Earth Surface Processes and Landforms
Earth Surf. Process. Landforms **28**, 1189–1209 (2003)
Published online in Wiley InterScience (www.interscience.wiley.com). DOI: 10.1002/esp.1034

WIND AND WATER EROSION AND TRANSPORT IN SEMI-ARID SHRUBLAND, GRASSLAND AND FOREST ECOSYSTEMS: QUANTIFYING DOMINANCE OF HORIZONTAL WIND-DRIVEN TRANSPORT

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Received 9 September 2002; Revised 17 April 2003; Accepted 21 April 2003

ABSTRACT

Soil erosion is an important process in dryland ecosystems, yet measurements and comparisons of wind and water erosion within and among such ecosystems are lacking. Here we compare wind erosion and transport field measurements with water erosion and transport from rainfall-simulation for three different semi-arid ecosystems: a shrubland near Carlsbad, New Mexico; a grassland near Denver, Colorado; and a forest near Los Alamos, New Mexico. In addition to comparing erosion loss from an area, we propose a framework for comparing horizontal mass transport of wind- and water-driven materials as a metric for local soil redistribution. Median erosion rates from wind for vertical mass flux measurements (g m^{-2} d^{-1}) were $1{\cdot}5 \times 10^{-2}$ for the shrubland, $8{\cdot}3 \times 10^{-3}$ for the grassland, and $9{\cdot}1 \times 10^{-3}$ for the forest. Wind-driven transport from horizontal mass flux measurements was greatest in the shrubland (15·0 g m^{-2} d^{-1}) followed by the grassland (1·5 g m^{-2} d^{-1}) and the forest sites (0·17 g m^{-2} d^{-1}). Annual projections accounting for longer-term site meteorology suggest that wind erosion exceeds water erosion at the shrubland by *c*. 33 times and by *c*. five times at the forest, but not the grassland site, where the high clay content of the soils contributed to greater amounts of water erosion: water erosion exceeded wind erosion by about three times. Horizontal transport by wind was greater than that by water for all three systems, overwhelmingly so in the shrubland (factor of *c*. 2200). Our results, which include some of the only wind erosion measurements to date for semi-arid grasslands and forests, provide a basis for hypothesizing trends in wind and water erosion among ecosystems, highlight the importance of wind erosion and transport in semi-arid ecosystems, and have implications for land surface geomorphology, contaminant transport, and ecosystem biogeochemistry. Copyright © 2003 John Wiley & Sons, Ltd.

KEY WORDS: wind erosion; water erosion; aeolian sediment transport; drylands semi-arid ecosystems

INTRODUCTION

Soil erosion can have significant adverse impacts on the land and its inhabitants (Toy *et al*., 2002). The redistribution or loss of soil by erosion from land surfaces can dramatically change topography, soil properties, productivity, and can contribute to public health concerns due to long-distance transport of soil-borne contaminants such as radionuclides (Whicker and Schultz, 1982) and hazardous chemicals such as those used in agriculture (Larney *et al*., 1999). Erosion problems are especially pronounced in arid and semi-arid environments where the relatively sparse vegetation cover allows more direct impact of wind and water energies to the soil surface. The relative roles of wind versus water erosion in a given system are important. The relative importance of the two types of erosion is hypothesized to play a central role in vegetation patch structure (Aguiar and Sala, 1999) and to moderate the desertification process of transitioning grasslands to shrublands (Schlesinger *et al*., 1990). Both types of soil erosion could be sensitive to climate change, and the relative importance of one to the other in an ecosystem could be affected (Valentin, 1996; Gregory *et al*., 1999). Similarly, their relative importance

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in different ecosystems affects human and ecosystem risks associated with contaminant transport. Despite the potential importance of both types of erosion and associated transport in dryland environments, our knowledge of the absolute and relative magnitudes of both wind and water erosion is limited.

Our knowledge of wind erosion is primarily limited to agricultural fields and deserts with comparatively little information on wind erosion in other dryland ecosystems such as semi-arid grasslands, shrublands, and forests. Most field measurements of wind erosion are from dryland crop fields (Skidmore, 1994) and have formed the basis for wind erosion models such as the Wind Erosion Equation (Woodruff and Siddoway, 1965), the Wind Erosion Prediction System (Hagen, 1991), and the Revised Wind Erosion Equation (Fryrear *et al*., 1998). The applicability of these cropland-based erosion models for non-agricultural dryland ecosystems is questionable because of fundamental differences in soil texture and vegetation structure that alter the microclimatic factors controlling wind erosion (Bonan, 2002).

