 5, 10, 20, 55, and 100 cm height were installed at the center of all three sites. Two additional sediment collector towers with collectors at 5, 10, and 20 cm height were installed at each site approximately 20 meters laterally from the site center, with one tower on either side of the axis of prevailing wind direction.

The aeolian sediment collectors were installed on September 28th at all three sites. Collections were made on October 30, November 30, and January 18, during fall/winter. The collectors remained in the field at all three sites after January 18th and were emptied once on May 14, 2007. Sediment collectors were then emptied on June 10th, July 21st, and August 17th, during spring/summer.

When the collectors were emptied, the sediment was placed in airtight bags, dried for 16 hours at 110 degrees C and weighed to 0.000 g precision. Sediment mass was averaged at each height, over each sampling interval, for the burned and the unburned sites, respectively. This resulted in potential sample sizes of n = 6 and n = 3 at heights of 5, 10, and 20 cm for the combined burned sites and the unburned site, respectively, and n = 2 and n = 1 at 55 and 100 cm for the combined burned sites and the unburned site, respectively. These are potential sample sizes because there were 9 instances on January 18th, and 6 on May 14th when a collector at a specific height could not be reliably sampled due to water or ice in the collector.

Data Analysis

The root mean squared difference (RMSD) of mean wind speeds between the burned and unburned sites was determined for each anemometer height using the mean wind speeds of simultaneous 5-minute intervals. This provided an average difference between the burned and unburned sites in mean wind speed at heights corresponding to the top and beneath the plant canopy height of the unburned site, for both fall/winter and spring/summer.

The proportion of each five-minute period during which impacts were recorded by the saltation sensor, termed saltation activity, was calculated by dividing the number of seconds with particle impacts by 300 seconds. Following Stout (2004), saltation activity is related to the minimum wind speed required to entrain saltating soil particles, termed the critical aeolian threshold, by the following equation:

$$\gamma = \Phi \frac{\overline{\mu} - \mu_t}{\sigma}$$

where γ is saltation activity, μ_t is the critical aeolian threshold wind speed of the higher anemometer in units of m s^{-1}, $\overline{\mu}$ is the mean wind speed of the higher anemometer in units of m s^{-1}, σ is the standard deviation of wind speed for the higher anemometer in units of m s^{-1}, and Φ is the normal distribution function. The critical aeolian threshold in wind speed units of m s^{-1} was determined by solving the previous equation for critical aeolian threshold (μ_t) (Stout, 2004).

Rain drops and splash can be falsely recorded as saltating sediment by the Sensit, so saltation activity and critical aeolian threshold were not reported on days in which rain was measured at the nearby USDA Aberdeen weather station. Critical aeolian threshold was determined for saltation activity values between 0.02 and 0.98, (Stout, 2004).

Aeolian sediment horizontal mass flux was calculated for each sediment sampling interval and height as:

$$mass * area^{-1} * time^{-1}$$

where mass is kg of sediment, area is size of the opening on the BSNE© sampler (0.0002 m^2 at heights 5 and 10cm, and 0.001 m^2 at 20, 55, and 100 cm), and time is the sampling interval in days. Sediment flux was normalized to a 30-day rate for each sampling interval (i.e. kg m^{-2} $30d^{-1}$), and plotted as a function of sample height. A power model was fit to this plot and integrated over 200 cm to calculate a sediment mass transport value of the form (VanDonk et al., 2003):

$$\int_{height=0}^{2m} mass * area^{-1} * time^{-1}$$

with units of kg m^{-1}. This resulted in 30-day sediment mass flux and sediment mass transport estimates over each sampling interval for the combined burned sites, and for the unburned site.

RESULTS

Fall/Winter

The unburned site had greater shrub and herbaceous vegetation cover than the burned sites, during fall/winter following wildfire (Table 1). The burned sites had more exposed bare soil. Shrub mean height and standard deviation were 2 to 3 times greater for the unburned site relative to the burned site. Mean wind speed at the top of (110 cm) and below (40 cm) the unburned vegetation canopy was noticeably smaller at the unburned compared to burned site for almost all periods of time during fall/winter (Figure 1). The mean difference between the burned and unburned sites for measured wind speeds at these heights was 1.79 and 1.55 m s^{-1}, respectively.

