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Feedbacks between fires and wind erosion in heterogeneous arid lands

Sujith Ravi,[1] Paolo D'Odorico,[1] Ted M. Zobeck,[2] Thomas M. Over,[3] and Scott L. Collins[4]

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[1] Shrub encroachment, a widespread phenomenon in arid landscapes, creates ''islands of fertility'' in degraded systems as wind erosion removes nutrient-rich soil from intercanopy areas and deposits it in nearby shrub-vegetated patches. These islands of fertility generally are considered to be irreversible. Recently, fire has been observed to alter this pattern of resource heterogeneity through the redistribution of nutrients from the fertile islands of burnt shrubs to the surrounding bare soil areas. Despite the recognized relevance of both fires and wind erosion to the structure and function of arid ecosystems, the interactions between these two processes remains poorly understood. This study tests the hypothesis that fire-induced soil hydrophobicity developing in the soils beneath burned shrubs enhances soil erodibility by weakening the interparticle wet-bonding forces. To test this hypothesis, the effects of grass and shrub fires on changes in soil erodibility and on the intensity of fire-induced soil water repellency are compared at both the field and patch scales in heterogeneous arid landscapes. Higher water repellency was observed in conjunction with a stronger decrease in wind erosion threshold velocity around the shrubs than in grass-dominated patches affected by fire, while neither water repellency nor changes in threshold velocity was noticed in the bare soil interspaces. Thus, fires are found to induce soil hydrophobicity and to consequently enhance soil erodibility in shrub-vegetated islands of fertility. These processes create temporally dynamic islands of fertility and contribute to a decrease in resource heterogeneity in aridland ecosystems following fire.

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1. Introduction

[3] In arid environments erosion processes redistribute soil particles and nutrients [*Schlesinger et al.*, 1990; *Okin and Gillette*, 2001], thereby affecting soil texture and soil water holding capacity [*Lyles and Tatarko*, 1986; *Offer et al.*, 1998] with consequent effects on the productivity, composition and spatial patterns of vegetation [*Schlesinger et al.*, 1990]. In these landscapes soil erosion is mainly due to aeolian processes [*Breshears et al.*, 2003], which maintain the local heterogeneities in nutrient and vegetation distribution through the removal of nutrient-rich soil from intercanopy areas and the subsequent deposition onto vegetated areas [*Schlesinger et al.*, 1990; *Okin and Gillette*, 2001]. Thus, wind erosion is often invoked as a major factor enhancing and maintaining soil heterogeneity, particularly in shrub encroached arid landscapes [*Okin and Gillette*, 2001].

[4] Dryland ecosystems are often prone to disturbances like fires and grazing, which may render soils more susceptible to wind erosion with important impacts on regional and global climate, human health, biogeochemical cycles, and desertification [*Nicholson*, 2000; *Rosenfield et al.*, 2001; *Fryrear*, 1985; *Whicker et al.*, 2006; *Duce and Tindale*, 1991; *Schlesinger et al.*, 1990]. Fires modify the interactions between eco-hydrological and land surface processes [*Ludwig et al.*, 1997], expose the soil surface to the erosive action of winds, and affect the relative abundance and distribution of shrubs and grasses in arid ecosystems [*Scholes and Archer*, 1997; *van Langevelde et al.*, 2003; *Sankaran et al.*, 2004]. On the other hand, vegetation type and patterns affect both the intensity and frequency of fires [*Anderies et al.*, 2002; *van Wilgen et al.*, 2003]. Although both wind erosion and fires play an important role in the dynamics of arid and semiarid ecosystems, the interactions between these two processes remain unknown. Fire is now a commonly used management tool in many aridland ecosystems to reduce shrub cover and to enhance grass growth. Thus, understanding how fire affects soil structure and resource heterogeneity is of fundamental and practical importance in systems where aeolian processes predominate.

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[1] Department of Environmental Sciences, University of Virginia, Charlottesville, Virginia, USA.

[2] Wind Erosion and Water Conservation Research Unit, USDA, Agricultural Research Service, Lubbock, Texas, USA.

[3] Department of Geology/Geography, Eastern Illinois University, Charleston, Illinois, USA.

[4] Department of Biology, University of New Mexico, Albuquerque, New Mexico, USA.