Wind erosion is an important process in dryland ecosystems (Ludwig *et al*., 1997; Valentin, 1996) and can be viewed as competing with water erosion (Kirkby, 1980; Baker *et al*., 1995; Valentin, 1996; Gao *et al*., 2002). Field measurements of wind erosion have been made in shrublands (Hennessey *et al*., 1986; Wolfe and Nickling, 1996; Gillette and Chen, 2001; Okin *et al*., 2001; Whicker *et al*., 2002a), but few field measurements exist for semi-arid grasslands and forests. Further, the relative magnitude of wind erosion relative to water erosion probably varies among these different dryland ecosystems, yet measurement-based estimates of the relative magnitudes of wind erosion and water erosion are largely lacking.

Water erosion studies have been more prevalent in dryland ecosystems and have resulted in a more extensive understanding of water erosion compared with wind erosion in different ecosystems (Dunne and Leopold, 1978; Dingman, 1994). A common method of quantifying water erosion has been through the use of rainfall simulators, where known amounts and intensities of precipitation applied can be directly related to the sediment yield from a plot. Although complications occur due to the effects of plot size and uncertainties in extrapolating plot measures to hillslope scales (Wilcox *et al*., 2003), the technique can permit meaningful comparisons among sites when these scale-dependent relationships are considered (e.g. Johansen *et al*., 2001a). Hence, use of a common and widely applied methodology has yielded a relatively robust understanding of water erosion in dryland ecosystems.

An analogous approach for wind erosion to that of rainfall simulation for water erosion is the use of wind tunnels to apply airflows of known velocity and duration to a known area. Although this approach has allowed the quantification of many fundamental relationships (Gillette *et al*., 1980; Nicholson, 1993; Marticorena *et al*., 1997), its application to measures of erosion in natural settings is limited by (1) the inability of wind tunnels to simulate the important factors of turbulence structure and gustiness of wind, and (2) the additional complications that the structure of taller vegetation pose, particularly in shrublands and forests, which have tall woody plants. For these cases, wind tunnels are often impractical because they must be positioned in areas between woody plants, thereby excluding a major determinant of surface roughness for such ecosystems. Therefore, the wind tunnel measurements in these isolated sub-sections of land may not reflect actual erosion rates from the larger, more complexly vegetated area.

Comparisons of wind and water erosion and transport rates are hampered not only by the lack of comparable wind erosion measurements but also – and perhaps primarily – by the differences in measurement methodology. Our overall goal was to evaluate how wind-driven soil erosion and transport varies in three different semi-arid ecosystems – a grassland, a shrubland, and a forest – and contrast wind erosion and transport in each ecosystem with water erosion and transport. Here, we initially propose a conceptual framework for comparing measurements of wind and water erosion and transport, which then allows us to formulate the specific objectives of our study.

Conceptual framework for comparing wind and water erosion and transport

Comparing wind and water erosion should be addressed by considering the aspects that both have in common, in addition to those aspects in which they are fundamentally different (Toy *et al*., 2002). While soil movement through wind and water is driven by different physical forces, in general their processes share three critical phases. The first phase is *detachment* of soil particles from the surface by water or wind movement. The second phase is *transport* of detached particles as either overland flow or aerosol movement. The third phase is *deposition* of these particles as wind and water velocities decline. *Erosion*, which may be defined as a removal of soil particles from the site, is a result of the combined action of these three phases.

Although wind and water erosional processes share these same phases, there are obvious differences in the processes involved. For example, the detachment and transport of soil by wind and water occur on different time scales. Water-driven transport occurs as a sporadic, event-based phenomenon associated with occasional, intense rains (Dingman, 1994). Large-scale, wind-driven detachment and transport also occurs in response to weather events, especially periods of high winds (Godon and Todhunter, 1998; Stout, 2001; Arimoto *et al*., 2002; Whicker *et al*., 2002a). However, some wind-driven events that result in both detachment and transport can be due to short bursts of wind on generally calm days (Stout and Zobeck, 1997) and may be expected to occur much more frequently than intense rainfall events. Some wind-driven detachment and transport may even be expected on almost a daily basis. The movements of detached particles by water and wind also differ in their transport directions. Water transport is a more uni-directional process with the primary direction being downslope and is largely irreversible – a particle removed in one rain event is unlikely to be returned in a subsequent rain event. In contrast, wind transport may occur in any direction in response to changing wind directions, and it is at least partially reversible: a particle dispersed downwind may be returned by a subsequent opposing wind. Wind-driven movement of particles, in contrast to water-driven, movement is two-dimensional. In a coarse sense, wind-driven soil movement can be thought of as having two-directional components as reflected by: (1) mass *vertical transport* (e.g. height >1 m) which suggests long-distance transport of smaller soil particles either into an area (deposition) or removed from the area (net loss); and (2) mass *horizontal transport* which is primarily composed of larger saltating soil particles being transported in a horizontal direction and close to the ground (e.g. height <1 m) and is generally indicative of local soil redistribution (Stout and Zobeck, 1996; Gillette *et al*., 1997).