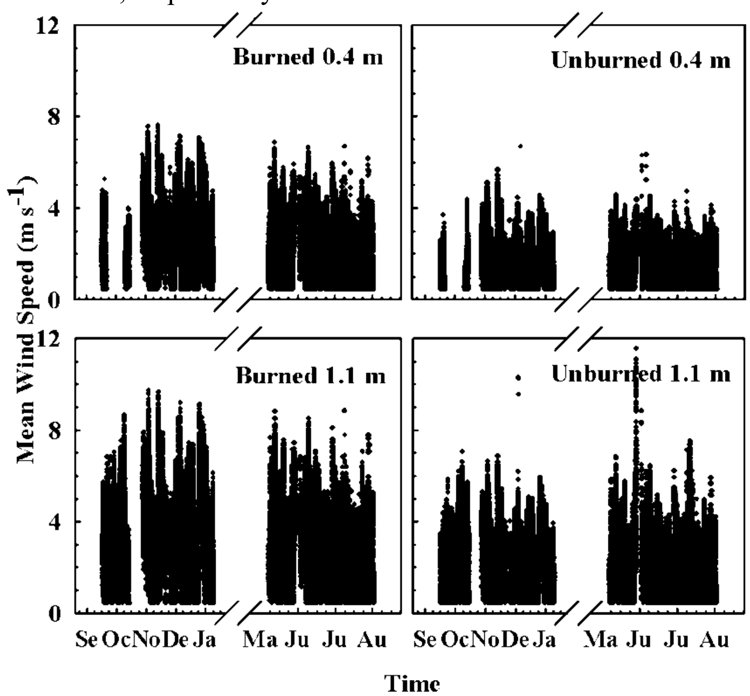

Figure 1. Mean wind speed measured below (40 cm above ground) the unburned vegetation canopy (upper panel), and at the top (110 cm above ground) of the unburned vegetation canopy (lower panel) at unburned and burned sites.

Saltation activity was more frequently detected, and greater in magnitude when detected, at the burned compared to unburned site over the course of fall/winter (Figure 2). Saltation was detected during less than 2% of the time, indicating an episodic nature of erosion events (Table 2). The fraction of time during which saltation was detected at the burned site was an order of magnitude greater relative to the unburned site.

Substantial differences were detected for saltation activity during the few periods in which both sites simultaneously achieved saltation activities greater than 0.02. There were only 39, 5-minute periods during which saltation activity was simultaneously determined for the burned and unburned sites in fall/winter. The mean saltation activity was 0.35 ± 0.04 SE in the burned sites, indicating 35% of 5-minute periods had saltation, compared to 0.05 ± 0.00 in the unburned sites.

Table 1. Mean (standard error) ground cover of sampling sites

		Ground Cover		
Period	Site	shrub %	herbaceous %	bare %
Fall 2006	Unburned (4 transects)	35 (2)	40 (3)	25 (4)
	Burned (8 transects)	6 (1)	3 (1)	91 (9)
Spring 2007	Unburned (4 transects)	40 (2)	54 (2)	6 (1)
	Burned (4 transects)	6 (1)	45 (9)	59 (6)

