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Temporal and Spatial Variation of Episodic Wind Erosion in Unburned and Burned Semiarid Shrubland

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ABSTRACT

Redistribution of soil, nutrients, and contaminants is often driven by wind erosion in semiarid shrublands. Wind erosion depends on wind velocity (particularly during episodic, high-velocity winds) and on vegetation, which is generally sparse and spatially heterogeneous in semiarid ecosystems. Further, the vegetation cover can be rapidly and greatly altered due to disturbances, particularly fire. Few studies, however, have evaluated key temporal and spatial components of wind erosion with respect to (i) erosion rates on the scale of weeks as a function of episodic high-velocity winds, (ii) rates at unburned and burned sites, and (iii) within-site spatial heterogeneity in erosion. Measuring wind erosion in unburned and recently burned Chihuahuan desert shrubland, we found (i) weekly wind erosion was related more to daily peak wind velocities than to daily average velocities as consistent with our findings of a threshold wind velocity at approximately 7 m s^{-1}; (ii) greater erodibility in burned vs. unburned shrubland as indicated by erosion thresholds, aerodynamic roughness, and near-ground soil movement; and (iii) burned shrubland lost soil from intercanopy and especially canopy patches in contrast to unburned shrubland, where soil accumulated in canopy patches. Our results are among the first to quantify post-fire wind erosion and highlight the importance of accounting for finer temporal and spatial variation in shrubland wind erosion. This finer-scale variation relates to semiarid land degradation, and is particularly relevant for predictions of contaminant resuspension and redistribution, both of which historically ignore finer-scale temporal and spatial variation in wind erosion.

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Published in J. Environ. Qual. 31:599–612 (2002).

Soil is redistributed and transported by wind erosion in semiarid ecosystems. Rates of wind erosion fundamentally depend on the characteristics of wind in a complex fashion (Bagnold, 1941). For example, wind erosion exhibits a threshold-like response to increasing wind velocities (Bagnold, 1941; Gillette et al., 1980; Nicholson, 1993; Belnap and Gillette, 1998). Similarly, concentrations of airborne dust and soil contaminants are predicted to increase as a power function with velocity (Woodruff and Siddoway, 1965; Anspaugh et al., 1975). Because of these nonlinear relationships, wind erosion may occur primarily during episodic, high-wind events (Helgren and Prospero, 1987; Cahill et al., 1996; Goossens and Offer, 1997; Godon and Todhunter, 1998; Stout, 2001). However, few studies have evaluated how wind erosion over longer time frames (e.g., weeks) relates to net soil redistribution and transport of associated nutrients and contaminants during episodic, high-velocity wind events.

Wind erosion also fundamentally depends on land-surface characteristics of vegetation structure and associated ground cover (Fryrear, 1985). Characteristics of the vegetation matrix that are particularly influential on wind erosion include the amount, type, and spatial pattern of vegetation (Raupach et al., 1993; Wolfe and Nickling, 1996). One of the key characteristics of this matrix is the proportion and spacing of two fundamentally different patch types: the canopy patches associated with woody plants (trees and shrubs) and the intercanopy patches that separate them (Belsky and Canham, 1994; Scholes and Archer, 1997; Breshears and Barnes, 1999). These two patch types differ in a number of

Abbreviations: DOE, United States Department of Energy; TSP, total suspended particulate; WIPP, Waste Isolation Pilot Plant.

microclimatic conditions (McPherson, 1997; Breshears et al., 1997, 1998; Martens et al., 2000), and variation in height and spatial pattern of the two patch types is a major determinant of aerodynamic roughness (Raupach et al., 1993; Wolfe and Nickling, 1996; Dong et al., 2001), an index of erodibility. Because these two patch types are the dominant components of the land surface matrix in semiarid shrublands, the amount of wind-driven redistribution of soil between these two patch types is tightly interrelated with the net loss or gain of soil by wind erosion at a site.

The vegetation matrix and associated ground cover can be changed rapidly at a site in response to disturbance, such as heavy grazing, drought, and human activities (Schlesinger et al., 1990; Bahre, 1995; Allen and Breshears, 1998; Breshears and Allen, 2002). These changes can, in turn, result in increased wind erosion (Zobeck et al., 1989; Okin et al., 2001; Reynolds et al., 2001). Of particular concern are the rapid and large changes in land surface characteristics that can accompany fire (Paysen et al., 2000). Historically, fire occurred frequently in semiarid shrublands and grasslands (McPherson, 1995). The fire probability in shrublands is probably related to the proportion of cover from herbaceous and woody plants. In general, when herbaceous cover is high, fire is likely to spread. When the amount of woody cover is intermediate and herbaceous cover is low, as in many degraded shrublands, fire is unlikely to spread; when the amount of woody cover is very high, fire is likely to spread and to be intense. The role of fire is likely to become more important in the future because fire suppression over the past century has resulted in greatly increased fuel loads as a result of increasing density of woody plants (Grover and Musick, 1990; Archer et al., 1995; Covington et al., 1997; Fulé et al., 1997; Mast et al., 1999; Van Auken, 2000). Further, frequencies of extreme climatic events that make fire more probable (e.g., drought) increased during the past century and are expected to increase further in coming decades (Easterling et al., 2000). Hence, fire is likely to become increasingly important (Swetnam et al., 1999; Swetnam and Betancourt, 1998) and may result in large increases in wind erosion rates in shrublands. Wind erosion in undisturbed and disturbed sites, including a burned site, was studied by Zobeck et al. (1989), but this study was limited to direct measurements of sediment catch and nutrient movement and did not include other more generally predictive metrics for wind erosion. Although this study found large increases in wind erosion at the burned site, there are few, if any, other studies of post-fire wind erosion in any ecosystem, for semiarid shrubland or otherwise.

Quantification of the complex dependencies of wind erosion on the effects of meteorological conditions (e.g., wind characteristics) and land-surface characteristics (e.g., vegetation structure and cover) has generally been approached in one of two ways. The first approach uses wind tunnels and provides a means for quantifying wind erosion relationships under controlled conditions. This approach has led to quantification of threshold velocities under controlled conditions as a function of land surface types (Gillette et al., 1980; Nicholson, 1993; Belnap and Gillette, 1998). However, this approach is limited in its potential to simulate the complexity of natural wind conditions, particularly winds associated with large, episodic wind events that may play a disproportionate role in determining wind erosion (Stout, 1998, 2001). Wind tunnel studies also require isolation of a subsection of the system of interest from its surrounding topography and surface cover, thereby including only a limited amount of spatial heterogeneity in vegetation (Okin and Gillette, 2001).

The second approach, frequently associated with monitoring for air quality, is based on measurements of air concentrations and wind velocities over periods of weeks to months. This approach has the advantage of capturing data on wind erosion in response to a full distribution of naturally occurring wind conditions. However, data obtained using this approach (e.g., aerosol concentrations) and associated meteorological conditions are often aggregated over the longer time scales of weeks to months to achieve required sensitivity (Cember, 1988). This aggregation precludes quantification of relationships of air quality with episodic wind events. Although the monitoring approach can be useful for comparing estimates of wind-driven soil fluxes from different sites, it generally does not assess net changes within a system, that is, whether there is a net loss or gain of soil from the site. Furthermore, neither the monitoring nor the wind-tunnel approach generally evaluates redistribution of soil within the system (e.g., among vegetation patch types) and how this spatial redistribution relates to net system changes in soil. A more comprehensive approach addressing some of the limitations of the two general approaches could yield an improved understanding of wind erosion dynamics.