In the following analysis we compare both the vertical and horizontal components of wind-driven transport to water-driven transport within shrubland, grassland, and forest ecosystem study sites. The comparison of vertical wind transport to water transport is proposed as a base or initial comparison of the relative importance of wind and water erosion. Specifically, we propose that a reasonable comparison of erosion, or soil loss, from an area is to contrast measurements of the wind-driven vertical flux with the sediment yield from the water erosion studies. For this, we assume that measurements of upward vertical fluxes made 1 m above the soil surface imply soil loss from an area, while downward fluxes indicate net deposition in an area. Even though source areas for both types of erosion can be spatially variable (Horst and Weil, 1994; Gillette *et al*., 1996; Baldocchi, 1997; Wilcox *et al*., 2003), vertical flux for wind-driven transport and rainfall simulation measurements both provide useful indices of erosion. The erosion estimates for both wind and water are in grams per square metre over a given period (e.g. daily or yearly). For water erosion the area corresponds directly to ground area, whereas for wind it corresponds to an area above the ground (at a height between 1 m and 3 m in this study). The comparison of horizontal wind transport to water transport is presented as a contrast between the masses of material being mobilized by the differing effects of wind and water forces.

To compare rates of horizontal transport by wind and water, we propose a framework whereby we consider field measures of the mass of material being transported horizontally across the soil surface through a ‘gate’ of fixed width and height (here we use 1 m width by 0·75 m height; Figure 1). This ‘gate’ is proposed as a simple construct whereby differing transport mechanisms, with differing time scales, can be initially compared. For water erosion, transport through the gate is measured using rainfall simulation studies. For wind erosion, transport through the gate is measured by continuously operating, passive dust collectors that measure mass transport as a function of height above the ground surface. The height of the gate, which is not required for water transport, is included to account for the horizontal flux of airborne material slightly above the ground surface. For water erosion, the gate may be considered as a fixed opening oriented perpendicularly to the slope. For wind erosion, the gate may be considered as an opening that rotates in response to the wind direction with the complication that particles moved through the gate in one direction may subsequently be moved through the gate in another direction. There are also differences in the distances that particles may travel by wind and water forces before they reach the gate and once they pass through. We acknowledge the differences in these gate measures, but submit they are reasonable estimates of that mass that is being subjected to potential erosion, redistribution, and loss by these contrasting physical forces.

