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ORIGINAL PAPER

Quantitative effects of vegetation cover on wind erosion and soil nutrient loss in a desert grassland of southern New Mexico, USA

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Abstract Wind is a key abiotic factor that influences the dynamics of arid and semiarid systems. We investigated two basic relationships on vegetation manipulation (grass cover reduction) plots at the Jornada Experimental Range in southern New Mexico: (1) wind erosion rates (horizontal mass flux and dust emission) versus vegetative cover, and (2) nutrient loss versus vegetative cover. The results indicate that wind erosion rates and nutrient loss by dust emission are strongly affected by plant cover; however, the importance of shrubs and grasses in reducing dust flux may not be equal. The dramatic increase of wind erosion between 75% grass cover reduction and 100% grass cover reduction suggests that sparsely distributed mesquites are relatively ineffective at reducing wind erosion and nutrient loss compared to grasses. Comparisons of nutrients between surface soils and wind blown dust indicate that aeolian transport is a major cause for the loss of soil nutrients in susceptible environments. We found that increased aeolian flux over three windy seasons (March 2004–July 2006) removed up to 25% of total organic carbon (TOC) and total nitrogen (TN) from the top 5 cm of soil, and about 60% of TOC and TN loss occurred in the first windy season (March–July 2004). The balance between net loss of nutrients by aeolian processes and the addition of nutrients by biotic processes changed from negative (net loss) to positive (net accumulation) between 50% grass cover reduction and 25% grass cover reduction. The estimated lifetime of surface soil TOC and TN of about 10 years on the plot with 100% grass cover reduction indicates that impacts of wind erosion on soil resources can occur on very short timescales.

Keywords Aeolian processes · Deserts · Soil nitrogen · Soil carbon · Vegetation cover · Wind erosion · Windblown sediment

Introduction

Aeolian processes, including erosion and transport of soils by wind, as well as deposition of windborne material, are principal geomorphic agents in the world's deserts. Aeolian transport is also a key abiotic mechanism for movement of soil nutrients within and out of arid and semiarid lands (Noy-Meir 1985; Coppinger et al. 1991; Schlesinger et al. 1996; Larney et al. 1998). Globally, wind is responsible for continental-scale transport of soil nutrients and particulate matter from arid and semiarid areas because water-based transport is limited by closed

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basins (Gillette and Pitchford 2004). This large-scale transport of windborne dust is important to the Earth's radiation balance, terrestrial soil formation and public health (Sokolik and Toon 1996; Chadwick et al. 1999; Griffin et al. 2001; Okin et al. 2004).

Wind erosion of soils is likely to play an important role in the desert grasslands of the southwestern United States, which have experienced dramatic vegetation changes including extensive encroachment by shrublands over the past 150 years (Buffington and Herbel 1965; Allred 1996; Gibbens et al. 2005). To explain this, Schlesinger et al. (1990) hypothesized that shifts in desert vegetation from grasses to shrubs are triggered by some environmental forcing (e.g., grazing, drought, warming, increased atmosphere CO_2) and are sustained by a redistribution of soil nutrients into "islands of fertility." Both water and wind can be the vectors for movement of soil nutrients (Schlesinger et al. 1990; Okin and Gillette 2001). However, much of the previous work has focused on the role of water in nutrient redistribution and the formation of islands of fertility in arid and semi-arid grasslands, and little is known about the loss and redistribution of soil resources by wind. Schlesinger et al. (2000) further pointed out that water erosion cannot, by itself, account for the depletion of soil fertility associated with land degradation in the Chihuahuan Desert. Only a few studies have investigated the role of wind in the removal or redistribution of soil resources in natural landscapes (e.g., Leys and McTainsh 1994; Larney et al. 1998). While Gillette and Pitchford (2004) have shown that conversion of grasslands to mesquite shrublands on sandy soil results in a significant increase in aeolian activity, no study to date has explicitly examined the effects of wind on soil nutrient depletion and degradation in semiarid desert grasslands under manipulated conditions.

Wind erosion rates can be expressed as aeolian sediment flux. Aeolian sediment flux has two principal components: horizontal mass flux (Q) and vertical dust flux (F_a) (Okin 2005). Horizontal mass flux (expressed in units of mass per unit distance perpendicular to the wind per unit time) is comprised mostly of saltating particles with diameter greater than 50 μm, which normally have a very local influence on vegetation and soils, and represent primarily the re-distribution of surface soil within the ecosystem (Larney et al. 1998). Vertical flux F_a (also called dust emission, units of mass per unit area per unit time) is the flux of the smallest soil particles (diameter less than 50 μm) from the surface, which are available for long-distance transport by wind (Shao et al. 1993). The primary ecological consequence of such fine particle transport lies in the fact that because these fine constituents contain a disproportionate share of soil nutrients, dust emission and re-deposition have significant impacts on soil nutrient status of local and downwind ecosystems (Leys and McTainsh 1994; Okin et al. 2004; Okin 2005). Additionally, the permanent loss of fine particles from soil by dust emission may reduce water-holding capacity and cation-exchange capacities (Hennessy et al. 1986; Leys and McTainsh 1994). This may increase the susceptibility of arid and semiarid grassland to drought and climate change.

In this study we use landscape-scale plant cover reduction experiments in a desert grassland to study the influence of aeolian processes on the nutrient content of soils subjected to enhanced wind erosion. In particular, we report (1) the relationship between horizontal mass flux, dust emission, and vegetative cover; (2) the relationship between soil nutrient flux on dust particles and vegetative cover; and (3) the relationship between soil nutrient depletion and horizontal aeolian flux rates.

Methods

Study site

This research was conducted at the Jornada Experimental Range (JER), 35 km northeast of Las Cruces, New Mexico, which is part of the National Science Foundation's Long Term Ecological Research (LTER) network. The JER is in the northern part of the Chihuahuan Desert, which occurs in the Mexican Highland section of the Basin and Range Province. The climate of JER is classified as warm and typical of arid, semi-desert grassland (Buffington and Herbel 1965). The average maximum temperature of 36°C is usually recorded in June; during January the average maximum temperature is 13°C (Buffington and Herbel 1965). Mean annual precipitation (1915–2002) recorded at the JER headquarters is 247 mm, with 52% typically occurring in brief but intense small-scale convective thundershowers during July to

September. The potential evapotranspiration is about 2,300 mm per year (Gibbens et al. 2005). The erosive winds at JER are highly consistent with 79% occurring from a southwesterly direction, and dominant wind erosion events happening during early March to May (Helm and Breed 1999). Elevation of the JER varies from 1,200 to 1,300 m. The topography of the Jornada Basin consists of gently rolling to nearly level uplands, interspersed with swales and old lake beds (Buffington and Herbel 1965). Soils in the JER are quite complex but sandy soils and sandy loams are generally well-distributed (Bulloch and Neher 1977).

Our field experiment was set up in Pasture 11 of the JER (32°56′ N, 106°75′ W). This site is on the "sand sheet" geomorphic surface that comprises most of the western and northern portions of the JER. Soils at the site have very sandy texture, normally with coarse sand (0.5–1.0 mm) >95% and silt and clay <5% (Fig. 1). Exposure of a thick underlying caliche layer is rarely seen. The site was chosen because invasion of honey mesquite (*Prosopis glandulosa*) into the former grasslands at this location was light compared to other areas of the sand sheet. The dominant grasses at the site belong to the genus *Sporobulus*, though there is substantial black grama (*Bouteloua eriopoda*) in some areas. There is also substantial cover of yucca (*Yucca elata*) at the site.