Improved quantification of wind erosion relationships is needed to better understand the role of wind erosion in semiarid land degradation. Extensive semiarid lands have become degraded over the past century through transformation of grasslands to shrublands, a process often referred to as desertification (Grover and Musick, 1990; Schlesinger et al., 1990; Burgess, 1995; Van Auken, 2000; Okin et al., 2001). These transitions are thought to be interrelated with long-term erosional changes at the scale of canopy and intercanopy patches (Schlesinger et al., 1990; Havstad et al., 2000), whereby soil and nutrients are thought to be redistributed to the canopy patches of the woody shrubs. Long-term historical studies have quantified changes in surface topography in semiarid shrublands (Gibbens et al., 1983; Hennessy et al., 1986; Havstad et al., 2000), but redistribution of soil with respect to canopy and intercanopy patches has not been quantified. An improved understanding of how this redistribution relates to net losses or gains of soil is needed, as well as how such changes relate to disturbance such as fire.

Understanding the relative role of redistribution of soil within a system, and how that redistribution depends on episodic events, is directly relevant to addressing the general issue of contaminant transport in semiarid ecosystems. There are many semiarid lands with low concentrations of contaminants over large areas. For example, within the Department of Energy (DOE) complex in the western United States, there are extensive arid and semiarid lands with low concentrations of radioactively and chemically contaminated soil (e.g., Rocky Flats, Hanford, Idaho National Engineering and Environmental Laboratory, the Nevada Test Site, and Los Alamos National Laboratory) (Riley et al., 1992). There are other sites within the DOE complex for which potential future contamination is of concern. One such site, the Waste Isolation Pilot Plant (WIPP) in southeastern New Mexico, is a repository for transuranic wastes where future drilling for natural resources could result in release of contaminants to surface soils, where they would become subject to wind erosion (Lee, 1997). Other federal agencies (e.g., U.S. Department of Defense) and private landowners (e.g., farms and mining companies) also have tracts of land where redistribution of contaminated soil by wind erosion may be of concern. Wind erosion, in contrast to water erosion or migration through the vadose zone, may be the dominant transport mechanism for contaminated surface soils across semiarid sites within the DOE complex sites, as suggested by model calculations (Johnson et al., 2000). The relative importance of wind erosion can become even greater following disturbance such as fire (Johnson et al., 2000). An improved understanding of the temporal and spatial variability associated with wind erosion prior to and following fire is required for accurate long-term assessment of public risk for nearby residents. Indeed, our ability to predict resuspension of contaminated soils is generally referred to as qualitative and preclusive of quantitative risk assessment (Nicholson, 1994).

Collectively then, few studies have evaluated key temporal and spatial components of wind erosion with respect to (i) relating erosion rates over longer periods of weeks to short, episodic, high-wind gusts; (ii) rates in undisturbed and disturbed sites, particularly unburned and burned sites; and (iii) within-site spatial heterogeneity at the scale of vegetation patches (e.g., canopies of woody plants vs. intercanopy patches between them). Our overall goal was to evaluate temporal and spatial components of variability associated with wind erosion at finer scales than typically studied and to specifically evaluate changes in wind erosion following wildfire. Our objectives were to (i) evaluate longer-term (i.e., weekly) wind erosion rates as a function of average or peak gust velocities, and supplement this work with shorter-term measurements (i.e., minutes) of airborne soil that also differentiate by soil particle size; (ii) evaluate a burned shrubland site and an unburned shrubland site with respect to several metrics related to erodibility, including aerodynamic roughness, erosion thresholds, and soil movement (<1 m height); and (iii) determine differences in net soil loss or gain at the canopy–intercanopy scale within the unburned and burned sites.

Our approach was to measure wind erosion at multiple temporal and spatial scales at unburned and recently burned sites in Chihuahuan Desert shrubland located near the WIPP. This shrubland ecosystem type is of particular interest because it has been the focus of wind erosion studies dating back decades to the 1930s (Gibbens et al., 1983; Hennessy et al., 1983, 1986), it is the basis for much of our knowledge about desertification (Schlesinger et al., 1990), and it is the dominant ecosystem type for the WIPP site and is similar to other shrubland sites within the DOE complex for which wind erosion of contaminated soil is of concern. Our study was designed to not only contribute to site-specific monitoring at WIPP, but also to provide more general insight into wind erosion processes. Our results indicate the significance of episodic winds and disturbance in increasing erosion, show differences at the canopy–intercanopy patch scale, and highlight the importance of accounting for finer temporal and spatial variation in wind erosion of soils for predictions related to land degradation and contaminant transport.

MATERIALS AND METHODS

Our primary study site was an unburned shrubland site that corresponded to an air quality monitoring site operated by the Carlsbad Environmental Monitoring and Research Center, operated by New Mexico State University. This site met the electrical power requirements for operation of our total suspended particulate air samplers over the study period. Our secondary site was a burned site where there was an intense wildfire that occurred about a week before the initiation of our measurements and resulted in loss of nearly all ground cover within 6 km^2. We measured several site characteristics and components of wind erosion at both the unburned and recently burned shrubland sites. Our measurements included friction velocities, aerodynamic roughness lengths, aerosol measurements (by mass and number concentrations, and by particle size), micrometeorological conditions, and changes in surface microtopography. The time scale of these measurements ranged from static to minutes to months.

Site Characteristics

Both the unburned and the burned sampling sites were on U.S. Bureau of Land Management lands and within 20 km to the east of the WIPP boundary. The WIPP is located in the southeast corner of the state of New Mexico approximately 42 km east of Carlsbad at an elevation of 1040 m. The sampling sites were representative of similar areas located in and around the WIPP site (Carlsbad Environmental Monitoring and Research Center, 1998).

Precipitation amounts in Carlsbad are approximately 300 mm annually, mainly from intense spring and summer thunderstorms (United States Department of Energy, 1980). Winds are predominately from the southeast. Over the course of the study, June 1998 through February 1999, the unburned site had a total rainfall of 124 mm and a mean daily wind velocity of 2.4 m s^{-1}, with monthly peak gusts ranging from 12.5 to 22.4 m s^{-1}. Similarly, the burned site had a total rainfall of 118 mm and a mean daily wind velocity of 2.9 m s^{-1}, with monthly peak gusts ranging from 15.2 to 22.4 m s^{-1}.

The two sites were also similar with respect to some key surface characteristics, although there were some differences associated with vegetation. Soil formed in dunes around shrubs at both sites. The soil texture for the top 3 cm of the profile, that portion most subject to wind resuspension, was similar between the unburned site (92% sand, 4% silt, 4% clay) and the burned site (93% sand, 2% silt, 5% clay). Further, the proportion of soil particles in the top 3 cm with sizes smaller than 10 μm in aerodynamic diameter, the proportion considered respirable, was measured using a pipette method for particle size analysis (Black et al., 1965) and was approximately 5% at both sites. With regards to vegetation, both sites are located in the Chihuahuan Desert, which generally include mesquite (*Prosopis glandulosa* Torr.), creosotebush [*Larrea tridentata* (Sesse & Moc. ex DC.) Coville], shinnery oak (*Quercus havardii* Rydb.), sand sage (*Artemisia filifolia* Torr.), various yucca species (*Yucca* spp.), smallhead snakeweed [*Gutierrezia microcephala* (DC.) A. Gray], and threeawn grasses (*Aristida* spp.) along with other various forbs and grasses (Dick-Peddie, 1993). We characterized vegetation at both study sites using a line transect to estimate overall ground cover, which included persistent litter, nonpersistent litter, vegetation ground cover, and rocks. Overall ground cover was similar at the two sites, 66% for the unburned site and 64% for the burned site, although more of the litter at the unburned site was nonpersistent. Percent canopy cover (as viewed from above) from shrubs was comparable between the sites, with a 28% cover for the unburned site and 18% cover for the burned site (based on measurements of the defoliated canopy following the fire). However, at the unburned site there was a higher shrub density (0.20 m^{-2} for the unburned site compared with 0.01 m^{-2} for the burned site). The shrubs at the unburned site were smaller, with an average shrub height of 0.73 ± 0.21 m (standard deviation) compared with 1.3 ± 0.19 m for the burned site, and an average shrub diameter of 1.2 ± 0.6 m for the unburned site compared with and 3.8 ± 1.2 m for the burned site. The dominant shrub at the unburned site was creosote, whereas mesquite was dominant at the burned site. Due to the differences in shrub patch structure, lateral cover or roughness density—a metric defining the density of the frontal silhouette area (Musick and Gillette, 1990)—was calculated for both sites (0.18 for the unburned site and 0.07 for the burned site). Although lateral cover differed somewhat between the two sites, calculations and literature data indicate that the differences we observed in erosion metrics between the sites were primarily the result of the effects of fire rather than differences between vegetation alone.