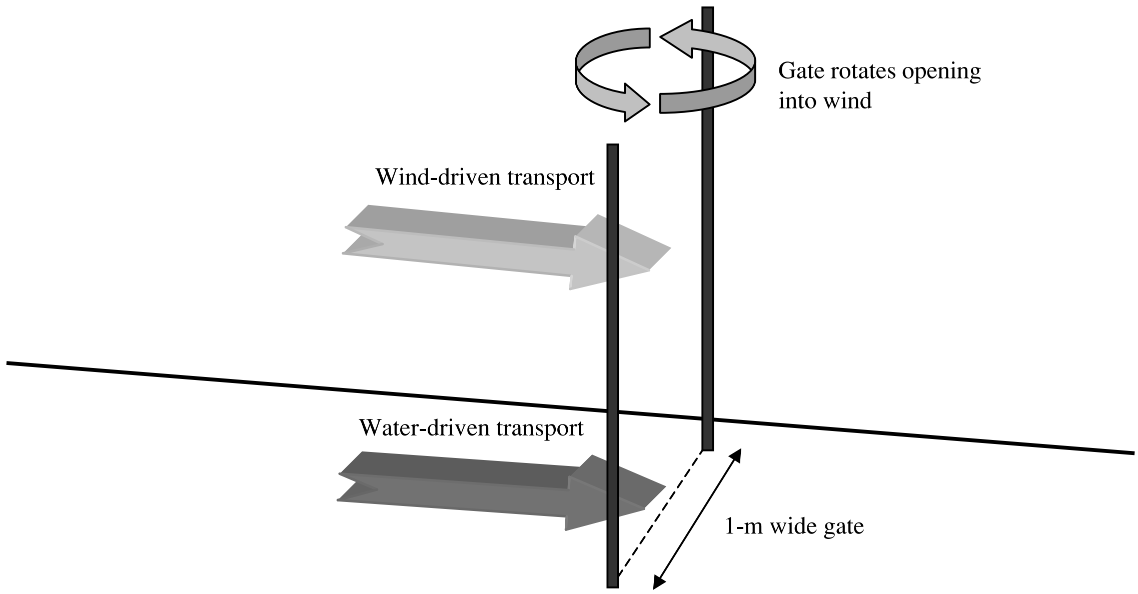


Figure 1. Conceptual comparison of horizontal wind- and water-driven soil transport through a 1 m ‘gate’. For water transport, the gate has a fixed orientation that is perpendicular to the slope. For wind transport, the gate rotates to orient perpendicular to wind direction

Objectives

Based on our conceptualization of the similar phases but contrasting multi-dimensional transports involved in wind and water erosion, and the ‘gate’ concept as a means of comparison, we developed a set of specific objectives for comparing wind and water erosion in semi-arid ecosystems. As noted, our overall goal was to evaluate wind and water erosion in semi-arid shrubland, grassland, and forest ecosystems. The specific objectives of the study were: (1) to measure and compare wind-driven mass flux in both vertical and horizontal directions across shrubland, grassland and forest ecosystems; (2) to compare estimates of annual wind and water erosion rates (i.e. g soil lost per m^2 per year) based on extrapolation from measurements of wind vertical flux and sediment yields from rainfall simulation in each ecosystem; and (3) to compare extrapolated estimates of annual wind- and water-driven horizontal transport, which we define as the horizontal movement of particles across the soil surface, via the ‘gate’ concept (i.e. g soil transported through the 1 m width of the gate per year) in each ecosystem. Our first objective addresses a lack of wind erosion and transport data from semi-arid ecosystems. Our second objective, the comparison of annual estimates of soil loss by runoff and vertical wind flux, addresses the more common definition and assessment of erosion. To address this objective, we draw on previously published studies of water erosion from the same sites as the wind erosion measurements. Our third objective, the comparison of horizontal transport, addresses a separate and potentially unappreciated contrast between wind and water effects. That is, the particles moving in horizontal transport may or may not be subsequently lost from the area but are being redistributed and are at risk of being lost. We will compare measures of horizontal transport by wind and water forces because we believe they are an important measure of potential erosion and soil redistribution by these contrasting processes. The quantity of particles in horizontal transport reflects the magnitude of the previous phase of detachment and removal and also anticipates the subsequent phase of deposition.

We employed only one site within each ecosystem type, and thus our findings provide only an initial comparison across these ecosystem types. Because differences among sites within an ecosystem type can affect our measurements and comparisons, data from other sites are required for a more complete assessment of the relative importance of wind and water erosion. Nonetheless, our results – which suggest that wind erosion and associated transport varies among ecosystem types, with horizontal transport being dominant in all three ecosystems – provide a baseline set of data for establishing the relative importance of wind and water erosion in semi-arid shrublands, grasslands, and forests.

METHODS

Our approach included measurements of wind- and water-driven transport at a semi-arid shrubland, grassland, and forest site, as described in detail below. Wind-driven transport measurements included measures of both vertical and horizontal mass flux; measurements of water-driven sediment transport were yields from rainfall simulation studies. The measured wind- and water-driven mass transport rates were extrapolated to estimates of annual erosion and horizontal transport rates for each of the three ecosystems.