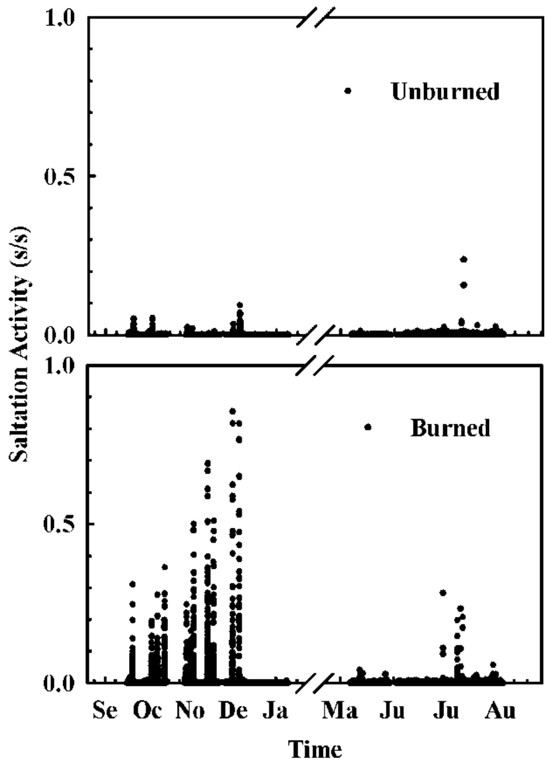

Figure 2. Saltation activity detected at unburned (upper panel) and burned (lower panel) sites. Saltation activity is unitless and is calculated as the number of seconds in a 300 second period during which particle counts were recorded.

Table 2. Fall, 2006 and Spring, 2007 saltation particle count results

Period	Site	Saltation Detected (minutes)	Fraction of Study Period (%)
Fall 2006	Unburned	220	0.1
	Burned	2825	1.9
Spr 2007	Unburned	35	0.0
	Burned	155	0.1

Reliable estimates of threshold wind speed could not be made for the unburned site because of the absence of sustained saltation events at this site (Figure 4). There were five days with sustained saltation events when threshold wind speeds could be determined at the burned site. Mean threshold wind speeds for these days were between of 9.15 and 10.62 m s^{-1} [October 19 (mean = 10.02 m s^{-1}, s.d. = 0.30 m s^{-1}, n

= 23), October 20 (mean = 10.10 m s^{-1}, s.d. = 0.32 m s^{-1}, n = 17), November 8 (mean = 9.15 m s^{-1}, s.d. = 0.71 m s^{-1}, n = 21), November 13 (mean = 10.62 m s^{-1}, s.d. = 0.63 m s^{-1}, n = 40), November 23 (mean = 10.12 m s^{-1}, s.d. = 0.54 m s^{-1}, n = 94).

Burned site horizontal sediment flux appeared comparable at most heights for the first two sampling intervals of fall/winter, and was substantially smaller in December and the first half of January (Figure 3). There was never a detectable quantity of sediment at any heights at the unburned site during fall/winter. The integrated power functions fit to burned site sediment flux measurements yielded sediment mass transport values of 5.40, 2.80, and 0.32 kg m^{-1} for the three fall/winter sampling intervals from the end of September through mid-January.

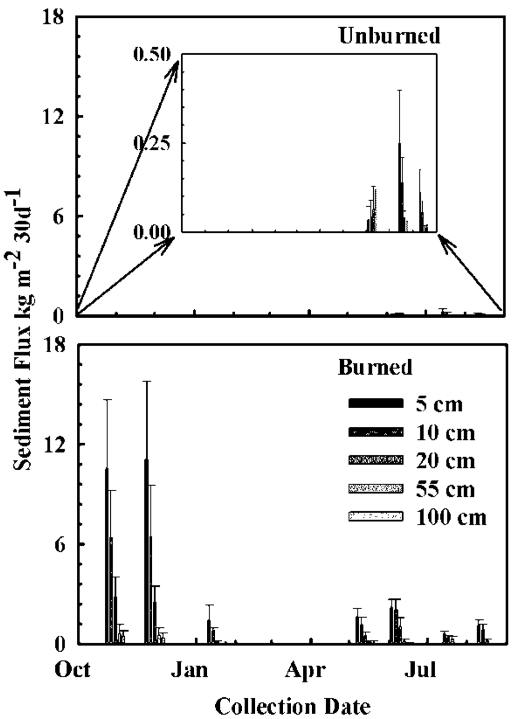

Figure 3. Mean horizontal sediment mass flux at unburned (upper panel) and burned (lower panel) sites. The unburned and burned sites were sampled at the same intervals and missing values for the unburned site represent periods when no measurable amount of sediment accumulated in the collector.