Aerosol and Meteorological Conditions at Varying Time Scales as Related to Vertical Soil Flux

Vertical Soil Flux Measurements

We measured vertical soil flux (F) using the gradient method (Stull, 1988), for which it is determined as the product of the eddy diffusivity coefficient (K_z) and the mass concentration (χ) gradient with height (z):

$$F = K_z \times \frac{d\chi}{dz} \quad [1]$$

The eddy diffusivity coefficient itself is a function of the friction velocity (u_*):

$$K_z = k_v z u_* \quad [2]$$

where k_v = von Karman dimensionless constant (approximately 0.4) and z = height of the measurement. The friction velocity, a measure of the boundary shear created as winds pass over vegetation and soils, can be estimated for a given terrain and wind velocity by measuring the wind velocity profile with height (Bagnold, 1941), or by using high-frequency, three-dimensional measurements of wind velocities (Stull, 1988):

$$u_*^2 = \left[\overline{u'w'}^2 + \overline{v'w'}^2\right]^{1/2} \quad [3]$$

where u', v', and w' are the instantaneous wind velocity components in both horizontal (x and y) and vertical (z) directions. We used the latter approach and measured the instantaneous wind velocity components using a factory-calibrated sonic anemometer (Model CSAT3; Campbell Scientific, Logan, UT) with sampling frequency set to 10 Hz.

Aerosol Measurements

We measured aerosol concentrations using a number of instruments over the study period. Our approach included sampling periods that ranged from a minute to several months, depending on the instrument, so that we could determine long-term averages as well as short-term fluctuations in wind erosion.

To address our first objective (evaluation of the short- and long-term relations between wind velocities and erosion), we collected total suspended particulate (TSP) samples weekly (long term) at two heights (1 and 3 m) allowing for determination of the concentration gradient with height ($d\chi/dz$). Here χ is the mass concentration measured during the week and z is the sampling height. We collected TSP air samples with a sampling rate of about 6.8 m^3 h^{-1} using a sampling inlet (Fig. 1a) based on the PM-10 design of Liu and Pui (1981). The inlet is not directionally dependent, and it provides accurate sampling for predominant airborne particle sizes (<10 μm in diameter) measured at our study sites and for intermediate wind velocities. Aspiration efficiencies for particles of aerodynamic diameters of 8.5 and 11 μm were 100 ± 10% at wind speeds up to 2.8 m s^{-1} (Liu and Pui, 1981). We modified this inlet to collect all airborne particulate rather than just particles less than 10 μm. These modifications included placing the filter close to the bottom plate and adding a coarse wire screen to keep insects and larger debris out of the filter. To supplement the Liu and Pui (1981) study, tests of the modified sampling inlet at high-wind velocities of 12, 15, and 17 m s^{-1} and for large particle sizes (5, 10, and 30 μm) indicated that collection efficiencies were approximately 120% (an oversampling of 20%) for 5-μm particles and approximately 50% for both 10- and 30-μm particles (Rodgers et al., 2000). The results of this study suggested that collection efficiencies at these high-wind velocities, although affected by particle size, were not affected by wind velocity in the range tested. A correction for sample efficiency was precluded because we could not measure particle size in real time during these measurement intervals.

We measured aerosol concentrations over 1-min time intervals (short-term) using two types of optical laser particle counters (LPCs) that measure number of particles as a function of particle size. Particles sizes > 0.5 and > 5.0 μm were measured at 1- and 3-m heights using the smaller LPCs (Model 7550; Particle Measuring Systems, Boulder, CO). Particle concentrations of 0.3, 0.5, 1, 5, and 10 μm were measured at a height of 3 m using the Met One optical counter (Model A2408; Met One, Grants Pass, OR). Sampling occurred during the daylight hours of 23 July 1998, 1 and 2 Sept. 1998, and 27 and 28 Oct. 1998.

Micrometeorological Measurements

At each sampling site, a 3-m-high Weather Monitor II meteorological station from Davis Instruments (Hayward, CA) was used to measure local meteorological conditions at a sampling frequency of 2 h. Horizontal wind conditions were measured using a cup anemometer for velocity (average and peak gust) and a wind vane for direction. For each sampling period (generally 7 d), two summary statistics were calculated for the horizontal wind velocities. One was the mean of the daily averaged wind velocities, and the second was the mean of the daily peak or maximum gust velocity. Temperature, relative humidity, and rainfall were also measured, and the means of the daily averages were calculated for each weekly sampling period. The sampling frequency was typically 2 h, but was adjusted to 60 s when the optical particle counters were used.

Indices of Erosion Potential at Unburned and Burned Sites

Erosion Thresholds, Friction Velocity, and Roughness Lengths

For the unburned and burned sites, we estimated erosion thresholds, friction velocities, and roughness lengths as partial indices of site erosion potential, all of which relate surface characteristics to airflow at the atmospheric boundary layer. Erosion thresholds were measured using 1-min LPC measurements of aerosol concentration evaluated with horizontal wind velocity and friction velocity. As noted above, friction velocities were estimated from sonic anemometer measurements using Eq. [3]. Aerodynamic roughness length, the height above the ground where the average wind velocity is equal to zero, was estimated for both sites by:

$$\ln z_0 = \ln z - \frac{\overline{U}k_v}{u_*} \quad [4]$$

where z_o is the aerodynamic roughness length, z is the measurement height (3 m), $\overline{U}$ is the average horizontal wind velocity during the measurement interval, k_v is the von Karman constant (0.4), and u_* is the friction velocity. For locations where the canopy cover is very dense and uniform, a displacement height should be considered in Eq. [4] (Stull, 1988). However, the vegetation in desert shrubland is generally sparse. Specifically, our measurements of L_c and area coverage at our two sites are within the range measured by Wolfe and Nickling (1996) in similarly vegetated desert shrubland sites. Although we did not make measurements needed to determine displacement height, Wolfe and Nickling (1996) found that the displacement height was insignificant in all their study sites. Therefore, we did not include a correction for displacement height.

While measurements of $\overline{U}$, u', v', and w' were made at a frequency of 10 Hz, they were averaged over a 30-min time period. To estimate friction velocities and roughness lengths, we used a subset of data that would more closely approximate the ideal conditions of neutral atmospheric conditions (Class D in the Pasquill stability class system). Measurements were mostly made at night in the late fall when the sun was low in the horizon, and we analyzed only those values where the horizontal wind velocity was greater than 2 m s^{-1} (Zannetti, 1989). To better determine the Pasquill stability class for our measurement periods, we used fine wire thermometers to measure temperature gradients. Mean temperature gradients were -0.03°C m^{-1} for the unburned site and 0.12°C m^{-1} for the burned site, suggesting that average atmospheric conditions were not always neutral and were mostly in the slightly stable category (Class E).

Soil Movement Measurements

We used passive soil collectors, which are self-orienting in the strong winds associated with wind erosion events, to collect windblown soil at six heights less than 1 m above the surface (Fig. 1b). At both sites, three of these samplers were placed in a triangle with samplers 20 m apart. Samples from the passive samplers were generally collected weekly and the dry mass in each of the boxes was measured. Although the sampling efficiency as a function of wind velocity and particle size has not been characterized like other passive samplers (Fryrear, 1986), the samplers provide a quantitative means for determining relative differences in wind-driven soil movement between the two sites.