Experimental design and sampling regime

The field experiment was established in March 2004 before the beginning of the windy season. Five treatments (including the control plot) were aligned parallel to prevailing winds, each 25×50 m^2 (Fig. 2). Inter-treatment buffers of 25 m were left to minimize the interference between treatments. Varying levels of cover were imposed by grass cover reduction. In the 100% grass cover reduction plot (simplified as T100), all grasses, perennial semi-shrubs such as *Gutierrezia sarothrae*, and perennial forbs were removed (hereafter referred to as "grass removal" only). In the 75% grass cover reduction plot (T75), every three out of four plants mentioned above were removed; and so on for 50% (T50) and 25% (T25) grass cover reduction. Shrub cover was low in the sites at the beginning of the experiment, and shrubs were not removed. Reduced cover on treatment plots was maintained during the entire experimental period. To equalize the impacts of grass cover reduction on the surface soil, the soil surface was lightly raked and litter was collected from the surface of each plot, including the control. Within each treatment, two 5×10 m^2 subplots (named U1 and U2) were set up to collect surface soil samples and to install windblown sediment samplers. A 10-m meteorological tower with five anemometers arranged at heights of 0.3, 0.7, 1.7, 4.5, and 10.0 m was also installed about 60 m upwind of the treatment area. Wind speed and directions were measured every 5 min, and data were recorded using a solar-powered, cellular-modem-equipped data logger. In addition, a tipping bucket rain gage was installed about 100 m upwind of the treatment area and the precipitation during the whole experimental period was recorded. Figure 3 shows the monthly precipitation from Jan.-May of 2004–2006 in the research site.

Soil sampling was conducted on one of the 5×10 m^2 subplots (U1) within each of the five treatments (Fig. 2). In each sampling area, 50 soil samples from the top 5 cm with 2.5 cm diameter were taken before the manipulation of the grass cover reduction (March 2004). Subsequent soil samples were taken in July 2004, 2005, and 2006. Thus, a total of 1,000 (5 treatments × 4 dates × 50 samples per date per treatment = 1,000) soil samples were collected and analyzed. Soil samples were randomly distributed with their locations chosen without regard to plant locations.

To collect aeolian sediments on each treatment, Big Spring Number Eight (BSNE) dust samplers were installed on all U1 and U2 subplots (Fig. 2).

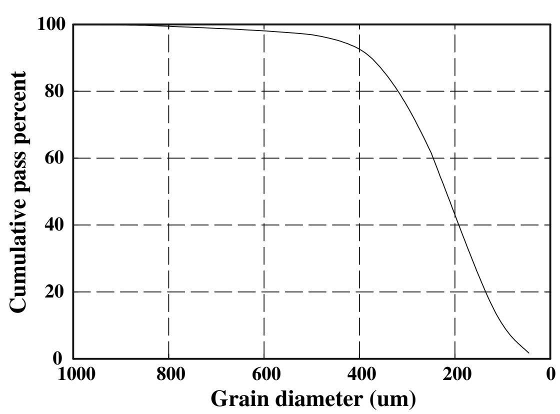

Fig. 1 Typical grain-size distribution curve for the surface soil before the implementation of the treatment

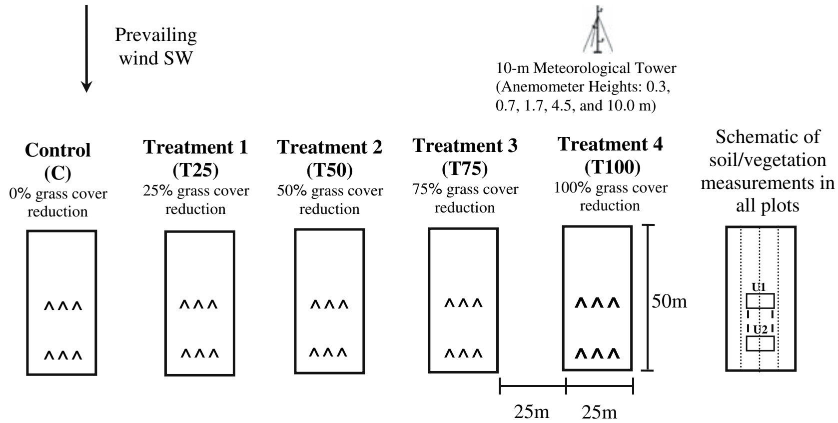

Fig. 2 Experimental layout in the field. "^" represents 1.2-m BSNE dust samplers; "**^**" represents 2.5-m BSNE dust samplers installed in the T100 treatment only. The right-most figure shows more detailed layout of sampling locations in each treatment. There are two 5 × 10 m^2 soil and BSNE subplots, where soil sampling was conducted in the U1 subplots and BSNEs were installed in both U1 and U2 subplots. Three 50-m line-intercept transects were set up for vegetation monitoring (denoted by fine dashed line), and one 10 × 20 m^2 plot was set up (thick dashed lines, including U1 and U2 subplots) where plant lateral coverage was measured. The field site was fence-protected

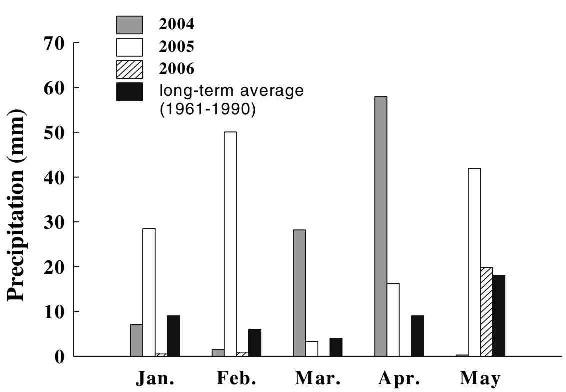

Fig. 3 Monthly precipitation (Jan.–May) of the research site during the experimental period (2004–2006) as compared to the long-term average. Precipitation of 2004 and long-term average was recorded by the Jornada LTER Weather Station

These collectors were developed by Fryrear (1986) and have been independently tested by Shao et al. (1993). The sampling cluster consists of an array of samplers attached to a pivoting wind vane and mounted at a different height on a supporting pole. The heights of individual samplers can be adjusted up or down the supporting pole. Each sampler has a vertical inlet area of about 2 cm wide and 5 cm high. Sediment-laden air enters the sampler inlet area and is slowed within the sampler by a diffuser section. Because the inlet is on the front and vertically oriented, the BSNE dust samplers collect outgoing dust while incoming dust is not measured. Atmospheric deposition was not measured in this study.

The BSNE samplers were placed 3–4 m apart approximately along the center of each 5 × 10 m^2 subplot, lined up perpendicular to the prevailing wind. We avoided placement of BSNE samplers directly downwind of a yucca, mesquite plant or mesquite dune. Three BSNE samplers were installed in each subplot to provide an estimate of horizontal flux independent of the effects of local vegetation and microtopography (Gillette and Pitchford 2004). In subplots other than T100 (U1, U2), 1.2 m-tall BSNE poles were used with four collectors situated at heights 0.1, 0.3, 0.6, and 1.2 m above the ground. In order to estimate vertical dust flux, two groups of 2.5 m-tall BSNE poles were installed in the T100 subplots (U1 and U2) in July 2004, separated by 15 m with six collectors located at heights 0.12, 0.3, 0.45, 0.9, 1.6, and 2.5 m. BSNE samples were collected twice per year in early March (sampling time of July to March of the following year) and middle July (sampling time of March–July) from 2004 to 2006. Only 95 windblown sediment samples were collected in July 2004 because some of the BSNE samplers were blown over; about 140 samples were obtained in each sampling time thereafter. A total of about 650 aeolian samples were collected.