Study sites

Our study included one site each in a shrubland, grassland, and forest. These sites were all semi-arid locations in the western USA where average annual precipitation was less than 500 mm a^{-1}. The sites are associated with US Department of Energy facilities, where wind and water erosion are of concern as potential transport mechanisms for current and/or future releases of radionuclides and other contaminants (Webb *et al*., 1997; Johansen *et al*., 2001a, b; Whicker *et al*., 2002 a, b). The shrubland study site was located near the Waste Isolation Pilot Plant at the Cactus Flats monitoring site (operated by the Carlsbad Environmental Monitoring and Research Center), near Carlsbad, New Mexico (32°13′5″ North and 103°41′45″ West), and was dominated by *Larrea tridentata* (Sesse&Moc. Ex DC.) Coville (creosote). The grassland site was located adjacent to the Rocky Flats Environmental Technology site near Denver, Colorado (39°52′40″ North and 105°11′20″ West) and was shortgrass steppe dominated by *Bouteloua gracilis* (H.B.K.) Lag. (blue grama grass). The forest site was located along the western edge of Los Alamos National Laboratory near Los Alamos New Mexico, and was dominated by the *Pinus ponderosa* Douglas ex P. & C. Lawson var. *scopulorum* Englem (ponderosa pine). The three sites differed in several characteristics related to meteorological conditions, soils, and vegetation (Table I). Measures of wind- and water-driven erosion and transport, within ecosystems, were made on plots of similar slope,

Table I. Site characteristics for each ecosystem type

	Shrubland site		Grassland site		Forest site	
Department of Energy Facility	Waste Isolation Pilot Plant, Carlsbad, New Mexico, USA		Rocky Flats Environmental Technology Site, Aurora, Colorado, USA		Los Alamos National Laboratory, Los Alamos, New Mexico, USA	
Location (latitiude N, longitude W)	32°13′05″, 103°41′42″		39°52′40″, 105°11′20″		35°51′49″, 106°19′13″	
Dominant vegetation	*Larrea tridentata* (creosote)		*Bouteloua gracilis* (blue grama grass)		*Pinus ponderosa* (ponderosa pine)	
Soil texture	Sand		Clay		Silt loam	
Sand (%)	92		34		24	
Silt (%)	4		21		58	
Clay (%)	4		45		18	
Cover: wind (water)						
% Ground cover	66 (80)		79 (72)		98 (52)	
% Woody canopy cover	28		0		75	
Mean woody canopy height (m)	0·75		Not applicable		12	
Annual precipitation (mm a^{-1})	300		370		500	
Wind velocities	Study[a]	Multiyear	Study[a]	Multiyear	Study[ab]	Multiyear[b]
2 h avg (m s^{-1})	2·6	3·2	2·1	3·0	0·81 (2·7)	(2·4)
2 h max. (m s^{-1})	12·1	11·8	8·9	17·8	2·45 (12·5)	(11·3)

[a] Measurement periods were Jun. 1998–Nov. 1998 for the shrubland site, Jun. 1999–Nov. 1999 for the grassland site, and Oct. 2001–May 2002 for the forest site.

[b] Numbers in parentheses are measurements at 12 m from nearby meteorological tower but outside the forest canopy. Numbers not in parentheses are measurements at 1 m height within the forest canopy.

vegetation, and soil at the shrubland and the grassland sties. However, obtaining measurements of water and wind transport at the forest site at Los Alamos required the use of three somewhat different locations within the forest. The plot for measurement of horizontal wind transport at the forest site was in a closed-canopy portion of the forest (35°51′49″ North and 106°21′4″ West). The rainfall simulation equipment required a more open canopy where the rotating boom could be located between trees so a nearby site was selected (35°51′38″ North and 106°19′13″ West). In addition, the vertical transport measurement plot was within 50 m south of the rainfall simulation plot where electrical power supplies were located. The difference in tree and ground cover among these plots was considered in subsequent comparisons of erosion and transport.

Measuring vertical and horizontal wind-driven transport among ecosystems

The first objective of this study was to measure the vertical and horizontal transport of wind-driven material in each of the three ecosystems. Consistent methods for measurements of both vertical and horizontal mass fluxes were made at each site allowing comparisons of wind-driven transport across the ecosystems.

Vertical transport. To estimate the mass of material leaving from or depositing in the study sites, we measured the vertical mass flux. Vertical transport or vertical flux was measured using the vertical flux gradient method as outlined in Stull (1988) and applied as described in Whicker *et al.* (2002a). This method requires measurements of the vertical concentration gradient of aerosol mass ($d\chi/dz$), which is then multiplied by the eddy diffusivity coefficient (K_z). K_z was calculated as the product of the von Karman constant (taken to be 0·4), the mid-height of the measurements (2 m), and the friction velocity (u_*, in units of m s^{-1}). At each of the sampling sites measurements of the vertical mass concentration gradient was made using total suspended particulate (TSP) samplers placed at 1 m and 3 m above the ground. The filters from these samplers were collected (generally weekly), dried, and weighed to determine ($d\chi/dz$). The friction velocities were calculated from high-frequency, three-dimensional measurements of the wind velocities using factory-calibrated sonic anemometers (Campbell Scientific, Logan UT) and calculated as described in Stull (1988). The median and ranges for vertical fluxes (g m^{-2} d^{-1}) were determined within each ecosystem. These fluxes were then statistically compared across ecosystems using the non-parametric Mann–Whitney U test (StatSoft, 1994).