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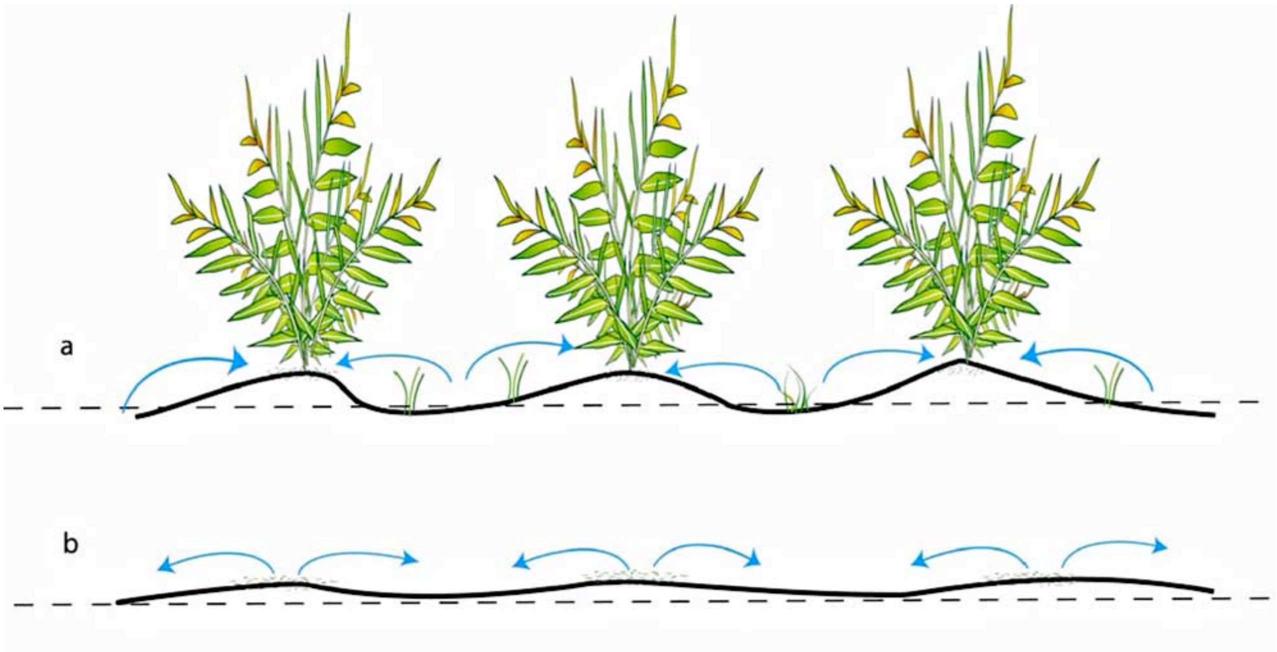

Figure 1. (a) Diagram showing the formation of islands of fertility around the shrub patches and increase in microtopography. (b) Diagram showing postfire enhancement of wind erosion leading to redistribution of resources to bare interspaces and reduction in microtopography.

[5] Several studies have shown that fires in shrub-encroached grasslands favor grass regrowth and limit further shrub encroachment [*van Auken*, 2000; *van Wilgen et al.*, 2003]. Recent studies [*White et al.*, 2006] have shown that, while the process of shrub encroachment favors the formation of a heterogeneous landscape with the concentration of resources beneath the shrub canopies and the formation of ''fertility islands'' (Figure 1a), fires tend to destroy this heterogeneity by enhancing wind-induced particle transport and erosion from fertile islands affected by the burning of shrub biomass. In fact, subsequent to fire occurrences microtopographic differences between vegetated islands and bare interspaces decrease [*White et al.*, 2006], indicating that the resources accumulated in the fertility islands are redistributed onto the interspaces, thereby reducing the spatial heterogeneity of the system (Figure 1b). In addition, soil organic matter increased in bare areas relative to vegetated patches, and in some sites, soil resources (e.g., nitrogen) were more homogeneously distributed for up to 22 months following fire. These findings are consistent with the observation that burned areas exhibit lower threshold velocities for wind erosion and higher volumes of soil loss than in similar unburned areas [*Whicker et al.*, 2002]. This difference in soil erodibility determines important structural changes in the landscape through postfire translocation of soil resources from burned ''fertility islands'' to bare soil areas. As a result, we hypothesize that fertility islands are not static, but rather dynamic features of the landscape. Thus wind erosion and fires influence the dynamics of dryland landscapes, and the interactions between these two processes play a major role in determining the composition and structure of vegetation.

[6] However, an important component of these dynamics remains unexplained, as it is unclear why adjacent sites, with similar surface roughness and exposed to the same winds, should exhibit differing susceptibility to wind erosion [*Whicker et al.*, 2002]. Recent studies on soils treated in the laboratory with water-repellent compounds [*Ravi et al.*, 2006a] have experimentally shown that soil hydrophobicity enhances soil erodibility. By affecting the strength of interparticle wet-bonding forces, water repellency enhances soil erodibility, causing a drop in wind erosion threshold velocity, the minimum velocity for erosion to occur. In this study we show that fire-induced water repellency creates the same effect, and we provide the first experimental evidence that postfire enhancement of soil erodibility is due to fire-induced soil hydrophobicity. Fires are known for having a major impact on infiltration, runoff and water erosion [e.g., *DeBano*, 2000]. The postfire increase in runoff and soil erosion is caused by the decrease in infiltration capacity resulting from fire-induced water repellency [*Krammes and DeBano*, 1965; *DeBano*, 1966]. In fact, burning vegetation releases fatty acids onto the underlying soil, with consequent effects on the physical-chemical properties of the soil grain surfaces; in particular, these organic compounds increase the contact angle formed by the air-water interface with the soil grains, thereby affecting the dynamics of moisture retention and the strength of interparticle bonding forces [*Ravi et al.*, 2006a].