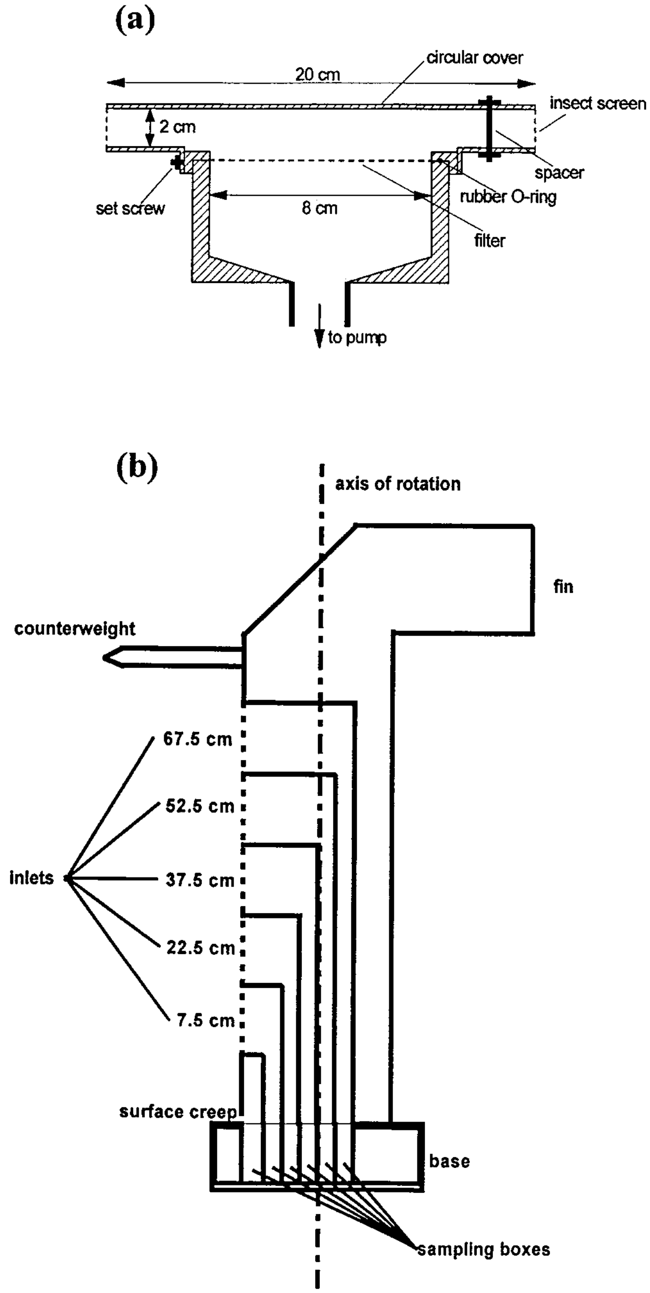

Fig. 1. Schematics of omni-directional sampler used for total particulate aerosol measurements (*a*) and the passive soil collector (*b*).

Spatial Heterogeneity in Erosion between Canopy and Intercanopy Patches

We measured changes in surface microtopography at the scale of canopy and intercanopy patches for both sites using erosion bridges (Wilcox et al., 1996). An erosion bridge is installed by placing two pieces of rebar into the ground such that an approximately 1.9-m level gauge across them is level. The distance from the bottom of the level to the top of the land surface was then measured at 5-cm intervals. We installed the rebar completely through the soil profile and several centimeters into the underlying caliche to ensure stability. At both sites, we installed six erosion bridges across canopy patches and six in intercanopy patches. Measurements were obtained on 2 Dec. 1998 and 13 May 1999, an interval of 162 d.

Statistical Analysis

Least squares multiple regression analysis was used to assess relationships of TSP concentrations and soil collection rates from the passive samplers with meteorological measurements. Analysis of the relationship of TSP concentration and peak wind was done using linear and nonlinear (power function) models. Additionally, statistical comparisons between sites were made of aerodynamic roughness lengths, soil collection rates, and changes in surface microtopography using the nonparametric Mann–Whitney *U* test (StatSoft, 1994). Least-squares regression analysis of friction velocities as related to horizontal wind velocity was conducted and the slopes and intercepts for each site compared using the Student's *t* test. A probability level of ≤5% was considered significant.

RESULTS

Aerosol Concentration and Vertical Flux as Related to Temporal Variations in Wind Velocities

To address our first objective, we estimated aerosol concentrations and vertical soil flux for the unburned site. Measured TSP mass concentrations ranged from approximately 13 $\mu g\ m^{-3}$ to approximately 40 $\mu g\ m^{-3}$, and are reasonable concentrations for rural air (Hinds, 1982). Fluctuations in the weekly concentrations were similar for samplers at the 1- and 3-m heights. Mass concentrations decreased with height, with mass concentrations at 3 m (median of 24.7 $\mu g\ m^{-3}$) significantly less than those at 1 m (median of 26.1 $\mu g\ m^{-3}$) based on paired comparison using the sign test (Huntsberger and Billingsley, 1981). The resulting median gradient was 0.44 $\mu g\ m^{-4}$, with a quartile range (25th through 75th percentile) of 0.75 $\mu g\ m^{-4}$. Multiple regression analysis showed that the only two meteorological measures correlated to the weekly aerosol mass concentration were the mean of daily averaged wind velocity and the mean of the daily peak gust velocities ($p < 0.01$). These relationships could be described by a linear model (Fig. 2a,b). The r^2 values for the linear correlation of TSP with the mean peak wind velocity were 0.48 for the 3-m height and 0.52 for the 1-m height. In comparison, the r^2 values for the linear correlation of TSP with the average wind velocity were 0.27 and 0.25 for the same sampling heights (Fig. 2a). On the basis of other studies (Linsley, 1978; Sehmel, 1980), we determined a best-fit power function for the relationship for concentrations at both 1 and 3 m with average weekly velocities. However, the nonlinear models did not improve the fit