Horizontal transport. We measured horizontal flux using passive dust samplers that use a tail fin to orient into the wind (Figure 2). These samplers, which have been used in previous studies of wind transport (Gallegos, 1978; Whicker *et al*., 2002a), collect airborne dust from the ground level (for surface creep) up to 75 cm through five 1 cm wide by 15 cm tall slots, or a total sampling area of 75 cm^2 ($7·5 \times 10^{-3}$ m^2). Three replicate samplers were at each site and were placed at the vertices of an equilateral triangle whose corners were separated by either 10 m (forest) or 20 m (grassland and shrubland) depending on vegetation structure characteristics. At the shrubland site the samplers were monitored from June 1998 to November 1998 (around 6 months). At the grassland site, the samplers were monitored from June 1999 to November 1999 (around 6 months), and at the forest site the samplers were monitored from October 2001 to May 2002 (around 8 months). Generally, samplers were not monitored during periods of snow cover.

The airborne dust samples were collected at intervals ranging from weekly to monthly, based on recent wind conditions and labour availability. The samples were screened to remove large debris and insects, oven-dried (generally at 100 °C for >24 hours and in some cases at 50 °C for least 72 hours; these different temperatures affected median weight by less than 1 per cent), and weighed. We expressed measurements from each sampling interval as a mass collected per day (g d^{-1}), and the horizontal flux was calculated by dividing this value by the total sampling area of $7·5 \times 10^{-3}$ m^2. The median and ranges for horizontal fluxes (g m^{-2} d^{-1}) were determined within each ecosystem. We compared average sampling rate across the replicate samplers ($n = 3$) and the different sampling periods. These fluxes were then statistically compared across ecosystems using the non-parametric Mann–Whitney U test (StatSoft, 1994).

To allow measures of horizontal transport from the passive dust collectors used in this study (Figure 2) to be compared with other published studies, we compared the collection rates (mass per day) of the passive samplers (at a height of 0·5 m) with collection rates of the commonly used Big Spring Number Eight (BSNE) sampler (also at a height of 0·5 m), for which collection efficiency has been characterized (Fryrear, 1986; Shao *et al*., 1993; Goossens and Offer, 2000). Our objective here was not to develop a specific calibration for our passive dust collectors, but rather to determine if there were substantial differences in collection efficiency from the

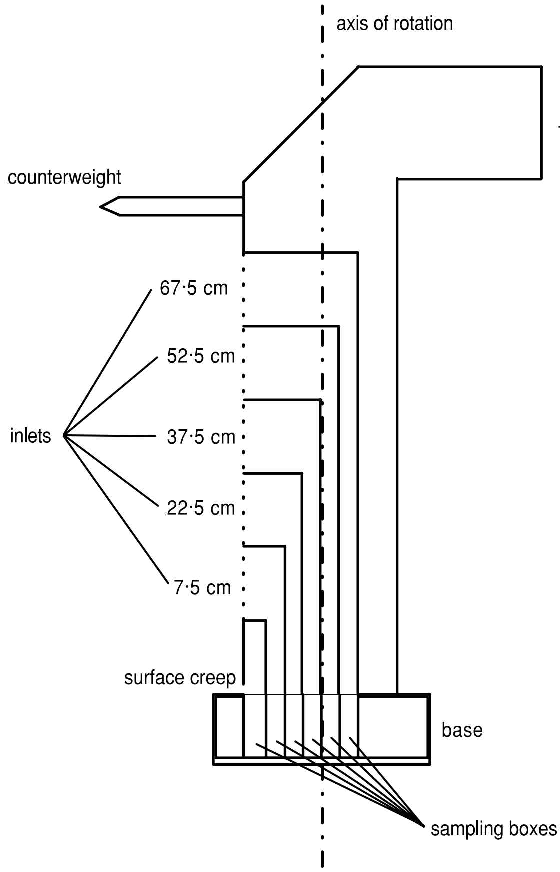


Figure 2. Schematic of passive dust collector used in measurements of wind-driven horizontal flux (reprinted from Whicker *et al*., 2002a with permission from the America Society of Agronomy, the Crop Science Society of America, and the Soil Science Society of America)