Plant cover and community composition on each plot were monitored by three 50-m line-intercept transects (Fig. 2). Fractional plant cover was calculated by adding the lengths of each plant along the transect and dividing by the total length of the transect. A subsequent visit to the research site in July 2004 found no noticeable re-growth of the perennial grasses after the initial removal. However, due to ample rainfall in January and February of 2005 (Fig. 4), considerable re-growth was found in March 2005, and site maintenance was conducted. Re-growth in March 2006 was only minor, and lesser maintenance was conducted. Although annuals were found in early spring, they are typically dead and gone by the beginning of the windy season, and annual cover is ignored.

To further quantify the effect of vegetation cover on wind erosion, the lateral plant cover was also measured. The lateral cover is generally defined as $\lambda = NA_p$, where N is the number density of plants, and A_p is the total area of plants projected onto a plane perpendicular to the ground and the direction of wind, divided by the ground area in which the plants were measured (Okin 2005). The locations of plants were measured using a Trimble 3600 Total Station System within the 10 × 20 m^2 plant investigation plot; height and basal area of each plant were also measured (Fig. 2). Fractional plant community composition before the implementation of grass cover reduction is shown in Table 1, and fractional plant cover, lateral plant cover and litter cover after the initial cover reduction are shown in Table 2.

Laboratory analysis

Soils had a volumetric moisture content of <3% at the time of collection. In the laboratory, each air-dried soil sample was sieved to 2-mm, and each BSNE aeolian sediment sample was weighed to 0.001 g. For analysis of soil total organic carbon (TOC) and total nitrogen (TN), about 5 g of sub-sample was obtained by passing each soil sample through an open pan riffle sampler (Model H-3962, Humboldt MFG. Corporation, Norridge, IL). Each of these 5 g sub-samples was then ground to a fine power by a ball mill (Model 2601Cianflone Scientific Instruments Corporation, Pittsburgh, PA). To further ensure the complete decomposition of all carbonates in TOC measurements, vanadium oxide (V_2O_5) was used as a combustion catalyst with the mass ratio of catalyst to sample of 1:1. TOC and TN analyses were conducted on a Shimadzu TOC-V_{CSN} total organic carbon analyzer with a SSM-5000 solid sample analyzer and a TNM-1 total nitrogen measuring unit. In this system, TOC was calculated as the difference between total carbon (TC) and total inorganic carbon (TIC). Because of the low mass and fine-grained particles, the BSNE dust samples were not ground for TOC and TN analyses except for those collected in the lowest samplers, where the mass of the dust samples usually exceeded 10 g.

For analysis of water-extractable anions in the soils in each sample, approximately 6 g of sieved soil was shaken with 30 ml of distilled water for 30 min, and centrifuged at 4,500 rpm for 10 min. Extracts were analyzed on a Dionex ICS-2000 ion chromato-

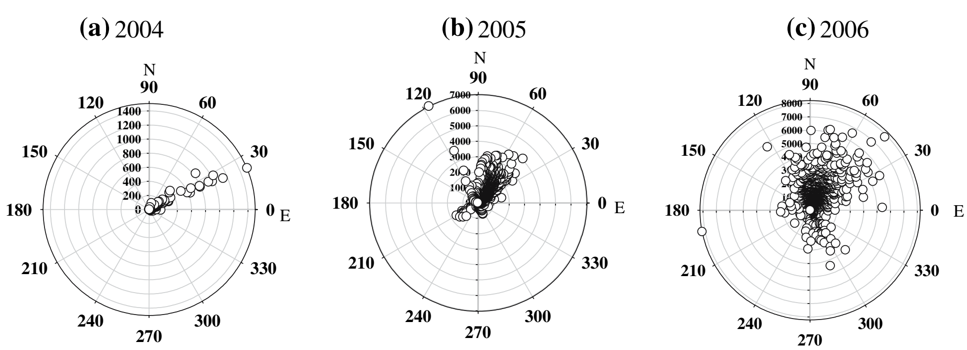

Fig. 4 Sand drift potential (Mar.–May) of the research site during the experimental period (2004–2006). Wind data of 2004 were obtained from the Jornada LTER Weather Station located about 3 km northeast of the research site. Available data were the average of hourly values

Table 1 Fractional plant cover (%) of the treatments before the manipulation of the grass cover reduction (March 2004), with 100%, 75%, 50%, and 25% grass cover reduction on T100, T75, T50 and T25, respectively. No cover reduction on the control plot

Treatments	Total	Grass	Gutierrezia	Forb	Prosopis
T100	25	9	4	1	11
T75	22	9	2	2	8
T50	22	10	<1	<1	9
T25	18	8	3	<1	5
Control	19	12	<1	2	0

Table 2 Percent of fractional cover, lateral cover, and litter cover of the treatments after the manipulation of the cover reduction (March 2004)

Treatments	T100	T75	T50	T25	Control
Fractional	11	13	16	15	19
Lateral	8	16	32	46	46
Litter	4	10	12	19	18

Appropriate coverage was maintained during the whole experimental period. Treatments are 100% (T100), 75% (T75), 50% (T50), and 25% (T25) grass cover reduction, no cover reduction on the control plot

graph with an Ion Pac AS 18 anion exchange column and a 4 μm inline filter for the concentration of Cl^-, O_x^{2-} (oxalate), NO_2^-, NO_3^-, SO_4^{2-}, and PO_4^{3-}. No anion analysis was performed for the windblown dust samples due to the insufficient sample mass. All results were converted into units of milligrams species per kilogram of soil.

In almost all soil samples, O_x^{2-} was not detected and data for O_x^{2-} were not subjected to further analysis; NO_2^- was also frequently near or below sensitivity and NO_2^- was added to NO_3^- and presented as $NO_2^- + NO_3^-$ in the following sections. For both soil and windblown dust samples, TIC was only a few mg/kg compared to generally more than 2000 mg/kg of TC, so TIC was omitted and TC results were taken as TOC.

Calculations

To describe the potential soil particle movement under wind erosion conditions, sand drift potential (*DP*) for each windy season was calculated according to Fryberger and Dean (1979):

$$DP = (u - u_t)\Delta t \tag{1}$$

where u is the wind speed recorded by the meteorological tower at the height of 4.5 m when $u > u_t$, u_t is the threshold wind speed for wind erosion. Gillette et al. (2006) suggest that the aerodynamic roughness length is about 5 cm in a mesquite and grass mixed grassland in the JER, while the u_t is about 4.5 m/s. Δt is the time that the wind blew from a specific direction at a specific speed. Δt was taken as 5 min for simplicity and results for all times with $u > u_t$ are given.

The mass of airborne particles collected by a BSNE sampler was divided by the area of the sampler's inlet (1,050 mm^2) and the time of the collection to obtain the time-averaged horizontal mass movement $q(z)$ (units of mass per unit area per time), where z is the height of the center of the inlet above the ground. The results of $q(z)$ were then fit to the empirical formula used by Shao and Raupach (1992) and Gillette (1997):

$$q(z) = ce^{(az^2+bz)}, \tag{2}$$

where a, b, and c are fitting constants. The fit of BSNE data to this equation gave an average value of $r^2 = 0.97$. No apparent differences in r^2 values with the location of BSNE samplers or sampling time were observed. The values for total horizontal mass flux Q were calculated by integrating the horizontal flux $q(z)$ from the ground to 1-m height:

$$Q = \int_0^{1m} q(z)\,dz, \tag{3}$$

where Q has units of mass per unit length per time. The maximum height of integration is 1 m because $q(z)$ decreases rapidly with height, and typically less than 10% of the flux occurs at heights >1 m.