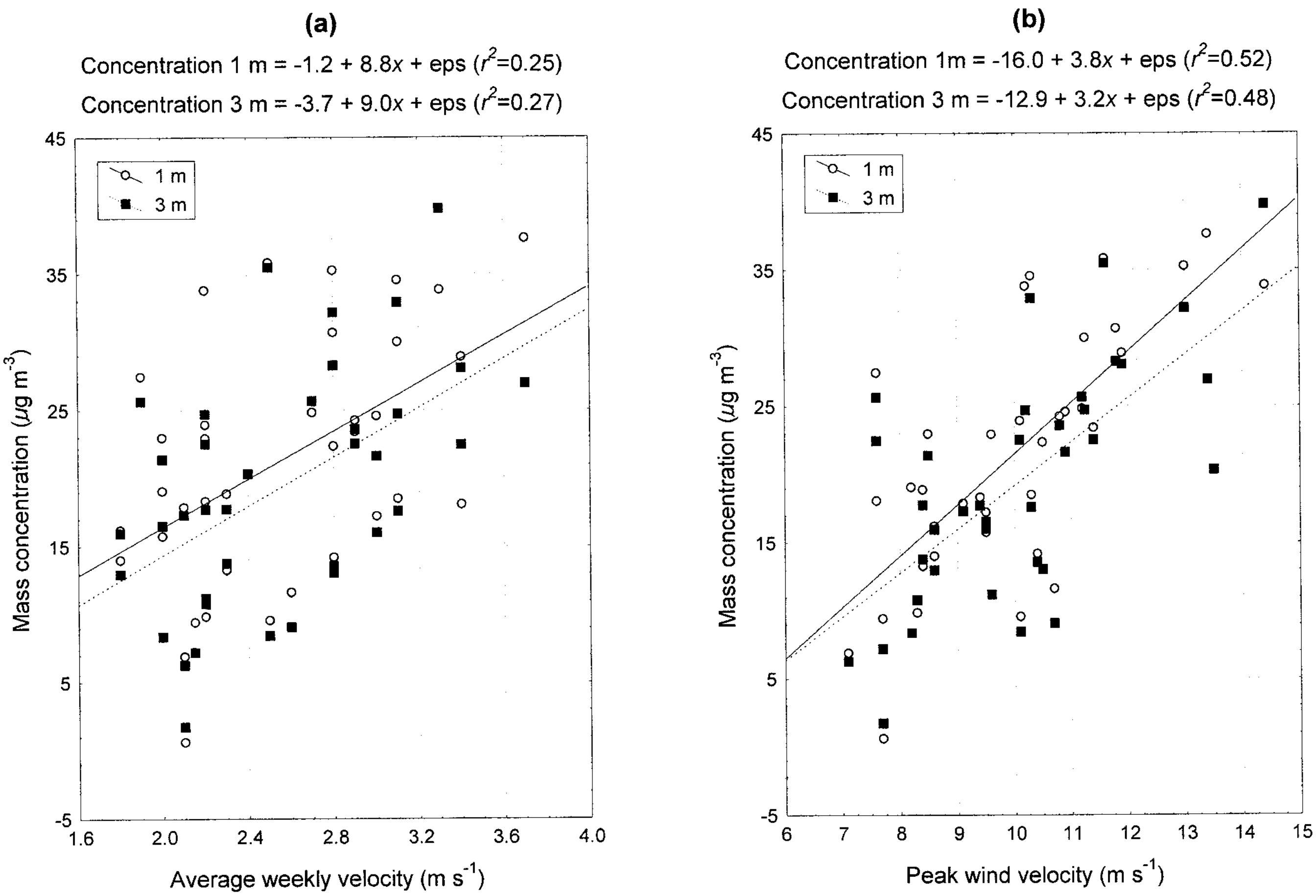

Fig. 2. Mass concentrations from the unburned site measured at 1 and 3 m as a function of average weekly velocity (*a*) and the peak wind velocity (*b*).

(r^2 values of 0.25 for 1 m and 0.27 for 3 m). An analysis of the linear model showed that the residuals were evenly distributed about the regression line at all peak wind velocities, indicating that there is not prediction bias over the velocities encountered. The vertical soil flux, F (μg s^{-1} m^{-2}), at the unburned site could therefore be estimated using the model as given by combining Eq. [1] and Eq. [2]:

$$F = 0.4 \times 3 \times (0.004 + 0.113 \times u_p) \times \left(\frac{\chi_1 - \chi_3}{2}\right) \quad [5]$$

In Eq. [5], 0.4 is the von Karman constant, 3 is the measurement height for wind velocity in meters, the linear equation with u_p is the function describing the relationship between the mean of the daily peak wind velocity (u_p) and the friction velocity at the unburned site (introduced later in Eq. [7]), and the concentration gradient is the difference of mass concentration of weekly measurements made at 1 and 3 m (χ_1 minus χ_3) divided by the difference in sampling heights (2 m). Equation [5] yields estimates of upward vertical soil flux at the unburned site of approximately 0.6 μg s^{-1} m^{-2} for median daily peak gusts of 10 m s^{-1} and median concentration gradient of 0.44 μg m^{-4}. The concentration gradient was found to be independent of all meteorological conditions and mass concentrations. The distribution of weekly measurements of vertical soil fluxes shows periods of accumulation (downward or negative soil fluxes) and periods of loss (positive upward soil fluxes), with a general trend of soil loss over the study period (Fig. 3).

Measurements more finely resolved in time also provided insight to wind erosion processes. Particle number concentrations measured by the optical counters showed considerable minute-to-minute variability over the 1-min measurement intervals. This variation was especially pronounced for larger particles. For example, 10-μm particles measured at a height of 3 m show a nonlinear increase in particle concentration with increases in horizontal wind velocity (Fig. 4). These data were obtained during the approach of a thunderstorm, with wind velocities increasing from approximately 4 m s^{-1} to more than 7 m s^{-1} in a minute. There was an increase in number concentration as the wind velocity exceeded a threshold of about 7 m s^{-1} for all particle sizes, with the concentration increase being especially dramatic for the larger particle sizes. For example, the concentration of 10-μm particles increased by more than a factor of 20, whereas the particle concentration of 0.3-μm particles only increased by a factor of 1.3. Interestingly, the first two concentration measurements following the rapid increase in wind velocity were not substantially greater than the typical levels measured prior to the increased wind velocity, suggesting a time lag in the aerosol concentration following wind gusts.

Aerodynamic and Erosion Comparisons between Burned and Unburned Sites

Friction Velocities

Friction velocities at the burned site were significantly less than at the unburned site (Fig. 5). The least square

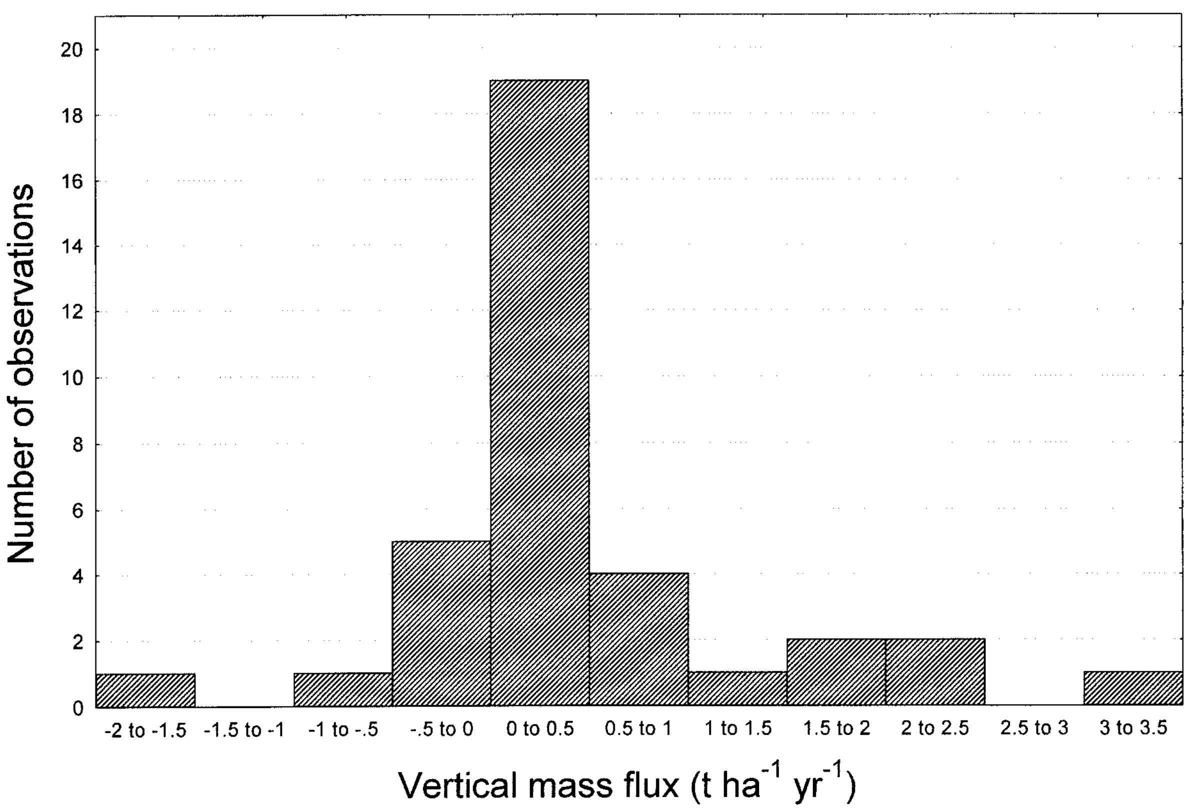

Fig. 3. Distribution of vertical soil flux at the unburned site.