Spring/Summer

After the emergence of herbaceous vegetation in the spring, large differences persisted between the amount of exposed bare soil and shrub cover at the unburned site and primary burned site, though the sites were more comparable in terms of herbaceous vegetation cover (Table 1). The mean difference between wind speeds at the burned and unburned sites was not as great for either anemometer height during spring/summer (RMSD = 1.15 m s^{-1} [110 cm] and 0.92 m s^{-1} [40 cm]) compared to fall/winter. Saltation activity was never greater than 0.30 during spring/summer at either the burned or unburned sites (Figure 2). Saltating particles were detected during less than 0.1% of the study period during spring/summer at both sites (Table 2). There were only 2, 5-minute periods during which saltation activity was simultaneously determined at the burned and unburned sites in spring/summer. Reliable estimates of threshold wind speeds could not be determined for spring/summer at either site, due to the scarcity and very short duration of saltation events.

Detectable quantities of sediment did accumulate in the unburned site collectors during spring/summer, in contrast to fall/winter. Sediment flux, however, was lower at all collector heights at the unburned site compared to the burned sites during this time period (Figure 4). Mean flux values did not fit power functions with high correlation for the June 10 and August 17 collection dates at the unburned site and were therefore not integrated to estimate sediment mass transport values. Unburned site mean flux values for the July 21 collection date fit a power function with a moderate correlation ($R^2 = 0.7$) and resulted in a mass transport value of 0.08 kg m^{-1}. Burned site mean flux values did fit power functions with high correlation ($R^2 \geq 0.9$) and resulted in mass transport values of 0.46, 0.83, 0.21, and 0.47 kg m^{-1} for the four spring/summer sampling dates (May – August).

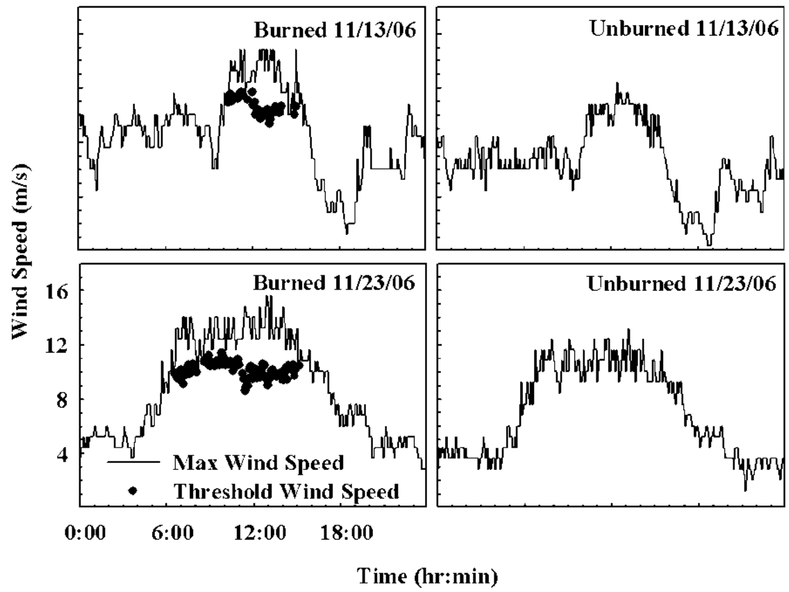

Figure 4. Examples of maximum wind speed and aeolian critical threshold determined for unburned and burned sites on November 11th and 23rd, 2006. Maximum wind speed and threshold wind speed data are determined for 5 minute periods.