Vertical dust flux (F_a) is difficult to measure directly. Shao et al. (1993) and Gillette et al. (1997) have developed a relatively simple relation between horizontal mass flux and vertical dust flux as:

$$F_a = kQ, \quad (4)$$

where k is a soil-specific constant related to soil texture, crusting, and moisture regime, with a typical value of 10^{-4}–10^{-5} cm^{-1}. Once a reliable value of k has been determined from an area, values of F_a can be easily determined from measured values of Q. Conservation of mass implies that divergence of the horizontal flux gives the vertical flux at any point. Therefore, k is calculated as:

$$k = \frac{\Delta Q/\Delta x}{Q}, \quad (5)$$

where x is the windward distance, Q is set as the average flux of the 2.5 m-tall BSNE towers, ΔQ is estimated as the difference between the average flux at the two sets of 2.5 m-tall BSNE towers on the T100 (100% grass removal) plot integrated from 0 m to 2.5 m, and Δx is the distance separating these two sets of towers, 15 m.

Assuming total mass of the soil layer does not change with time and dust emission F_a is constant, Okin et al. (2001) characterized the soil nutrient change over time as:

$$\frac{dC_i^s}{dt} \approx -e_i \frac{F_a}{\rho_s D} C_i^s, \quad (6)$$

where ρ_s is the bulk density of the soil assumed to be uniform within the layer D, C_i^s is the concentration of nutrient i in the surface soil, and the factor e_i is the ratio of C_i^d (concentration of nutrient i in the dust) to C_i^s defined as enrichment ratio for nutrient i. Integrating Eq. 6 gives:

$$C_i^s(t) = C_i^s(0) \exp\left(-\frac{e_i F_a}{\rho_s D} t\right), \quad (7)$$

where $C_i^s(t)$ is the concentration of nutrient i in the soil at time t, and $C_i^s(0)$ is the initial concentration of nutrient i in the soil. This equation states that nutrient content decreases exponentially with time under conditions of constant wind erosion, and that the time for the concentration of nutrient i to drop to *1/e* times $C_i^s(0)$ (the e-folding time) is given by

$$t_i = \frac{\rho_s D}{e_i F_a} = \frac{T}{e_i}, \quad (8)$$

where T is the time for a complete excavation of a layer of depth D.

To quantify the soil nutrient loss by dust emission, the nutrient flux N_i (units of mass of nutrient per area per time) is introduced. Flux of nutrient species i can be calculated by two ways:

$$N_i = F_a C_i^d \quad (9)$$

or

$$N_i = F_a C_i^s e_i, \quad (10)$$

Equation 10 is used when concentrations of certain nutrient species are difficult to obtain in the dust but easy to measure in the surface soil, and the enrichment ratio can be approximated by the easy-to-measure species, such as TN.

Results

Precipitation and sand drift potential

During the experimental period (2004–2006), precipitation that occurred from January to May was not evenly distributed (Fig. 3). Long-term records show that rainfall from January to May accounts for about 17% of the whole year. In 2004, 24% of the year's rainfall occurred in January to May, mostly in March and April. Spring of 2005 may be described as "wet," with 56% of the year's rainfall occurring in January to May. On the contrary, spring of 2006 was extremely dry, with almost no rainfall occurring from January to April.

Figure 4 shows that during the windy seasons, predominant sand drift occurred toward the northeast, parallel to the direction of the experimental layout (Fig. 2). Although sand drift direction in 2006 showed a little more variability compared to those of 2004 and 2005, most of the high drift potential still occurred between north and east. The predominant northeast sand drift direction suggests that the potential lateral movements of soil particles among the treatments were minor.

Wind erosion rates and vegetative cover

Mean horizontal mass flux, Q, generally increased with increasing grass cover reduction (Fig. 5). ANOVA shows that significant increases (compared to the control) in Q ($P = 0.05$) were mostly found on 100% (T100), 75% (T75) and 50% (T50) grass cover reduction plots, while no significant difference in Q between the 25% grass cover reduction plot (T25) and the control plot was found. For the mass flux in individual plots, except for the March 2006 sampling period, Q_{T100} was 3–5 times greater than that of Q_{T75} and more than 10 times greater than that of the control plot, where the lowest flux was observed during all five sampling periods. Horizontal mass flux obtained in March 2006 was substantially lower than all the other sampling periods, and the variation of the mass flux between plots was small as well. Horizontal mass flux from the 100% grass cover reduction plot accounted for about 80% of the total horizontal mass flux from all plots. Seasonal variation of horizontal mass flux is noteworthy. For example, mean daily horizontal fluxes measured from March to July (Fig. 5a) were substantially greater than those measured from July to March (Fig. 5b), with more than 75% of the mass flux occurring from March to July. These results agree with Helm and Breed (1999) that dominant wind erosion events happen during early March to May in the Jornada Basin.

Horizontal mass flux did not increase linearly with plant cover reduction. We observed considerably more flux on the T100 plot (100% grass cover reduction) relative to other plots. Indeed, mass flux decreased only slightly with grass cover reduction ranging from 75% (T75) to 0% (control). This pattern is especially clear for all the wind periods except for March 2006. Table 2 shows that post-treatment fractional covers of all plots were quite close, especially T100 (11%) and T75 (13%), while their lateral covers were strikingly different. The distribution patterns of horizontal mass flux with vegetation cover (both fractional and lateral cover) may suggest that a threshold plant cover might exist between 100% grass cover reduction (T100) and 75% grass cover reduction (T75). Translating to lateral cover, the threshold of lateral cover may be about 9%, above which wind erosion is close to "background" levels, and below this threshold, wind erosion increases dramatically.

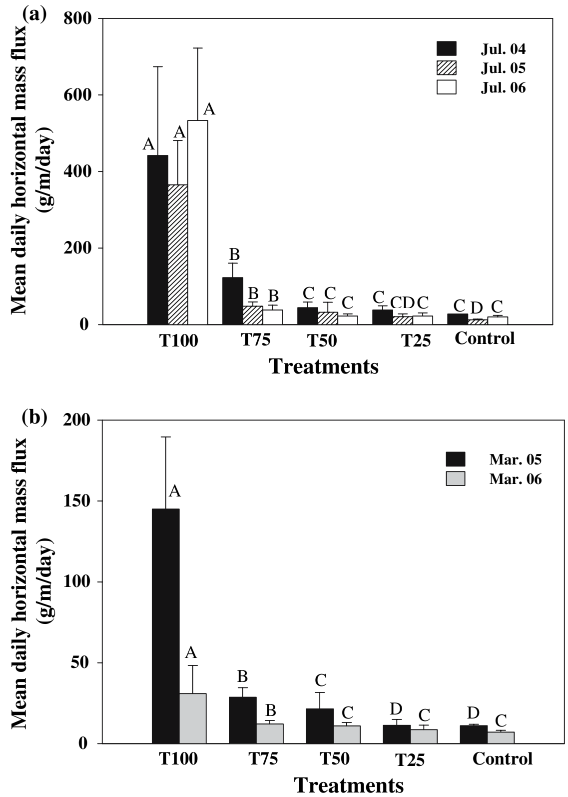

Fig. 5 Mean horizontal mass flux versus different levels of grass cover reduction measured in July (**a**) and in March (**b**). The average period for July and March sampling time is 4 and 8 months, respectively. Each mean flux was calculated based on six BSNE samplers within each treatment except for July 2004 collection when some BSNE samplers failed to collect dust. Error bars are one standard deviation. ANOVA for treatments within each sampling period, $P < 0.05$ for significance, labeled by different letters. No statistics were conducted for different sampling periods. Treatment notations are 100%, 75%, 50%, and 25% grass cover reduction on T100, T75, T50, and T25, respectively, and no cover reduction on the control plot