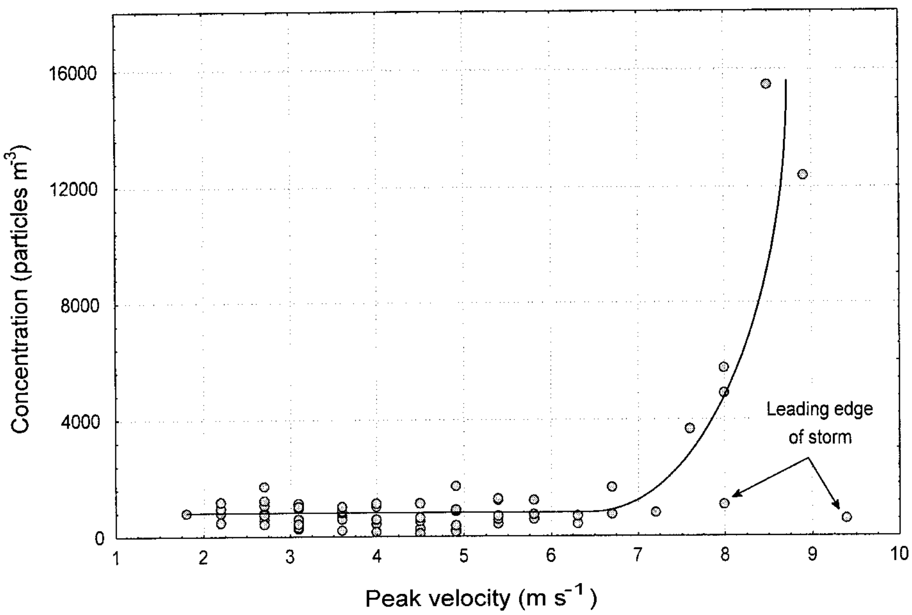

Fig. 4. One-minute concentrations of 10-μm particles as a function of horizontal wind velocity. Line is hand drawn and is for illustrative purposes only.

regressions relating friction velocity to horizontal wind velocity were:

$$\text{Burn Site: } u_*(\text{m s}^{-1}) = -0.07 + 0.99 \times u_p \quad (r^2 = 0.90) \qquad [6]$$

$$\text{Unburned Site: } u_*(\text{m s}^{-1}) = -0.004 + 0.11 \times u_p \quad (r^2 = 0.90) \qquad [7]$$

where u_p is the mean of the daily peak wind velocity in m s^{-1}. The residuals were scattered evenly about the regression line, suggesting that the linear relationship model is appropriate for these two data sets. The slopes and the intercepts of the two equations were significantly different (Student's *t* test) and the unburned site had greater intercept and slope. Using a threshold wind velocity of 7 m s^{-1} (Fig. 4), Eq. [7] gives a threshold friction velocity of around 0.80 m s^{-1} for the unburned site.

Aerodynamic Roughness Lengths

Distributions of aerodynamic roughness lengths appear skewed, with median lengths of 0.02 m (quartile range 0.02 m) for the burned site and 0.1 m (quartile range 0.03 m) for the unburned site (Fig. 6). The roughness lengths at the unburned site were significantly greater than those at the burned site (Mann–Whitney *U* test).

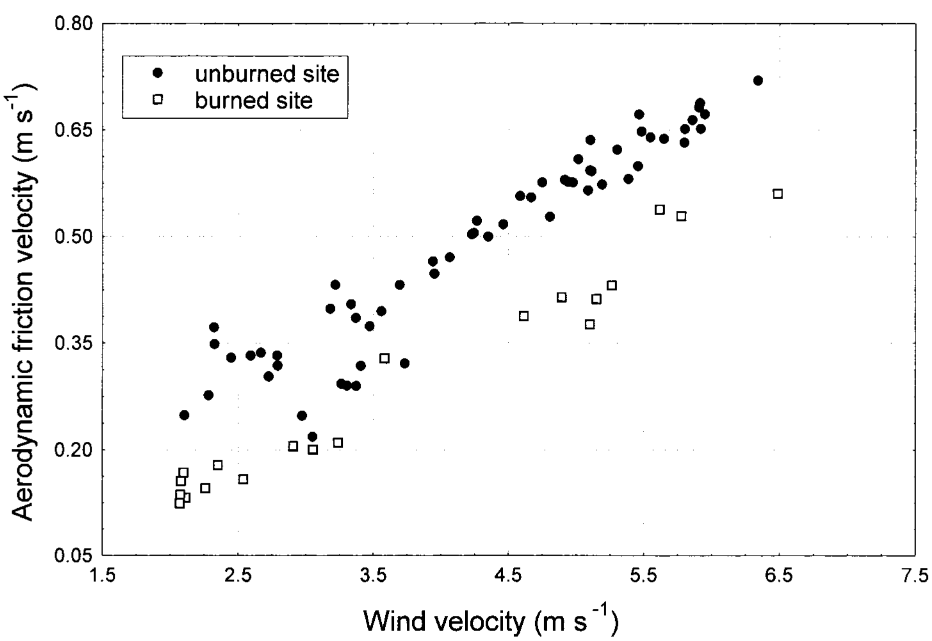

Fig. 5. Aerodynamic friction velocities measured at the burned and unburned sites as a function of the horizontal wind velocity.

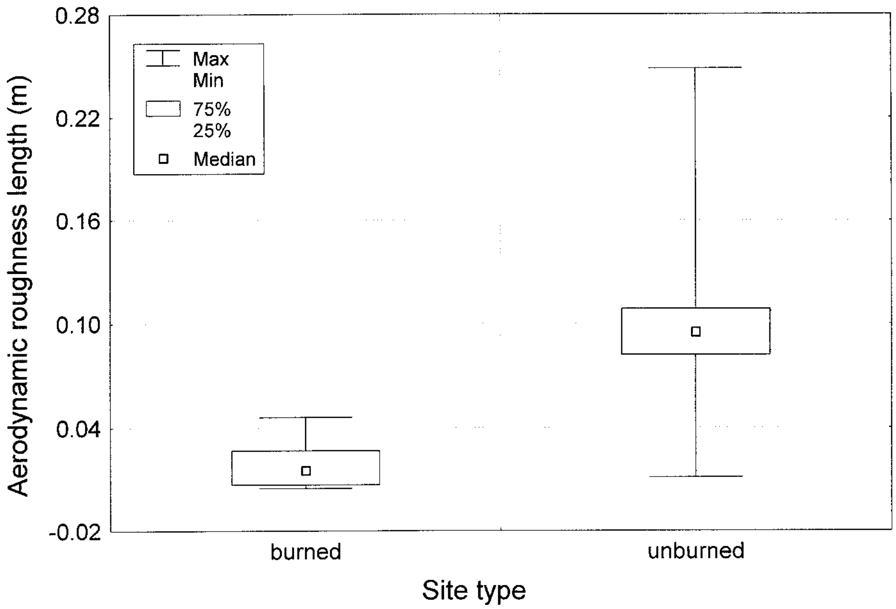

Fig. 6. Aerodynamic roughness length distributions measured at the burned and unburned sites.

The roughness lengths for the unburned site, which had a less-homogeneous surface and more variation in wind direction during the measurements (270° verses 180°), were more variable than those at the burned site.

Soil Collection Rates

There were significant differences in soil collection rates between the burned and unburned sites, based on measurements with the passive soil collectors. Soil collection rates at each site were an exponential function of sampling height (Fig. 7). Variation in collection rates, summed over all heights, was large, particularly for the burned site: average collection rates ranged from 0.2 to 14.7 g d^{-1} with a median rate of 0.3 g d^{-1}, whereas collection rates at the unburned site ranged from 0.01 to 1.8 g d^{-1} with a median rate of 0.1 g d^{-1}. Collection rates at the burned site were significantly greater than those at the unburned site, particularly during the summer months immediately following the fire, when the mean collection rate at the burn site was 70 times that at the unburned site. The greatest soil collection rates at the burned site were measured during sampling periods with high winds. Soil collection rates were significantly related to the mean of the daily peak gust velocity at the burned site (r^2 = 0.36), but not at the unburned site (r^2 = 0.01).