[7] In this paper, a study of the soil hydrophobicity caused by fire in arid landscapes and the subsequent effect on wind erosion thresholds in two different arid ecosystems is presented. In the first part of the study, two systems with different land cover, an arid grassland and a shrubland, were compared. The second part of the study concerns a heterogeneous arid landscape with a mosaic of vegetated shrub and grass patches separated by bare interspaces. The first part of the study investigates the relative importance of plant communities on the enhancement of soil susceptibility to wind erosion. The second part concentrates on differ-

ences in soil erodibility within a heterogeneous ecosystem, i.e., in grass- and shrub-dominated soil patches. Here we hypothesize that the postfire enhancement of soil erosion is stronger in shrub-dominated than in grass-dominated soils and that this difference is the result of the different level of soil-water repellency developed by the burning of shrub versus grass vegetation [*Adams et al.*, 1970]. We argue that the stronger enhancement in soil erodibility induced by burning shrubs causes the observed decrease in soil heterogeneity. To test this hypothesis, we investigate, with field and laboratory measurements, changes in soil erodibility and other soil properties in soil plots affected by burning biomass.

2. Materials and Methods

[8] The burn experiments were conducted in two different ecosystems from the southwestern U.S which are prone to fires and wind erosion, namely, the Cimarron National Grassland (KS) and the Sevilleta National Wildlife Refuge (NM). The Cimarron National Grassland (37°7.29′N, 101°53.81′W) is a short grass prairie ecosystem (Blue grama), with significant shrub encroachment (Sage brush and Yucca) in some areas. On February 5, 2006, a large fire burned approximately 1700 ha in part of the Cimarron National Grassland [*U.S. Forest Service*, 2006]. Thus, after all of the above-ground vegetation had burned, the soil surface was left exposed to high wind erosion activity. Two sites were chosen at Cimarron: a pure grassland site and a grassland encroached by shrubs. Two sets of soil samples were collected on burned and unburned soils across the fire line on three replicated pairs of (burned and unburned) plots at each site. The second set of experiments were conducted at the Sevilleta National Wildlife Refuge, located in the northern Chihuahuan Desert approximately 80 km south of Albuquerque, New Mexico (N 34°23.961′ and W 106°55.710′). The site chosen for our study was a desert grassland/scrubland with a mosaic of soil patches dominated by grasses (*Sporobolis*) and shrubs (Four wing saltbush and Snake weed) with bare interspaces. In the three field sites used for the study grass cover was minimal near the base of the shrubs and the landscape was heterogeneous with distinct patches of grasses and shrubs with bare interspaces. This patchy landscape is typical for these arid shrublands [*Kurc and Small*, 2004]. Soil samples were randomly collected at each site from an area of about 5 m^2 before and after the prescribed burn. As the focus of this study is on wind erosion and fires, soil samples were taken only from the surface (top 2 cm) under the grasses, around the shrubs, and from the bare interspaces in the three replicated plots before and after the prescribed burning.

[9] The soil samples were passed through a 2 mm sieve and kept in metal trays for 5–6 hours before each wind tunnel test to equilibrate with the ambient atmospheric humidity and temperature. Surface soil moisture changed only in response to fluctuations in ambient air humidity as there was no control on the atmospheric humidity and temperature, and the soil samples were not artificially wetted or dried [*Ravi et al.*, 2004; *Ravi and D'Odorico*, 2005]. To account for the effect of air humidity on surface moisture content, the wind tunnel tests were repeated at two ranges of relative humidity: 10–30% and 40–60%. A nonrecirculating wind tunnel (10.0 m long, 0.5 m wide and 1.0 m high) was used for this study. The soils were placed in the wind tunnel on removable metal trays (1.5 cm × 46.0 cm × 100.0 cm). The wind velocity was measured at different heights inside the tunnel using a series of Pitot tubes connected to pressure transducers. These measurements were used to calculate the surface roughness (Z_o = 1.17 mm) and to express the wind speed (v) in terms of shear velocity (u_*). Saltation was measured by a particle impact sensor (SENSIT), soil temperature by an infrared thermometer (Exergen Corp, IRT/C.2 with Type K Germanium lens), near surface temperature and relative humidity (2 mm from surface) by a RH/T probe (Vaisala, Inc. Humitter 50U). For each wind tunnel test, the air flow was initially increased stepwise to attain a wind speed just below the estimated threshold value and then increased slowly till the particle sensor indicated particle movement, i.e., an abrupt increase from zero to more than 10 particle impacts per second. Three replicates of the control and burned soils were used for each set of wind tunnel tests and these sets were repeated at two different humidity ranges. Statistical tests (t-test) were carried out to assess the significance of the results.