The k factor (relationship between horizontal and vertical flux) estimated from 100% grass cover reduction plot (T100) using all available horizontal mass flux data from 2004 to 2006 was $2.0 \pm 0.2 \times 10^{-4}$ cm^{-1}, matching the results of Gillette et al. (1997) from Owens Dry Lake. The k factor calculated based on July windblown dust was slightly greater than that from March, however, very little variation was found among the July sampling periods. Assuming k is a constant for all plots, vertical dust flux (F_a) for the other plots was calculated using Equation 4 (Fig. 6). Average

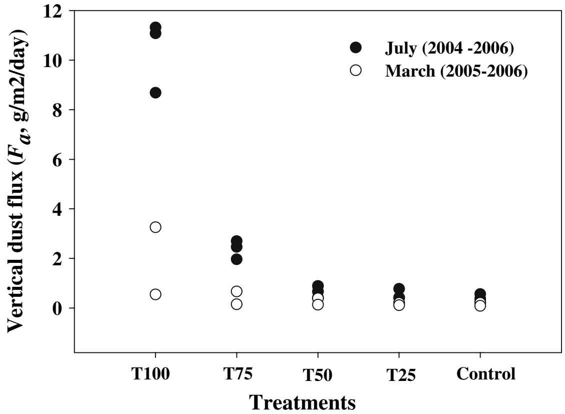

Fig. 6 Vertical dust flux (dust emission) on treatments measured in July and March. Dust emission was calculated based on the two sets of 2.5-m BSNE samplers in the 100% grass removed treatment (T100). Treatment notations are the same as in Fig. 5

March–July F_a = 10.36 g/m^2/day, while F_a = 1.90 g/m^2/day averaged from August to February of 2005 and 2006. The yearly average of F_a was obtained by summing these two periods.

Nutrient enrichment in windblown sediment

Average enrichment ratios of TOC and TN calculated in the topmost BSNE samplers (1.2 m) in different plots are shown in Table 3. During the 3-year experiment, average enrichment ratio of TN, e_{TN}, varied from 2.7 in the100% grass cover reduction plot (T100) to 4.0 in the control plot; TOC enrichment ratio, e_{TOC}, was slightly greater than that of TN, varying from 3.2 to 6.0. Although both e_{TOC} and e_{TN} generally decreased with decreasing vegetation cover, significant differences only occurred in the 100% grass cover reduction plot (T100) compared to the control plot.

Average C/N ratio in soils fell within the range of 8–10 (Table 4). ANOVA showed that there were no significant differences (P = 0.05) among plots before the grass removal in March 2004. However, soil C/N

Table 3 Mean enrichment ratio e of TOC and TN versus different levels of grass cover reduction

Treatments	T100 (100%)	T75 (75%)	T50 (50%)	T25* (25%)	Control
TOC	3.24^{a}	4.81^{ac}	5.51^{bc}	6.33^{b}	5.92^{bc}
TN	2.69^{a}	3.39^{ab}	4.04^{b}	3.92^{b}	4.02^{b}

Enrichment ratio was calculated based on windblown sediments (trapped at 1.2 m) and surface soils collected in July of 2004–2006. An ANOVA test was performed across different plots, significance was indicated by different letters at the P < 0.05 level (n = 18 for plots other than T25 and n = 12 for T25). Percent of grass cover reduction is indicated by the numbers in the parentheses in each treatment

* Only July 2004 and 2005 data were used

Table 4 Statistical analysis of surface soil and windblown sediment C/N ratio from 2004 to 2006

Treatments		T100	T75	T50	T25	Control
Soil C/N ratio	Pre- treatment (Mar. 2004)	8.78^{a}	9.19^{a}	8.63^{a}	8.66^{a}	8.76^{a}
	Post-treatment (Jul. 2006)	7.67^{a}	8.41^{ab}	7.89^{ac}	8.59$^{ab\#}$	8.94^{b}
	Change (%)	−12.69*	−8.53*	−8.57*	−0.84	+2.09
Sediment C/N ratio	Jul. 2004	11.58	12.33	13.07	13.31	13.62
	Jul. 2006	10.86	12.01	11.83	12.82	12.93
	Change (%)	−6.22	−2.58	−9.43*	−3.68	−5.07

Sediment C/N ratio was calculated using the TC and TN content of the top BSNE traps (1.2 m). "−" and "+"indicate C/N ratio decrease and increase, respectively. For soil C/N ratio, a two-way ANOVA was performed for "plots" and "years," followed by a one-way ANOVA for the difference between treatments within each year. A paired t-test was conducted to test the difference before (2004) and after (2006) treatment, significant results were indicated by "*" (P = 0.05). For C/N ratio of windblown sediment, only a paired t-test was conducted for C/N ratio in 2004 and 2006 in each treatment, significant change was indicated by "*" (n = 50 for surface soil and n = 6 for windblown sediment). Treatments are 100% (T100), 75% (T75), 50% (T50), and 25% (T25) grass cover reduction, and no cover was reduced on the control plot

\# Jul. 2005 data were used

Table 5 Mean and standard deviation of lifetime (year) and *e*-folding time (year) of TOC, TN, and PO_4^{3-} in the top 5 cm soils on treatments with different grass cover reduction

Treatments		T100	T75	T50	T25	Control
e-folding time (year)	TOC	3.3 ± 0.9	12.7 ± 2.9	13.9 ± 5.7	14.0 ± 1.0	21.6 ± 8.6
	TN	5.6 ± 2.2	27.0 ± 7.8	26.1 ± 3.1	36.4 ± 0.2	50.3 ± 18.1
	PO_4^{3-}	4.6 ± 1.3	25.3 ± 8.2	23.4 ± 7.4	36.0 ± 0.9	51.1 ± 20.8
Lifetime (year)	TOC	10.6 ± 1.7	59.7 ± 9.3	75.7 ± 10.4	86.4 ± 18.1	124.4 ± 34.5
	TN	13.8 ± 2.4	84.5 ± 5.8	105.0 ± 16.7	136.5 ± 24.1	189.3 ± 46.3
	PO_4^{3-}	11.5 ± 2.0	78.1 ± 4.3	91.1 ± 14.5	135.2 ± 28.0	191.1 ± 56.4

Assuming soil density = 1.25×10^3 kg/m^3. Grass cover reductions for treatments T100, T75, T50, and T25 are 100%, 75%, 50%, and 25%, respectively, and no cover was reduced on the control plot

ratio measured in July 2006 shows significant differences among plots, especially between 100% grass cover reduction plot (T100) and the control plot ($P = 0.05$). Table 4 also shows that on plots where grass was removed, soil C/N ratio generally decreased after 3 years of wind erosion, whereas on the control plot a slight increase was detected. A paired *t*-test for individual plots further shows that this decrease was significant ($P = 0.05$) for plots T100, T75 and T50.