DISCUSSION AND CONCLUSIONS

The combination of greater vegetation cover, shrub height, and variability in shrub height at the unburned site would be expected to result in fundamentally different wind-height relationships at burned and unburned sites (Hagen and Armbrust, 1994; Driese and Reiners, 1997; Campbell and Norman, 1998; Lancaster and Baass, 1998). We predicted our data to reflect this primarily through observed differences in wind speed at the two measurement heights, because our sites were not instrumented with enough anemometers to determine wind profiles and derivative measures (i.e., zero plane displacement, aerodynamic roughness, and friction velocity). Simultaneous wind speed measurements were lower on average at the unburned site, suggesting greater wind attenuation within the unburned plant canopy relative to the burned site. They show greater differences between burned and unburned wind speeds in fall/winter relative to spring/summer, as well.

These observations corresponded with substantial differences in the magnitude and frequency of saltation activity at burned and unburned sites during fall/winter. They also corresponded with the observed greater similarity between sites in spring/summer, when there was a smaller difference between the fraction of the time that saltation was detected at the burned and unburned sites (Table 2). Overall, the frequency of saltation events was low at both sites compared to more erodible landscapes such as sand dunes or playas (Stout, 2004, 2007). These results appear to corroborate findings of previous studies of wind erosion in burned and unburned semiarid rangelands; highlighting the episodic nature of wind erosion in these environments (Vermeire, 2005; Whicker, 2002). Whereas these previous studies of post-fire erosion occurred on sandy soils, our study occurred on finer textured soils in which appreciable erosion has likely not occurred for decades and probably centuries, and so our sites were presumably more prone to erosive losses.

Our saltation activity and sediment flux data for fall/winter show more near-ground sediment transport at the burned compared to unburned sites, providing indirect evidence of the relative erodibility of the burned and unburned soil surfaces (Whicker et al., 2002). The determination of critical aeolian threshold wind speeds provides another measure of erodibility for an aeolian surface, with a higher measured threshold suggesting a lower erodibility and vice versa. Differences amongst the mean threshold values observed during erosion events in October and November provide evidence of temporal variability in erodibility within the burned shrubland. Due to the lack of sustained erosion events at the unburned site, threshold could not be used as a means to compare the relative erodibility of unburned and burned sites. Whicker et al. (2002) suggest fire can result in differences in erodibility at burned and unburned locations due to both the direct effects of burning on the soil surface and to vegetation differences. Our findings appear to indicate that fire increased the potential for wind erosion by temporarily excluding the boundary layer protection provided by the vegetation to the soil surface. Experiments using portable wind tunnels or experimental modification of vegetation would help explain whether the difference in wind erosion potential between burned and unburned sites in our environment is due additionally to direct effects of fire on the erodibility of the soil surface.

We observed a great deal of within-site variability in sediment flux, though this is not evident in our presented results due to the averaging of sediment catch from all collectors within a site. Figure 5 shows a photo taken on June 10, 2007 of a blowout that formed in the preceding fall/winter, post-fire, at the secondary burned site adjacent to a sediment collector. This blowout persisted throughout the spring/summer study period and the nearby collector had consistently higher sediment fluxes than many of the other burned site collectors. Spatial variability in wind erosion within burned sites was described in one previous study (Whicker et al., 2002).