Spatial Variations in Microtopography at the Burned and Unburned Sites

Erosion bridge measurements of local microtopography showed significant differences between the burned and unburned sites (Fig. 8). There was significantly greater soil loss at the burned site compared with the unburned site (Mann–Whitney *U* test). The mean

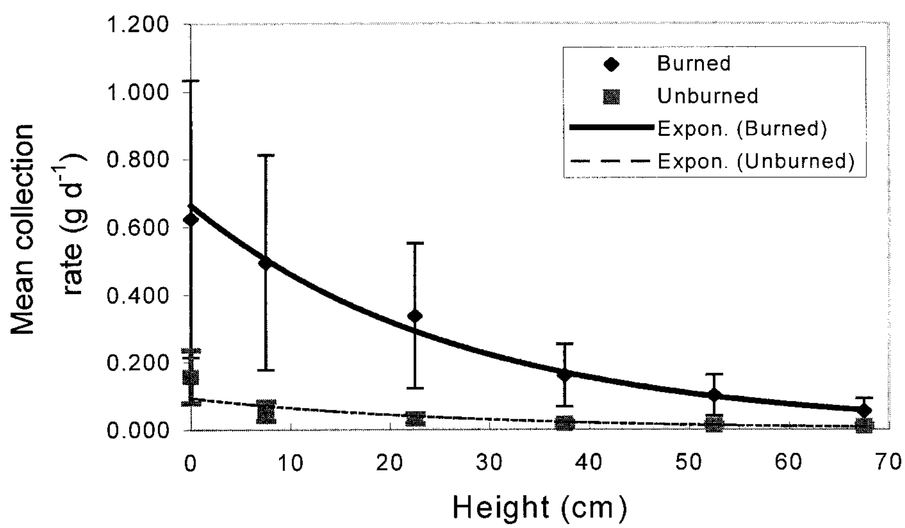

Fig. 7. Mean collection rates for different sampling heights at the burned and unburned sites with exponential fits. Error bars represent one standard deviation.

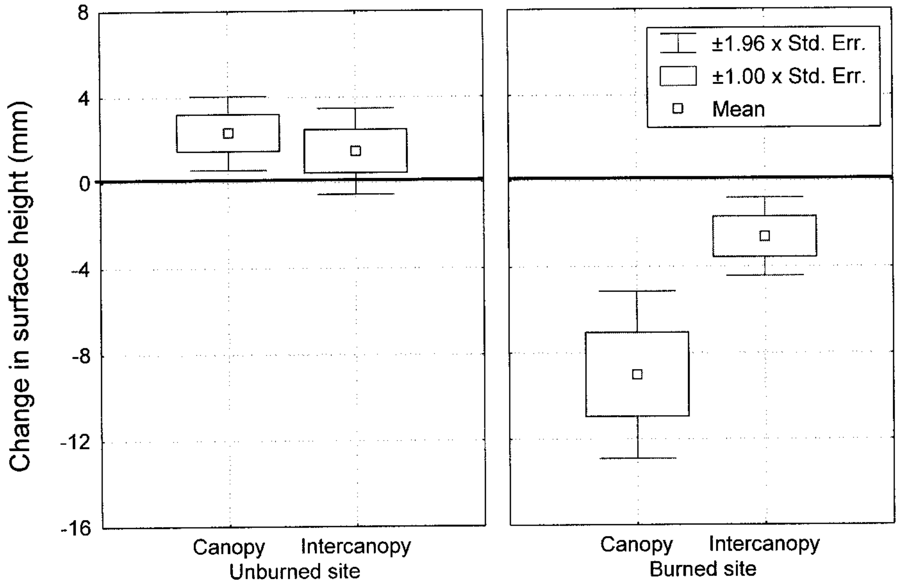

Fig. 8. Changes in surface height categorized by site and intercanopy verses canopy.

change in surface topography at the burned site was −5.8 mm (standard deviation of 18.0 mm) while the mean change at the unburned site was 1.9 mm (standard deviation of 11.0 mm). We also found a significantly greater loss of soil in the canopy compared with the intercanopy patches at the burned site, in contrast to the unburned sites, where we did not find significant differences between patch types. Changes in soil topography were normally distributed, except at the burn site in the canopy, which showed a slight bimodal distribution with a small grouping of measurements (14 out of 132) where there was significant loss of surface soil (≥50 mm).

DISCUSSION

The Episodic Nature of Wind Erosion

Our results highlight episodic dynamics in wind erosion and indicate that improved prediction of wind erosion requires consideration of short-term, high-velocity wind events. We were able to predict weekly wind erosion rates better using means of daily peak gust velocities instead of using means of the daily average wind velocities (Fig. 2). At a finer time scale of minutes, we observed large increases in wind erosion once a threshold wind velocity of about 7 m s^{-1} was exceeded. Our results also show how the effect of exceeding the threshold wind velocity can differentially resuspend soil particles of different sizes, as predicted from theory (Slinn, 1974). Although we have only limited data for wind erosion over relatively short time frames, these field measurements of particle number concentration for multiple-sized aerosol particles over time and as a function of wind velocity are rare. Additional field studies are needed to quantify more systematically the relationships between airborne soil particles of different sizes and meteorological conditions. Our study is consistent with other field studies finding a threshold wind velocity at which wind erosion greatly increases (Helgren and Prospero, 1987; Godon and Todhunter, 1998; Stout, 2001), and it builds on them by highlighting how very short-term, high-velocity wind events may translate into prediction of wind erosion rates on a longer-term, weekly basis. In addition, our study indicates how these relationships depend on particle size. Although the importance of episodic wind events in estimating wind erosion certainly has been recognized previously, this knowledge is not always factored into improving longer-term predictions of wind erosion.

Effect of Fire on Wind Erosion

Comparisons of erosion thresholds, roughness lengths, and soil collection rates all showed significant differences between sites and indicated that the burned site is much more erodible than the unburned site. The large differences in these metrics are not readily attributable to the site differences in shrub size and density, but rather appear to be largely attributable to the effects of burning. For instance, our estimates of lateral cover were 0.18 for the unburned site and 0.07 for the burned site with foliage. Both these values correspond with high threshold friction velocities relative to that for bare soil (Raupach et al., 1993). In contrast, the threshold friction velocity at the burned site, with no foliage present and hence little lateral cover, should approach that of bare soil. Our estimate of the threshold friction velocity at the unburned site was about 0.8 m s^{-1}, which is significantly higher than that found by Gillette et al. (1980) for bare sand dune soils, which were similar in soil texture and ground cover to that at our burned site (range from 0.25 to 0.59 m s^{-1}) or for highly disturbed landscapes (Gillette and Chen, 2001). Further, the friction velocities and roughness lengths at the unburned site compare well with other similarly vegetated sites, whereas the same metrics at the burned site compare favorably with values reported for highly disturbed sites with little or no vegetation (Wolfe and Nickling, 1996). Finally, we found a threshold wind velocity of about 7 m s^{-1} at the vegetated site, whereas Stout (2001) found a much lower threshold wind velocity for dry bare soil of 4 m s^{-1} in the neighboring state of Texas. Hence, the differences that we observed in erosion thresholds, roughness lengths, and friction velocities are probably due primarily to effects of the fire rather than simply differences in vegetation cover.

In addition to differences in erosion thresholds, roughness lengths, and friction velocities, we found significantly greater soil collection rates for the burned site. Soil movement was greater at the burned site by a factor of 3 over the whole study period and a factor of 70 times shortly following the fire during periods of high-velocity winds. These results highlight the interaction between episodic high winds and changes in vegetation cover, and concur with the results from Zobeck et al. (1989) showing that a burned area had significantly higher rates of wind erosion relative to other less-disturbed areas. Even though there were some differences in vegetation cover between our two sites, they were generally more similar than those in the Zobeck et al. (1989) study, and thus provide a more direct comparison of wind erosion in burned and unburned shrubland.

In summary, our results indicate that all metrics of erodibility were much greater at the burned site in comparison with the unburned site. These differences are unlikely due primarily to the site differences in shrub density, but rather are probably related to large increases in soil erodibility following fire. More generally, our study highlights how disturbance can dramatically increase wind erosion.