[10] Determination of several soil properties was needed for interpretation of the experimental results. These included particle size distribution and soil wettability (i.e., the degree of hygroscopicity/hydrophobicity of the soil grains). Fire-induced water repellency was determined using both the water drop penetration time (WDPT) and the molarity of ethanol solutions (MED) instantaneously infiltrating into the soil [e.g., *Letey*, 2001]. For the WDPT (laboratory method) a pipette was used to place water drops on the soil surface. The time required for the drop to penetrate the surface was measured. The water drop penetration (WDPT) time was determined for each sample as the mean WDPT for 10 droplets. In the molarity of an ethanol droplet (MED) test, standardized solutions of ethanol in water of known surface tensions were used to characterize the severity of water repellency in the soil [*Doerr*, 1998; *Roy and McGill*, 2002]. Drops of the ethanol-water solutions with increasing concentrations were placed on the surface of the water repellent soil sample. As the molarity of the solution increases, the surface tension decreases and at a certain critical concentration (or critical surface tension, CST) the drop penetrates the soil surface instantaneously (within 3 s). Values of WDPT and MED for the soils used in the study are reported in Table 1. Soil texture was determined using the using the standard hydrometer method [*ASTM*, 1981]. A soil hydrometer (Fisher brand Specific Gravity Scale Soil Hydrometer) was calibrated to measure the specific gravity of the soil suspension; the size fractions were calculated based on the settling time of the suspended particles (Table 1). (Identification of experimental apparatus is for information purposes only and does not imply endorsement by ARS-USDA.)

3. Results

[11] The results from the wind tunnel tests on the soils from the Cimarron National Grassland show that the threshold friction velocity of burned soils from both the grass-dominated and shrub-dominated areas were significantly less than for the control soils collected from adjacent unburned areas. Moreover the threshold velocity values

Table 1. Textural and Wetting Properties of the Soils Used in This Study

Study Sites	Land Cover	Particle Size Distribution			Soil Hydrophobicity[a]	
		Clay, %	Silt, %	Sand, %	WDPT, s	MED, molarity
Cimarron National Grassland, KS	Shrubland	9	12	79	120	3
	Grassland	8	10	82	30	1
Sevilleta National Wildlife Refuge, NM	Shrub patch	28	19	53	50	2
	Grass patch	33	26	41	<10 s	<1
	Bare interspaces	25	18	57	0	0

[a] Soil hydrophobicity was not observed in control soils.

were significantly different for the soils from the burned and the control plots even though the surface moisture contents were not significantly different. These differences were consistently observed for the two humidity ranges considered in this study (Figures 2a and 2b). The differences in threshold velocities between the soils from control plots and burned plots were found to be statistically significant both for the grassland ($p < 0.001$ in the 10–30% RH range and $p < 0.001$ in the 40–60% RH range) and shrubland at ($p < 0.001$ in the 10–30% RH range and $p < 0.00001$ in the 40–60% RH range). Further for each treatment the threshold values increased with increasing values of air humidity, indicating a clear dependence of threshold velocity on air humidity as seen in our previous studies [e.g., *Ravi et al.*, 2006a, 2006b]. Higher postfire enhancement of soil erodibility resulted from the burning of shrubs than of grasses (Figures 2a and 2b) as evidenced by the results of the t-test was carried out between the differences in threshold shear velocities of burned and control plots in the grassland and shrubland. The test clearly showed that - in both humidity ranges considered in this study - the differences between threshold shear velocities of control and burn plots were significantly higher in the shrubland sites compared to grassland sites ($p < 0.026$ in the 10–30% RH range and $p < 0.005$ in the 40–60% RH range). These results indicate that this difference is severe at higher humidity values. Further, the severity of fire-induced water repellency was higher for the shrubs than for grasses (Table 1). The surface soil moisture of the burned and control soils were not significantly different for both the humidity ranges considered, whereas the surface soil moisture increased with increasing air humidity as reported in previous studies on unburned soils treated in the lab with water-repellent chemicals [e.g., *Ravi et al.*, 2006a].

[12] The second part of the study focused on a heterogeneous arid landscape at the Sevilleta National Wildlife Refuge (NM), which exhibits a mosaic of patches dominated by shrubs, grasses and bare soil interspaces. In this system fires induced higher water repellency in the soils around the shrubs compared to grass-dominated patches, while no water repellency was noticed in the bare interspaces (Table 1). Laboratory wind tunnel tests showed that even before the burn experiment different erosion thresholds existed in the same field due to the textural heterogeneity of soils sampled under shrubs, grasses and in bare soil patches (Figure 3). After burning, wind erosion thresholds decreased in the vegetated patches. The differences in threshold velocities between the soils from control plots and burned plots were found to be statistically significant both for the soils under grass patches ($p < 0.001$ in the 10–30% RH range and $p < 0.03$ in the 40–60% RH range) and under shrubs at ($p < 0.0001$ in the 10–30% RH range and $p < 0.003$ in the 40–60% RH range), while – as expected – this difference in threshold velocity was insignificant between the control and the burned bare interspaces($p > 0.40$ in the 10–30% RH range and $p > 0.49$ in the 40–60% RH range). Moreover, the decrease in threshold velocity was stronger in the case of soils affected by the burning of shrubs; in fact, in both the humidity ranges the difference between threshold shear velocities of control and burn plots were significantly