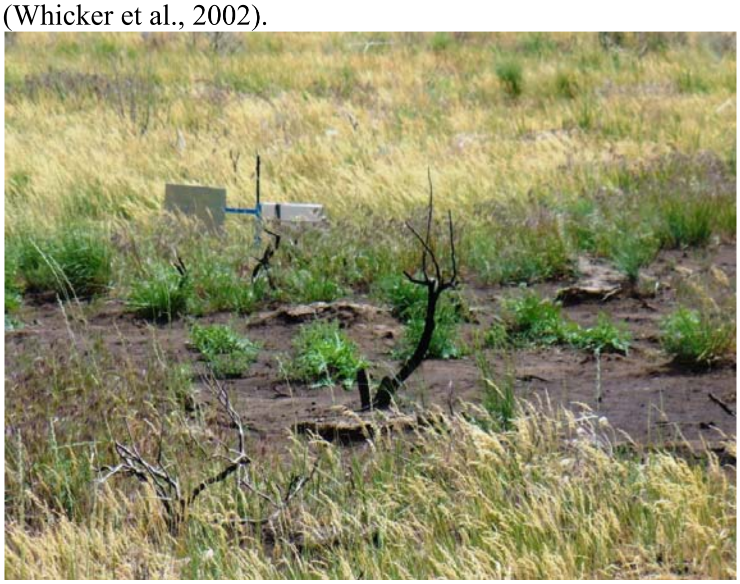

Figure 5. Photo taken on 06/10/07 showing herbaceous vegetation surrounding a blowout that developed in fall, post-fire (note sediment collector in background).

Our sediment flux data provide insight into the longevity of erosion potential differences between the burned and unburned sites post-fire. The sediment mass transport values determined for fall/winter show a decrease in aeolian sediment transport during this time period. The values for the last sampling interval in fall/winter and throughout spring/summer are considerably smaller than those for the first two sampling intervals post-fire. The bulk of wind erosion at our sites, therefore, occurred during the initial months following fire. This is similar to the results of a two-year study of post-prescribed fire wind erosion that reported more aeolian sediment transport at burned relative to unburned sites during the first 5 months following burning (Vermeire et al., 2005). Their study environment was a shrubland in a more humid climate than ours, and erosion of the greatest magnitude occurred at their sites during the dormant season of December through April; several of the months when the ground at our environment would typically be covered by snow. The magnitude and frequency of sediment transport became much more similar at our burned and unburned sites following the emergence of herbaceous vegetation in late spring, though horizontal sediment flux appeared to remain slightly greater at the burned site. This is also comparable to findings of the post-prescribed burn study (Vermeire et al., 2005), in which the researchers suggested that observed differences in spring and summer erosion between the two years of their study might be dependent on interannual variations in climate of the dormant season and resultant degree of vegetation growth in the spring. Both burned and unburned pastures in the Vermeire et al. (2005) study were subjected to grazing disturbance during spring and summer, however.

Our findings suggest that erosion increased following burning, similar to findings of two previous studies specific to wind erosion following wildfire (Whicker et al., 2002; Zobeck et al., 1989). The scope of our findings in this regard is limited by the lack of spatial replication, with only paired comparisons possible between a single burned site and single unburned site throughout the course of the study period. This degree of replication is comparable to previous studies of post-wildfire wind erosion (Whicker et al., 2002; Zobeck et al., 1989), however, and to studies of wind erosion in semiarid shrublands in general (Breshears et al., 2003), and highlights the challenges of instrumenting such study sites to monitor aeolian transport. We present our work as a case study, and hypothesize that greater variability in the frequency and magnitude of erosion events might be expected amongst and within burned and unburned areas, could more sites be instrumented for a spatially explicit characterization of post-wildfire aeolian transport during future research.

In contrast to previous studies (Whicker et al., 2002; Zobeck et al., 1989), our study additionally examined the longevity of increased erosion following wildfire. Our study also took place in cold desert shrub steppe versus warm desert shrubland and prairie of previous studies, highlighting fundamental differences in the potential timing of fire and erosion due primarily to hydroclimatology at our sites versus those previously studied. In our study, the potential for wind erosion initially increased following a wildfire initiated by lightning in an August thunderstorm, decreased throughout the fall and early winter, then remained similarly low through the following late spring and summer, and appeared to be only slightly greater in the burned shrub steppe one year after the wildfire. Future research that examines the relative roles of vegetation removal and burning effects on soil erodibility, with greater spatial and temporal replication, would help land managers determine the degree to which seeding and other restoration efforts which are widely implemented following fires in this region are necessary to stabilize the soil surface following fire in cold desert shrub-steppe.

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LITERATURE CITED

Bagnold, R. A., 1941. The Physics of Blown Sand and Desert Dunes. Methuen, New York.