Spatial Heterogeneity in Wind Erosion

Our erosion bridge measurements build on our findings of differences in metrics of wind erosion between the burned and the unburned site. Although we measured substantial near-surface soil movement and upward vertical flux at the unburned site, these measurements do not provide direct evidence of a net loss of soil. Therefore, erosion bridge measurements were made to provide a better estimate of net changes in soil. The results of the erosion bridge measurements suggest a net loss of soil from the burned site (Fig. 8). In contrast, the unburned site did not lose soil, but rather had a slight increase of soil in the canopy. Water erosion can contribute to the measured net soil movement, but it was not likely significant relative to wind erosion at our study. Rainfall simulation studies at a nearby study site indicate that annual water erosion rates may be orders of magnitude less than annual wind erosion rates (Johansen et al., 2001).

The soil erosion bridge measurements highlight the importance of evaluating differences between canopy and intercanopy patches. At the burned site, the topographically higher canopy patches (raised sand dunes after the fire) lost significantly more soil than the intercanopy patches (Fig. 8). In contrast, at the unburned site there was not a significant difference between patch types. While our study quantifies some important differences between canopy and intercanopy patches, it is insufficient to fully address hypotheses about the importance of redistribution of soil between patch types relative to net system soil loss or gain. In a long-term study of wind erosion in the Chihuahuan Desert from 1933 to 1978, there was large soil loss from a grassland–shrubland ecotone, smaller loss from a site with partial shrubland encroachment, and accumulation in a shrubland with large dunes (Gibbens et al., 1983). Size sorting of soil was also evident in areas of high wind erosion (Hennessy, 1986). Vegetation changes occurred during the study period, with the site that was originally a grassland–shrubland ecotone converting to shrubland over the course of the study (Hennessy et al., 1983). Nutrients were concurrently redistributed to canopy patches, apparently with little net loss of nutrients from the site (Schlesinger et al., 1996; Havstad et al., 2000). Our findings are consistent with hypotheses of soil accumulation in shrublands, although they do not allow us to test specifically if that process is related to redistribution of soil from intercanopy to canopy patches, as has been hypothesized.

The relationship between local redistribution and net system loss or gain has largely been unaddressed in wind erosion studies. However, recent water erosion studies have documented such relationships. At a semiarid piñon (*Pinus edulis* Engelm.)–juniper [*Juniperus monosperma* (Engelm.) Sarg.] woodland site with relatively high percent ground cover, there is an enormous amount of redistribution of water and eroded sediment between patch types within the site with very little loss from the site as a whole (Reid et al., 1999; Wilcox et al., 1996). Hence, understanding small-scale generation of runoff and water erosion alone is insufficient to understand the larger-scale processes that lead to losses from the system (Davenport et al., 1998; Ludwig et al., 1997). These advances in understanding water erosion are relevant for wind erosion because each of the wind erosion–related measurements, including vertical flux, aerosol concentrations, near-ground soil movement–saltation, and changes in microtopography, only provides partial information about the redistribution and net change of soil for a site.

Implications of Finer Temporal- and Spatial-Scale Variability in Wind Erosion

Our results indicate the importance of post-fire increases in wind erosion. More generally, they highlight the importance of considering finer temporal- and spatial-scale variability in wind erosion measurement and estimation. Improved predictability can result from factoring in metrics that characterize the short-term episodic nature of wind events and changes in vegetation cover and patch pattern following disturbance. An improved understanding of these processes, particularly as related to the formation of canopy dunes and the redistribution and loss of nutrients, is needed to better address degradation of semiarid grasslands and shrublands and associated desertification processes. An improved understanding of these processes, in turn, is needed to address the more specific problem of contaminants in semiarid ecosystems.

A more quantitative understanding of wind-driven contaminant transport is needed to improve monitoring methods, to better predict contaminant transport, and to improve risk assessments. Our results suggest that short-term, high-velocity wind events should be accounted for in predicting contaminant transport (e.g., airborne radionulcides), whereas current approaches are based on longer-term averages (Linsley, 1978; Garger et al., 1999). Existing empirically based models, generally calibrated with site specific conditions, are thought to be accurate to within an order of magnitude (Garger et al., 1999). These models do not account for short-term resuspension associated with high-velocity wind events that translate into longer-term transport rates as suggested by our results. Over even longer time frames of years to decades, less frequent but higher-intensity wind events such as tornadoes or dust devils become more likely and should be considered. Tornadoes are relatively frequent in this portion of Chihuahuan shrubland, with more than 15 reported tornadoes within a 1° longitude and latitude of the WIPP area in a 12-yr period (United States Department of Energy, 1997).

The particle-size dependencies on wind erosion relationships are of particular concern for contaminant monitoring and risk assessment. Our field measurements confirm the predictions of theoretical and wind tunnel studies that resuspension depends on particle size, such that larger particles are more resuspendable than smaller particles up to a certain size. Therefore, it is important to know the particle size of contaminants and the size of soil particles to which the contaminants are attached. Particle size influences not only wind-driven transport, but also the resipirable fraction that drives risks of inhaled contaminants, with the inhalation risk for a particle with a 2-μm aerodynamic diameter (AD) being much greater than that posed by a particle greater than 10-μm AD (International Commission on Radiological Protection and Measurements, 1997). Particle sizes of contaminants, host soil particles, and the sizes of the airborne particles have been studied for plutonium in soils in one DOE site (Volchok et al., 1972; Sehmel, 1976, 1978; Tamura, 1976). These particle-size dependencies, then, need to be integrated into improved assessment of risks that account for the effects of episodic wind events.

Longer-term risk assessments also need to consider the potential for increased contaminant mobility following disturbances such as fire. For instance, during the spring and summer of 2000, major fires spread over DOE facilities in Los Alamos, Hanford, and Idaho, and the probability of these disturbance events increases with time. Even so, risk assessments that must span years, decades, or longer do not generally consider these processes. Our study suggests that fire can increase resuspension rates by more than an order of magnitude following fire. Similarly, Kashparov et al. (2000) found that during resuspension rates during fire in burning forests near Chernobyl increased by several orders of magnitude. The probability of disturbance events such as fire, the recovery times following disturbance, and the effects of those events on contaminant transport need to be accounted for in future assessments.

The spatial differences in erosional loss of soil are important considerations for risk assessment as well. We have little knowledge about the relative importance of redistribution of contaminants within an area vs. off-site transport of those contaminants. Wind erosion models do not account for spatial heterogeneity (VandenBygaart et al., 1999), such as that at the scale of canopy and intercanopy patches. Contaminants, such as ^{137}Cs, may be spatially heterogeneous with greater concentrations under canopy patches (Coppinger et al., 1991), which could then be more vulnerable to post-fire mobility (Fig. 8).

In conclusion, our study of wind erosion at unburned and recently burned sites in Chihuahuan Desert shrubland yielded findings that (i) weekly wind erosion was related more closely to the mean of daily peak gust velocity than to the mean of daily average velocity, consistent with findings of a threshold wind velocity at approximately 7 m s^{-1}; (ii) erodibility indices such as the aerodynamic roughness, friction velocity, and near-ground soil flux all indicated greater erosion at the burned than the unburned site; and (iii) the burned site had soil loss from intercanopy and canopy patches, in contrast to the unburned site, which had soil accumulation in the canopy patch type. Our results highlight the importance of accounting for finer temporal and spatial variation in wind erosion of soils and associated nutrients, and are particularly relevant for predictions of contaminant resuspension and redistribution, which historically ignore finer-scale temporal and spatial variation in wind erosion.

ACKNOWLEDGMENTS

We would like to thank Sondra Sage and Jim Yahr for sampling support. We also thank Michael H. Ebinger, Dennis G. Vasilik, Shawna L. Eisele, Robert T. Devine, and three anonymous reviewers for very helpful manuscript review. This work was supported by DOE EMSP Project 60015-TTP AL17SP23. Partial funding was also provided by the Undergraduate Research Semester Program at Los Alamos National Laboratory.

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