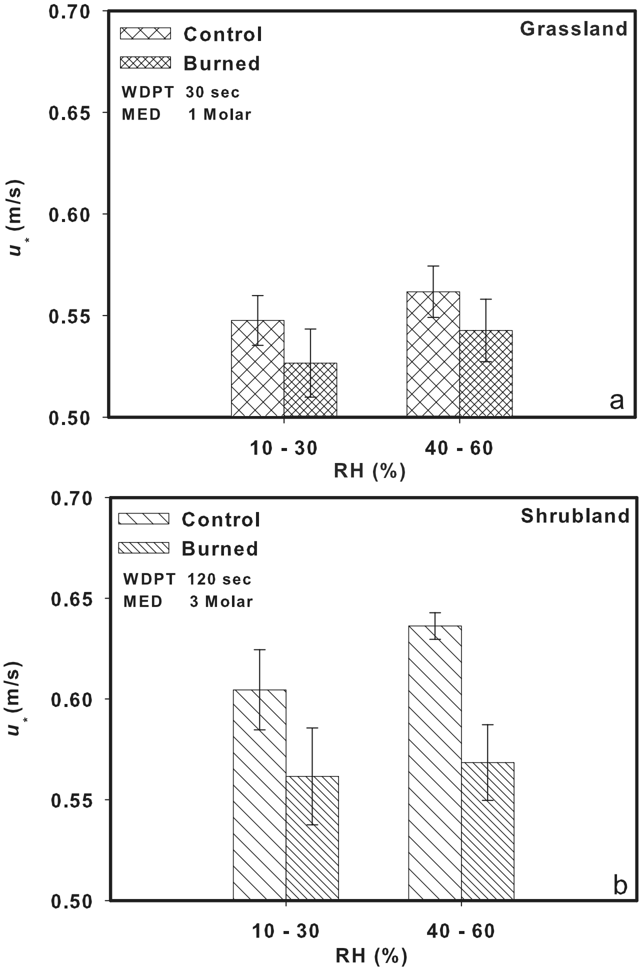

Figure 2. Threshold friction velocity (u_*) as a function of atmospheric relative humidity (RH) as determined by wind-tunnel tests for control and burned surface soil from the Cimarron National Grassland. The error bars represent the standard deviation of threshold shear velocity within each class of relative humidity. The WDPT and MED values are for the burned soil.

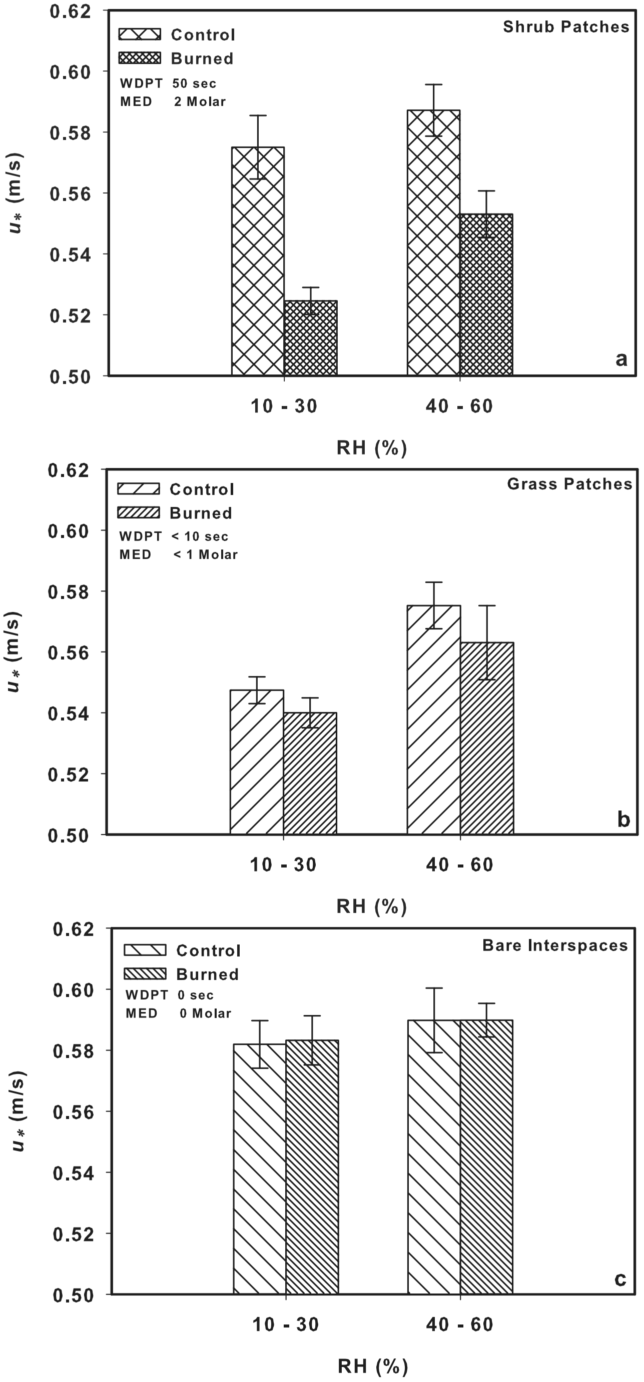

Figure 3. Threshold friction velocity (u_*) as a function of atmospheric relative humidity (RH) as determined by wind-tunnel tests for control and burned soils from shrub patches, grass patches, and bare interspaces at the Sevilleta National Wildlife Refuge. The error bars represent the standard deviation of threshold shear velocity within each class of relative humidity. The WDPT and MED values are for the burned soil.

higher in soils from burnt shrub patches compared to soils from burnt grass patches ($p < 0.003$ in the 10–30% RH range and $p < 0.013$ in the 40–60% RH range). Thus, by decreasing the wind erosion thresholds in burnt shrub patches, fires enhance the erodibility of soils from beneath the burnt shrubs, thereby contributing to nutrient loss from the fertility islands and to the consequent reduction in landscape heterogeneities.