C/N ratios in windblown sediment at 1.2 m high were about 30% higher than in the surface soil, and decreased with increasing grass cover reduction. Sediment C/N in the windy season immediately following the grass removal fell in the range of 11–13, and 2 years later, this ratio was slightly lower for both cover-reduced plots and the control plot. Similarly, a paired *t*-test for individual plots was performed and a significant decrease of C/N ratios ($P = 0.05$) over time only occurred in the 50% grass cover reduction plot (T50).

Nutrient flux and soil nutrient depletion

Figure 7 shows that the flux of all nutrients from the experimental treatments also increased with increasing grass cover reduction. The fluxes of TOC and TN from experimental treatments were much greater than those of anions due to the relative size of these pools. Although the enrichment factor was low on this plot, the total nutrient flux on the 100% grass cover reduction plot (T100) was still substantially greater than the other plots because of much higher dust flux. The pattern of nutrient flux from our experimental treatments is different from that observed in horizontal mass flux (Fig. 5). Horizontal mass flux exhibited a threshold between T75 (75% grass cover reduction) and T100 (100% grass cover reduction), whereas nutrient flux, especially TOC and TN, appears to exhibit a threshold between T50 (50%

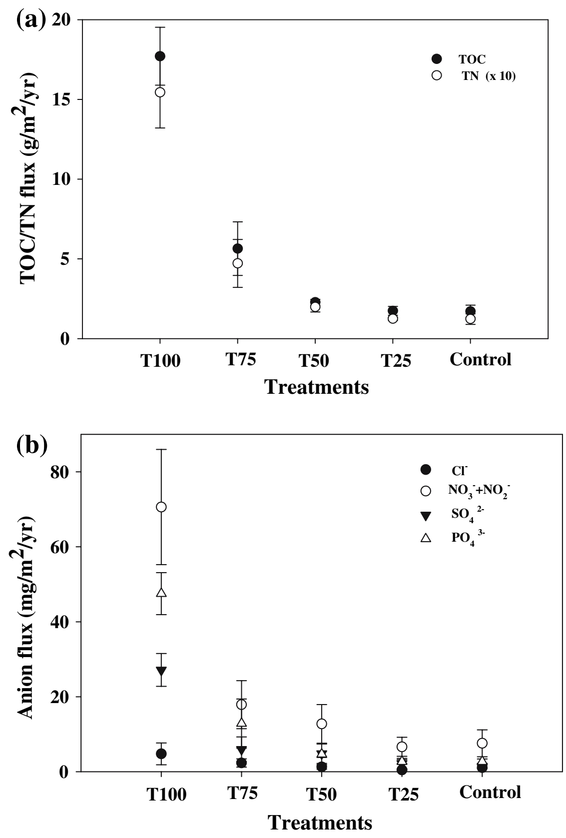

Fig. 7 Soil nutrient removal (nutrient flux) by wind erosion from 2004 to 2006 for (**a**) TOC and TN (g/m^2/year), and (**b**) anions (mg/m^2/year). Vertical dust flux was adopted from Fig. 5. Note the unit difference between TOC/TN and anions. Notations for treatments are 100%, 75%, 50%, and 25% grass cover reduction on T100, T75, T50, and T25, respectively

Table 6 Net soil nutrient change (%) from surface 5 cm after three windy erosion seasons (March 2004–July 2006) versus different vegetative cover

Element	T100	T75	T50	T25[a]	Control
TOC	−26.8*	−25.2*	−10.8	−3.3	+12.0*
TN	−27.8*	−17.8*	−0.4	+14.0*	+14.1*
Cl^-	−86.1*	−48.1*	+17.5*	−13.7*	−61.8*
SO_4^{2-}	−89.9*	−30.0*	−11.0	−42.1*	−73.9*
NO_3^-+NO_2^-	−78.5*	−28.7*	−23.9*	−11.4*	−15.1*
PO_4^{3-}	−13.5*	+5.4	+23.2*	+4.5	+6.1

Calculations were based on the mean results of 50 samples. Nutrient loss was donated by "−" and "+" was used to represent nutrient enhancement. Nutrient status before wind erosion (Mar. 2004) and after wind erosion (July 2006) in each treatment was compared using paired *t*-test. Significant differences were indicated by "*" (P = 0.05, n = 50)

[a] Jul. 2005 data were used

grass cover reduction) and T75 (75% grass cover reduction).

The *e*-folding time (t_i) and lifetime (T) of a nutrient with respect to loss of that nutrient by wind erosion are important indices of the impact of wind erosion on nutrient cycling. During the 3-year experimental period, for D = 5 cm, dust emission shown in Fig. 6, and a soil bulk density of 1.25×10^3 kg/m^3, the *e*-folding times of TOC, TN, and PO_4^{3-} on the 100% grass cover reduction plot (T100) are about 3–5 years, with t_{TOC} being slightly shorter than the other two. Table 5 also shows that the lifetime of TOC, TN, and PO_4^{3-} on the 100% grass cover reduction plot (T100) is only about 10 years, whereas the lifetime of these elements on the control plot is more than 100 years. It is noteworthy that the standard deviations of T and t_i are fairly large based on 2004–2006 wind erosion data. This may be attributed to differences in nutrient removal rates, which in turn, are the results of the differences in horizontal mass fluxes through the experimental period (Fig. 5).

After 3 years of enhanced wind erosion, an appreciable loss of soil nutrients was observed (Table 6). For instance, more than 25% of the TOC and TN were removed from the top 5 cm of soils of the 100% grass cover reduction plot (T100). Soil nutrient loss during this period was also larger for the anions, such as NO_3^-, that may limit primary production. Nearly all the soil nutrient losses are significantly greater (P = 0.05) in T100 (100% grass cover reduction) and T75 (75% grass cover reduction) plots compared to the other treatments, except for PO_4^{3-} in T75, which surprisingly showed a 5.4% increase. In contrast, soil TOC, TN and PO_4^{3-} had significant increases (P = 0.05) in the control plot during the experimental period. The loss of soil nutrients during the 3-year experimental period declined through time. For example, about 15% TOC and TN was lost from the 100% grass cover reduction plot (T100) in the first windy erosion season immediately following the grass removal, whereas in the following 2 years, about 7% additional loss was detected (Fig. 8). For non-limiting nutrients, such as Cl^- and SO_4^{2-}, no regular pattern was found related to different amounts of vegetation removal. However, a very significant loss of Cl^- and SO_4^{2-} was detected in T100 and T75 plots. The loss of Cl^- and SO_4^{2-} in the control plot presumably is related to the removal of plant litter at the beginning of the field experiment. TOC, TN and NO_3^- + NO_2^- exhibited some increase from T75 to T50 during the experimental period. We thereby propose that another threshold may exist between 50% grass cover reduction (T50) and 25% grass cover reduction (T25), and probably closer to the latter.