Breshears, D.D., Whicker, J.J., Johansen, M.P., and Pinder, J.E., 2003. Wind and water erosion and transport in semi-arid shrubland, grassland and forest ecosystems: quantifying dominance of horizontal wind driven transport. Earth Surface Processes and Landforms 28, 189-1209.

Campbell, G.S., Norman, J.M., 1998. An introduction to environmental biophysics. Springer Science + Business Media Inc., New York, NY, USA.

Chadwick, O.A., Derry, L.A., Vitousek, P.M. Huebert, B.J., Hedin, L.O., 1999. Changing sources of nutrients during four million years of ecosystem development. Nature 397, 491-497.

Clawson, K.L, Start, G.E., Ricks, N.R. (eds.), 1989. Climatography of the Idaho National Engineering Laboratory, 2nd Edition, DOE/ID-12118. U. S. Department of Commerce, National Oceanic and Atmospheric Administration, Environmental Research Laboratories, Air Resources Laboratory, Field Research Division, Idaho Falls, ID, USA.

Driese, K.L., Reiners, W.A., 1997. Aerodynamic roughness parameters for semi-arid natural shrub communities of Wyoming, USA. Agricultural and Forest Meteorology 88, 1-14.

Hagen, L.J., Armbrust, D.V., 1994. Plant canopy effects on wind erosion saltation. Transactions of the ASAE 37, 461-465

Lancaster, N., Baas, A., 1998. Influence of vegetation cover on sand transport by wind: field studies at Owens Lake, California. Earth Surface Processes and Landforms 23, 69-82.

NCSS Web Soil Survey. http://websoilsurvey.nrcs.usda.gov/app/.

Okin, G.S., Gillete, D.A., Herrick, J.E., 2006. Multi-scale controls on and consequences of aeolian processes in landscape change in arid and semi-arid environments. Journal of Arid Environments 65, 253-275.

Okin, G.S., Gillete, D.A., 2004. Modeling wind erosion and dust emission on vegetated surfaces, In *Spatial Modeling of the Terrestrial Environment*., edited by R. Kelly and N.A. Drake, pp. 137-156, John Wiley, N.J.

Okin, G.S., Mahowald, N., Chadwick, O.A., Artaxo, P., 2004. Impact of desert dust of phosphorous in terrestrial ecosystems. Global Biogeochemical Cycles 18.

Reynolds, J.F., Belnap, J., Reheis, M., Lamothe, P., Luiszer, F., 2001. Eolian dust in Colorado Plateau soils: nutrient inputs and recent change in source. Proceedings of the National Academy of Sciences 98, 7123-7127.

Stout, J.E., 2004. A method for establishing the critical threshold for aeolian transport in the field. Earth Surface Processes and Landforms 29, 1195-1207.

Stout, J.E., 2007. Simultaneous observations of the critical threshold of two surfaces. Geomorphology 85, 3-16.

USGS. http://geomaps.wr.usgs.gov/parks/province/basinrage.html

Vermeire, L.T., Wester, D.B., Mitchell, R.B., Fuhlendorf, S.D., 2005. Fire and grazing effects on wind erosion, soil water content, and soil temperature. Journal of Environmental Quality 34, 1559-1565.

Whicker, F.W., Hinton, T.G., MacDonell, M.M., Pinder, J.E., Habegger, L.J., 2004. Avoiding destructive remediation at DOE sites. Science 303, 1615-1616.

Whicker, J.J., Pinder, J.E., Breshears, D.D., 2006. Increased wind erosion from forest wildfire: implications for contaminant-related risks. Journal of Environmental Quality 35, 468-478.

Whicker, J.J., Breshears, D.D., Wasiolek, P.T., Kirchner, T.B., Tavani, R.A., Schoep, D.A., Rodgers, J.C., 2002. Temporal and spatial variation of episodic wind erosion in unburned and burned semiarid shrubland. Journal of Environmental Quality 31, 599-612.

Zobeck, T.M., Fryrear, D.W., Pettit, R.D., 1989. Management effects on wind-eroded sediment and plant nutrients. Journal of Soil and Water Conservation 44, 160-163.