4. Discussion

[13] The experimental results support the hypothesis that fires enhance soil susceptibility to wind erosion in areas affected by biomass burning and that this effect is more significant in the case of shrub-dominated compared to grass-dominated areas, both at the field (Figures 2a and 2b) and at the patch scale (Figures 3a–3c). In the case of patchy landscapes, the postfire redistribution of soil and nutrients from shrub patches to bare (or sparsely grass-covered) interspaces can occur only if the soils in and around the shrub patches become more erodible compared to the interspaces. Our results indicate that this effect can occur in the field due to the relatively higher fire-induced soil hydrophobicity occurring in and around the shrub patches, compared to either grass-dominated or bare soils. Hence, following fire events shrub-dominated fertility islands exhibit lower erosion thresholds compared to grass patches and bare interspaces.

[14] To assess the magnitude and significance of the enhancement of soil erodibility resulting from the postfire decrease in threshold shear velocity, we used wind velocity records taken (at 2 m height) from a bare soil plot at the Sevilleta site for a three-week period in the middle of the 2007 windy season (March–April). These velocity measurements were compared with the threshold velocity values determined in the wind tunnel. To this end, we used the Prandtl-von Karman logarithmic law [e.g., *Campbell and Norman*, 1998] to convert the threshold shear velocities measured for burned and control soils (Figure 3) into threshold wind speed values at 2 m height. The decrease in threshold velocity (at 2 m height) between burned and control soils was in the range of 1.0–1.5 m/s, which corresponds, in this short wind record, to a 70% increase in the number of occurrences with wind velocity exceeding the threshold velocity. This result indicates that the decrease in threshold velocity observed in the burned soils can cause a significant increase in wind erosion activity after fires.

[15] The level of soil hydrophobicity developed by fires depends on fire intensity, vegetation cover, and soil texture. The experimental results from our study show that a higher hydrophobicity develops in the soils beneath burning shrub vegetation (Table 1). This higher repellency found under the burnt shrubs is explained by the fact that shrubs typically contain more water-repellent organic compounds and are subjected to higher fire intensities compared to grasses [*Tyler*, 1995; *Moreno and Oechel*, 1991]. Indeed, at the Sevilleta, fire temperatures beneath shrubs were 60–100°C higher than in grass patches. The shrub patches are also characterized by high surface accumulations of organic matter and leaf debris which can enhance the severity of water repellency induced by fire [*Pierson et al.*, 2001]. Further, the soil under shrub patches contained more sand (Table 1), and the severity of water repellency is higher in the case of sandy soils due to their relatively smaller specific surface area [*DeBano*, 2000]. The higher fire-induced hydrophobicity explains the stronger decrease in wind erosion threshold velocity observed in the soil patches that

prior to burning were vegetated by shrubs, as compared to soils under burned grasses or in the bare interspaces. In the latter case no decrease in wind erosion threshold was observed, due to the limited effect of fires on the bare soil patches.

[16] The enhancement of soil erodibility by fires may also depend on other factors in addition to soil water repellency, including the formation of cryptobiotic crusts by algae, fungi and soil bacteria, which may enhance the soil-water repellency [e.g., *Savage et al.*, 1969] and are susceptible to destruction by fire. Previous laboratory studies showing the ability of soil hydrophobicity to weaken interparticle bonding forces [*Ravi et al.*, 2006a] were carried out on clean sands treated with water repellent organic compounds and were not affected by confounding factors such as microbial crusts or organic matter. The effects of microbial crusts are not considered in this study because they are mostly found in the bare interspaces [*West*, 1990; *Schlesinger and Pilmanis*, 1998; *Stursova et al.*, 2006] as they avoid competition with vegetation for resources [*Harper and Belnap*, 2001; *Li et al.*, 2002]. Although we acknowledge the role played by microbial crust in the process of soil erosion, it is not clear how they could contribute to the postfire enhancement of soil erodibility from fertility islands. Conversely, the fact that fires induce soil-water repellency [e.g., *DeBano*, 2000; *Doerr et al.*, 2000] is well-established, and the ability of repellency to enhance soil erodibility has been tested in the laboratory and mechanistically explained [*Ravi et al.*, 2006a]. In this study we also eliminated the effects due to the higher topography of shrub mounds and the soil disturbances caused animals, by selecting study sites in areas where these effects were negligible. This study shows that the levels of soil hydrophobicity developed by typical rangeland fires are able to enhance soil erodibility. Moreover this effect is stronger in shrub than in grass patches, and nonexistent in the bare interspaces. Thus, resource islands in aridland ecosystems are not static but rather highly dynamic patch types in response to fire and perhaps other disturbances such as drought. These differences in the enhancement of soil erodibility provide the mechanism behind the recent observational evidence of loss in landscape heterogeneity subsequent to fires [e.g., *White et al.*, 2006] and demonstrate the possible value of prescribed fire as a tool to mitigate the early stages of the processes of fertility island formation and desertification.