Our results indicate that dust flux of >1 g/m^2/day (365 g/m^2/year) is common in our plots (Fig. 6), whereas Reheis (2003) reported that long-term atmospheric dust deposition is normally less than 10 g/m^2/year in southwestern United States. In addition, analyses of samples collected in Reheis-type (Reheis 2003) dust collectors throughout the JER in 2003–2004 (unpublished data) suggest approximately 0.1 g/m^2/year N deposition, compared to our estimate of ~1.5 g/m^2/year N emission from the T100 plot (Fig. 7). Furthermore, measurable loss of N from surface soils in the T100 plot (Fig. 8)

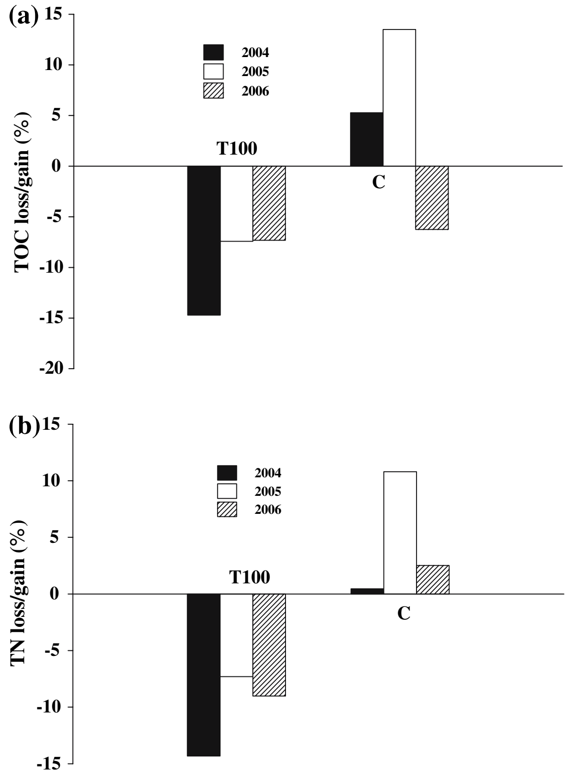

Fig. 8 Yearly net loss or gain (%) of TOC (**a**) and TN (**b**) on the 100% grass cover reduction plot (T100) and the control plot (C) from 2004 to 2006. Net loss was donated by negative numbers and net gain was represented by positive numbers

shows that on this plot nutrients lost from wind erosion are not being replaced by atmospheric deposition. Thus, although we did not measure atmospheric deposition in this study, our results show that sediments and nutrients lost from at least some of the plots are not replaced by atmospheric deposition.

Discussion

Sediment flux

The ability of plant cover reduction to increase the transport capacity of wind has been the subject of very few studies in natural ecosystems. In the Kalahari Desert, Wiggs et al. (1994) found that the cover reduction by fire, grazing, or drought resulted in up to 200% increase in near-surface wind velocity, and reduction of vegetation cover by burning reduced the mean shear stress velocity by about 60%. However, the authors did not investigate the actual increase of dust flux after such plant cover reduction.

Our results of sediment flux on different plots support the hypothesis that lateral cover of vegetation, rather than fractional cover, is more important in reducing wind erosion, but is highly nonlinear in its efficacy (Marshall 1971; Raupach et al. 1993; Liu and Westphal 2001). They are also consistent with other observations (e.g., Lancaster and Baas 1998; Gillette and Pitchford 2004) and models (e.g., Gillette and Passi 1988; Okin 2004, 2007). We found strong indication of the existence of a threshold for both horizontal mass flux and dust emission, though it should be noted that there is still horizontal flux at higher lateral covers. This threshold exits between 100% (T100) and 75% grass cover reduction (T75). Although actual plant cover on the 100% grass cover reduction plot (11%) is only slightly lower than that of the 75% grass cover reduction plot (13%), wind erosion is much higher in the former than in the latter. Plant community composition monitoring indicates that almost all 11% cover on the T100 plot (100% grass cover reduction) is mesquites, however, at least 5% of the cover is composed of grasses on the T75 plot (75% grass cover reduction). The lateral cover, which accounts for both density and dimension of plants, more accurately reflect the effective cover difference between T100 and T75, with lateral cover of 8% on the former plot and 16% on the latter (Table 2). These results indicate that sparsely distributed multi-stemmed mesquite plants provide much less protection than grasses of the same fractional cover value. Also in the JER, Gillette and Pitchford (2004) found that ecosystems dominated by mesquite-coppice dunes generated far more aeolian sand flux than other important ecosystems in the region, such as creosote bush shrublands and grama grasslands, even when the ecosystems have comparable aboveground biomass.

The reduction of vegetation cover at the study sites has likely resulted in a change in the threshold wind velocity necessary to initiate particle movement (Raupach et al. 1993). Breed and McCauley (1986) found that erosivity varies with the cube of excess velocity over the threshold velocity. The decrease of threshold velocity in areas with lower cover occurs through at least two mechanisms: (1) some of the energy of saltating soil particles is transferred to surface soil particles, and the threshold velocity is effectively reduced once the saltation process has initiated; (2) the removal of grasses increases the momentum transferred to the soil surface due to reduced momentum extraction by vegetation. Additionally, the removal of plant litter and lack of subsequent input of plant litter in the 100% grass cover reduction plot not only exposed surface soil to wind directly, but also may have decreased soil moisture, resulting in the increase of susceptibility of the surface to wind erosion.

Nutrient flux

With the continuous transport of soil particles by wind, fine materials are winnowed from the surface, causing the soil texture to become coarser and less fertile (Larney et al. 1998). However, reported enrichment ratios in arid and semiarid regions vary. Larney et al. (1998) reported an average enrichment ratio of only 1 for both organic C and total N for windblown dust trapped at 0.25 m height on a semiarid Canadian prairie. In the Mallee area of Australia, Leys and McTanish (1994) found windblown soils contained 10 times more organic C and N in the dust collected at 3 m height than the surface soil. These studies suggest that (1) finer particles with higher nutrient content tend to transport at greater heights, and (2) enrichment ratio is a site specific factor which may be related to the intensity of wind erosion, soil texture and the height at which the airborne sediment was collected. In addition, enrichment ratio may be also related to the depth of surface soils being sampled since nutrient contents can decline rapidly with soil depth. For example, Cihacek et al. (1993) compared the 0–7.5 cm surface layer with windblown sediment properties and reported higher enrichment ratios of organic C and total N than Larney et al. (1998), where the surface soils of 0–2.5 cm were collected. Additionally, our results suggest that loss of vegetation cover could result in a significant decrease of enrichment ratio. The low enrichment ratio in the 100% grass cover reduction plot (T100), however, does not necessary mean slow loss of nutrients by aeolian processes. In fact, our data show that the T100 treatment actually experiences greater nutrient loss in dust, and that the increased dust emission at this site easily overcomes the low enrichment ratio on this treatment.

In our experiment, enrichment ratios generally increased with increasing vegetative cover. We believe that this increase is the dual consequence of both nutrient conservation and plant addition in plots with more vegetation. Our results show that large nutrient flux occurred in the plots with less vegetation (Fig. 7). Although we did not measure nutrient additions by plants, West and Skujins (1977) and Coppinger et al. (1991) have suggested that plants contribute nutrients to surface soils through multiple pathways that should lead to more fertile soils on plots with higher plant cover: (1) the uptake of soil nutrients by vegetation and the subsequent deposition of litter; (2) the canopy through fall; and (3) the deposition of windblown materials captured by the plant canopy. The combination of these processes leads to more fertile soils on the plots with higher plant cover.

The C to N ratio further indicates the importance of biotic processes in adding nutrients to surface soils (Table 4). C/N ratios in both surface soil and windblown sediment are low, indicating that (1) the quality of the organic material is high, and (2) the decomposition of this material may occur rapidly. This rapid decomposition may be due to the high temperature, the intense ultraviolet light and the fragmentation by intermittent rainfall in desert ecosystems, especially during the summer (Pauli 1964; Gehrke et al. 1995). C/N ratio was generally higher in the windblown sediment compared to the surface soil, suggesting that light-weight windblown material has a higher proportion of lesser-decomposed organic material than the bulk surface soil.

In terms of ecosystem function, aeolian transport of soil nutrients may be of greater importance than transport of surface soil. Figure 7 shows that the threshold at which nutrient flux increased dramatically occurs at a higher cover than the threshold for sediment flux, most probably between T75 (75% grass cover reduction) and T50 (50% grass cover reduction). Although dust emission from the 100% grass cover reduction plot (T100) is much greater than that of the 75% grass cover reduction plot (T75) (Fig. 6), the concentration of TOC and TN in sediments is higher in T75 than T100. Therefore, flux of TOC and TN, which is the product of vertical flux and TOC or TN content in the windblown dust, shows an increase in T75 relative to T100. This results in a shift of the threshold for nutrient loss to a higher plant cover level. This shift in the threshold for nutrient flux, relative to vertical dust flux, indicates the tight coupling between aeolian transport of nutrients and biotic processes of C and N accumulation in desert soils.

Although soil nutrient loss by aeolian transport has been discussed in many studies, the short lifetime (Table 5) and fast depletion of nutrients (Table 6) such as TOC and TN are striking. Our study shows that TOC and TN loss in the windy season immediately following the grass removal accounted for approximately 60% of the total loss. This observation suggests that it may be particularly important to protect ecosystems from wind erosion immediately after the disturbances such as fire, grazing and anthropogenic activities. Okin et al. (2001) reported about 80% of available N loss at a scraped site in the Jornada Basin in 8 years. They further estimated that the *e*-folding time for N is approximate 5–10 years. Results from our 100% grass cover reduction site (T100) suggest that *e*-folding time for N could be as short as 3.5 years, depending on the intensity of the wind erosion. These results indicate that aeolian processes have the ability to trigger very rapid soil nutrient depletion in ecosystems dominated by wind erosion.

Our results further suggest a threshold between the T50 (50% grass cover reduction) and T25 (25% grass cover reduction) treatments where nutrient change (TOC, TN and NO_2^- + NO_3^-) in the top 5 cm either goes from negative to positive or the decrease drops dramatically. The location of this threshold differs from both the threshold of the horizontal mass flux (between T100 and T75), and the vertical nutrient flux (between T75 and T50).

Nutrient budgets in desert grasslands at a landscape scale have not been effectively addressed. Peterjohn and Schlesinger (1990) studied nitrogen loss from deserts in the southwestern United States, and the authors summarized that N fixation is the major mechanism of N input, and the dominant output mechanisms for N are by wind erosion and volatile loss of N species through decomposition and other biogeochemical processes. About 20% of the input N will be stored in the soil. In a wind-dominated site such as 100% grass cover reduction plot, nutrient additions by plants are limited because of low plant cover. Meanwhile, the very sparse distribution of mesquites in this plot also makes it ineffective at capturing windblown litter and dust from the upwind area. As plant cover increases, soil nutrient loss by wind erosion decreases and plant cover also traps more windblown materials from outside the system. The decomposition of plant litter produces more soil organic matter and increases soil nutrient content. The threshold of TN and TOC loss may be reached when soil nutrient loss by wind is balanced by biotic additions such as N fixation and photosynthetic fixation of C and subsequent decomposition of the dead plant tissues.

Results from this 3-year experiment provide insights into the importance of aeolian processes on plant–soil interactions in wind-erodible desert ecosystems. They provide corroboration of the importance of lateral cover, instead of fractional canopy cover, as a key variable in controlling wind erosion in natural environments. Although this parameter has been used since the work of Marshall in the 1970s (Marshall 1971), few studies examining the utility of lateral cover in natural environments have been conducted (e.g., Wiggs et al. 1994; Lancaster and Baas 1998). Although our experiments show that lateral cover provides a useful estimate of vegetation threshold below which aeolian processes dramatically increase, our results do pose a somewhat puzzling problem of significant flux at even high lateral cover ($\sim$45%), which is not predicted by the dominant model of shear-stress partitioning on vegetated surfaces (Raupach et al. 1993). These results require a new physical explanation that has been provided by Okin (2007).

Our experiments also highlight the tradeoffs between nutrient addition by biotic processes localized at plants and nutrient losses that occur between plant canopies due to wind erosion and dust emission. These results provide experimental evidence for a dual mechanism for the reduction of soil nutrients in areas where vegetation is removed by anthropogenic or natural causes. Specifically, reduced vegetation cover both increases the loss of nutrients by wind and decreases the addition of nutrients to the soil through biotic pathways.

Wind erosion does not simply mean a net loss of all species. Non-limiting species, such as Cl^- and SO_4^{2-} did not show a consistent pattern with the increase of plant cover, suggesting that non-biotic processes such as aeolian deposition, and redistribution of soil nutrients by saltation may play important roles in spatial and temporal distribution of such species.

Conclusions

We present results from a 3-year, landscape-scale wind erosion experiment in a typical desert grassland of southern New Mexico. Our results show that plant lateral cover, which accounts for both number density and vertical dimensions of plants, is a better metric of plant cover than plant fractional cover for the purposes of monitoring wind erodibility in vegetated desert ecosystems. Our data suggest that as lateral cover decreases below about 9% (between the 100% (T100) and 75% (T75) grass cover reduction plots in our experiment), wind erosion and dust flux increase dramatically and that this level of lateral cover may be a key threshold for land managers wishing to control wind erosion.

Although nutrient loss by dust emission was generally greater in plots with lower grass cover, we observed that a threshold for nutrient loss occurs between 50% (T50) and 75% grass cover reduction (T75) plots for most nutrient species. This threshold is at a greater grass cover than the threshold for wind erosion and dust emission. Our results further show that significant nutrient depletion measured in surface soil occurred during the 3-year experimental period. For example, more than 25% of TOC and TN, and as much as 78% of $NO_2^- + NO_3^-$ was lost from the 100% grass cover reduction plot (T100). We also found a third threshold between the 50% (T50) and 25% (T25) grass cover reduction at which total soil nutrient change goes from negative to positive. We suggest that the balance between soil nutrient removal by aeolian transport and biotic addition controls this threshold and that plots with 25% grass cover reduction may still have enough vegetation to counteract nutrient losses by wind. The observed increase of enrichment ratio with increasing vegetation cover further supports our conclusion that biotic processes continually add nutrients to the surface soil that are then removed by aeolian processes. Our results further show that the removal of nutrients from the surface soil can occur very rapidly. For instance, the *e*-folding time for TOC and TN of less than 5 years in the 100% grass cover reduction plot (T100) indicates that wind erosion has the potential to impact soil fertility and ecosystem dynamics in areas of disturbance on very short timescales.

The existence of three distinct thresholds—one for sediment flux, one for nutrient flux, and one for nutrient balance—suggests that land stewards wishing to manage for aeolian processes may need to tailor their management plans for different impacts as well as different timescales. If, for instance, a manager simply wishes to limit the amount of dust emitted from the landscape, the lateral cover threshold of $\sim$9% may be sufficient over the short term as a simple guide. However, if a manager wishes to maintain long-term productivity of the land and avoid initiation of a positive feedback in which reduced nutrient levels lead to further decrease in cover, then our data suggest that loss of 25–50% of vegetation can result in net loss of nutrients in the soil surface, potentially reducing plant establishment rates.

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