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References

Adams, S., B. R. Strain, and M. S. Adams (1970), Water-repellent soils, fire, and annual plant cover in a desert scrub community of southeastern California, *Ecology*, *51*, 696–700.

American Society for Testing and Materials (ASTM) (1981), *Annual Book of ASTM Standards*, Philadelphia, Pa.

Anderies, J. M., M. A. Janssen, and B. H. Walker (2002), Grazing, management, resilience and the dynamics of fire-driven rangeland system, *Ecosystems*, *5*, 23–44.

Breshears, D. D., J. J. Whicker, M. P. Johansen, and J. E. Pinder III (2003), Wind and water erosion and transport in semiarid shrubland, grassland, and forest ecosystems: Quantifying dominance of horizontal wind-driven transport, *Earth Surf. Processes Landforms*, *28*, 1189–1209.

Campbell, G. S., and J. M. Norman (1998), *An Introduction to Environmental Biophysics*, 286 pp., Springer, Berlin.

DeBano, L. F. (1966), Formation of non-wettable soils involves heat transfer mechanism, *Res. Note PSW-132,* 8 pp., For. Serv., U.S. Dep. of Agric., Pac. Southwest For. and Range Exp. Stn., Berkeley, Calif.

DeBano, L. F. (2000), The role of fire and soil heating on water repellence in wildland environments: a review, *J. Hydrol.*, *231*, 195–206.

Doerr, S. H. (1998), On standardizing the "Water Drop Penetration Time" and the "Molarity of an Ethanol Droplet" technique to classify soil hyrdophobicity: A case study using medium textured soils, *Earth Surf. Processes Landforms*, *23*(7), 663–668.

Doerr, S. H., R. A. Shakesby, and R. P. D. Walsh (2000), Soil water repellency: its causes, characteristics and hydro-geomorphological significance, *Earth Sci. Rev.*, *51*, 33–65.

Duce, R. A., and N. W. Tindale (1991), Atmospheric transport of iron and its deposition in the ocean, *Limnol. Oceanogr.*, *36*, 1715–1726.

Fryrear, D. W. (1985), Soil cover and wind erosion, *Trans. ASAE*, *28*(3), 781–784.

Harper, K. T., and J. Belnap (2001), The influence of biological soil crusts on mineral uptake by associated vascular plants, *J. Arid Environ.*, *47*, 347–357.

Krammes, J. S., and L. F. DeBano (1965), Soil wettability: a neglected factor in watershed management, *Water Resour. Res.*, *1*, 283–286.

Kurc, S. A., and E. E. Small (2004), Dynamics of evapotranspiration in semiarid grassland and shrubland ecosystems during the summer monsoon season, central New Mexico, *Water Resour. Res.*, *40*, W09305, doi:10.1029/2004WR003068.

Letey, J. (2001), Causes and consequences of fire-induced soil water repellency, *Hydrol. Processes*, *15*, 2867–2875.

Li, X. R., X. P. Wang, T. Li, and J. G. Zhang (2002), Microbiotic soil crust and its effect on vegetation and habitat on artificially stabilized desert dunes in Tengger Desert, North China, *Biol. Fertil. Soils.*, *35*(3), 147–154.

Ludwig, J. A., D. J. Tongway, D. O. Freudenberger, J. C. Noble, and K. C. Hodgkinson (Eds.) (1997), *Landscape Ecology, Function and Management: Principles From Australia's Rangelands*, CSIRO, Melbourne, Australia.

Lyles, L., and J. Tatarko (1986), Wind erosion effects on soil texture and organic matter, *J. Soil Water Conserv.*, *46*, 191–193.

Moreno, J. M., and W. C. Oechel (1991), Fire intensity effects on germination of shrubs and herbs in southern California chaparral, *Ecology*, *72*(6), 1993–2004.

Nicholson, S. (2000), Land surface processes and Sahel Climate, *Rev. Geophys.*, *38*, 117–139.

Offer, Z. Y., E. Zaady, and M. Shachak (1998), Aeolian particle input to the soil surface at the northern limit of the Negev desert, *Arid Soil Res. Rehab.*, *12*(1), 55–62.

Okin, G. S., and D. A. Gillette (2001), Distribution of vegetation in wind dominated landscapes: Implications for wind erosion modeling and landscape processes, *J. Geophys. Res.*, *106*, 9673–9683.

Pierson, F. B., P. R. Robichaud, and K. E. Spaeth (2001), Spatial and temporal effects of wildfire on the hydrology of a steep rangeland watershed, *Hydrol. Processes*, *15*(15), 2905–2916.

Ravi, S., and P. D'Odorico (2005), A field-scale analysis of the dependence of wind erosion threshold velocity on air humidity, *Geophys. Res. Lett.*, *32*, L21404, doi:10.1029/2005GL023675.

Ravi, S., P. D'Odorico, T. M. Over, and T. M. Zobeck (2004), On the effect of air humidity on soil susceptibility to wind erosion: The case of air-dry soils, *Geophys. Res. Lett.*, *31*, L09501, doi:10.1029/2004GL019485.

Ravi, S., P. D'Odorico, B. Herbert, T. Zobeck, and T. M. Over (2006a), Enhancement of wind erosion by fire-induced water repellency, *Water Resour. Res.*, *42*, W11422, doi:10.1029/2006WR004895.

Ravi, S., T. M. Zobeck, T. M. Over, G. S. Okin, and P. D'Odorico (2006b), On the effect of wet bonding forces in air-dry soils on threshold friction velocity of wind erosion, *Sedimentology*, *53*(3), 597–609.

Rosenfield, D., Y. Rudich, and R. Lahav (2001), Desert dust suppressing precipitation: A possible desertification feedback loop, *Proc. Natl. Acad. Sci.*, *98*(11), 5975–5980.

Roy, J. L., and W. B. McGill (2002), Assessing soil water repellency using the molarity of ethanol droplet (MED) test, *Soil Sci.*, *167*(2), 83–97.

Sankaran, M., J. Ratnam, and N. P. Hanan (2004), Tree-grass coexistence in savannas revisited – insights from an examination of assumptions and mechanisms invoked in existing models, *Ecol. Lett.*, *7*, 480–490.

Savage, S. M., J. P. Martin, and J. Letey (1969), Contribution of some soil fungi to natural and heat-induced water repellency in sand, *Soil Sci. Soc. Am. Proc.*, *33*, 405–409.

Schlesinger, W. H., and A. M. Pilmanis (1998), Plant soil interactions in deserts, *Biogeochemistry*, *42*, 169 – 187.

Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford (1990), Biological feedbacks in global desertification, *Science*, *247*(4946), 1043 – 1048.

Scholes, R. J., and S. R. Archer (1997), Tree-grass interactions in savannas, *Ann. Rev. Ecol. Syst.*, *28*, 517 – 544.

Stursova, M., C. L. Crenshaw, and R. L. Sinsabaugh (2006), Microbial response to long-term N deposition in a semiarid grassland, *Microb. Ecol.*, *51*(1), 90 – 98.

Tyler, C. M. (1995), Factors contributing to postfire seedling establishment in chaparral: Direct and indirect effects of fire, *J. Ecol.*, *83*(6), 1009 – 1020.

U.S. Forest Service (2006), Investigation continues into wildfire on Cimarron Grassland, Pueblo, Colo. (Available at http://www.fs.fed.us/r2/psicc/news/2006/index.shtml)

van Auken, O. W. (2000), Shrub invasions of North American semiarid grasslands, *Ann. Rev. Ecol. Syst.*, *31*, 197 – 215.

van Langevelde, F., et al. (2003), Effects of fire and herbivory on the stability of savanna ecosystems, *Ecology*, *84*(2), 337 – 350.

van Wilgen, B. W., W. S. W. Trollope, H. C. Biggs, A. L. F. Potgieter, and B. H. Brockett (2003), Fire as a driver of ecosystem Variability, in *The Kruger Experience: Ecology and Management of Savanna Heterogeneity*, edited by J. T. Du Toit, K. H. Rogers, and H. C. Biggs, pp. 149 – 170, Island Press, Washington, D. C.

West, N. E. (1990), Structure and function of microphytic soil crusts in wildlife ecosystems of arid to semi-arid regions, *Adv. Ecol. Res.*, *20*, 179 – 223.

Whicker, J. J., D. D. Breshears, P. T. Wasiolek, T. B. Kirchner, R. A. Tavani, D. A. Schoep, and J. C. Rodgers (2002), Temporal and spatial variation of episodic wind erosion in unburned and burned semiarid shrubland, *J. Environ. Qual.*, *31*, 599 – 612.

Whicker, J. J., J. E. Pinder, D. D. Breshears, and C. F. Eberhart (2006), From dust to dose: Effects of forest disturbances on increased inhalation exposure, *Sci. Total Environ.*, *368*(2 – 3), 519 – 530.

White, C. S., R. L. Pendleton, and B. K. Pendleton (2006), Response of two semiarid grasslands to a second fire application, *Rangeland Ecol. Manage.*, *53*, 52 – 61.

S. L. Collins, Department of Biology, University of New Mexico, Albuquerque, NM 87131, USA. (scollins@sevilleta.unm.edu)

P. D'Odorico and S. Ravi, Department of Environmental Sciences, University of Virginia, Charlottesville, VA 22904-4123, USA. (paolo@virginia.edu; sujith@virginia.edu)

T. M. Over, Department of Geology/Geography, Eastern Illinois University, Charleston, IL 61920, USA. (tmover@eiu.edu)

T. M. Zobeck, Wind Erosion and Water Conservation Research Unit, USDA, Agricultural Research Service, Lubbock, TX 79415, USA. (tzobeck@lbk.ars.usda.gov)