BSNE samplers. Although comparisons between collection rates of the two sampler types were made, a correction for collection efficiency was precluded because we could not measure particle size or meteorological conditions in real-time during the measurement intervals. We used measurements from a Los Alamos location that was intermediate in wind and soil texture conditions relative to the three ecosystem sites and was adjacent to a long-term meteorological station so that additional wind data could be evaluated. This comparison was conducted over an 11-month period at Los Alamos National Laboratory at a site adjacent to the water erosion plots (TA-6 meteorological station). The ground cover and soil texture were similar to that for the water erosion plots at Los Alamos. The passive dust collectors that we used did not differ significantly in collection rates from that of the BSNE samplers. The average ratio of mass collection rate in the BSNE sampler to the mass collection rate for the passive dust collector over the 11-month study was $1{\cdot}08 \pm 0{\cdot}62$ ($n = 12$ sampling intervals). Although not a complete test for all sites, these findings suggest that the sampling efficiency of the passive dust samplers that we used was similar to that of the more commonly used BSNEs. Our samplers, like the BSNE samplers, do not have complete collection efficiency and hence our estimates of horizontal wind transport are expected to be conservatively low.

Estimation of annual wind and water erosion rates

To address our second objective of comparing and contrasting annual rates of wind and water erosion in each of the three ecosystems, we drew on previously published studies of water erosion and developed a method for extrapolating both wind and water erosion data to annual rates in a consistent manner. A simple means of projecting the wind erosion rates from our sampling period, which corresponded to 6–8 month intervals, is to multiply the rates that we report on a daily basis by the number of days in a year. This approach is useful, but there are three issues associated with it. First, this form of projection assumes that wind rates over the measurement periods are similar to those for the entire year. Second, if we are interested in estimating a long-term rate for a site, it assumes that the year in which the measurements were obtained is representative of longer-term climate trends. Third, our water erosion data from rainfall simulation are tied to an event of specified precipitation intensity and duration, and cannot be projected in a similar manner. Hence, we have used a detailed projection method to address the issues above.

To extrapolate wind and water erosion to annual rates in a consistent manner, both between the two processes and across the three sites, we based our projection on erosion-driving events. This approach has three steps. First, we defined a threshold at which an erosion event occurs. For wind erosion this threshold is based on wind velocity, whereas for water erosion it is based on soil infiltration rate corresponding to a given rainfall intensity. Second, we used our measurements to provide an estimate of the amount of erosion or transport per event. Third, we used multi-year meteorological data to estimate a mean number of erosion-driving events per year. Using the product of the amount of erosion per event and the number of events per year, we estimated annual erosion rates. Although there are limitations in this approach, it is the most consistent approach given limitations associated with lack of long-term, high resolution data at some of the study sites and the differences between wind and water erosion measurement methods. We conducted these extrapolations for erosion (Table II) and horizontal transport (Table III).

Annual wind erosion extrapolation. Our extrapolation of wind erosion to an annual basis required use of several different data sets on wind velocity. Differences among these data sets constrained our extrapolation approach. During the study period, wind velocities and directions were monitored at each of the study sites. For the grassland and shrubland sites, the wind measures were made at 3 m above the ground surface, which is above the vegetation. For the forest site, wind data were collected at 12 m above the ground surface outside the forest canopy at the Los Alamos National Laboratory's TA-6 meteorological station. This station is approximately 2 km east of the forest collection site where horizontal flux measurements were obtained. Additional data on wind velocities and directions were obtained from 1 m above the ground surface within the forest site where horizontal flux was studied. Wind velocity data were obtained as 2 hour averages using meteorological weather stations from either Davis Instruments (Hayward CA) or Campbell Scientific Instruments (Logan UT).

Longer-term climate data from each of the sites was determined from multi-year wind velocity data obtained from monitoring stations within 2 km of each of our wind erosion study plots. To maintain consistency among sites, we were limited to three years of data that had high-resolution 2 hour averaged wind velocities. For the shrubland site, we used available 10 m wind velocity data from 1997–1999 collected by Carlsbad Environmental Monitoring and Research Center (T. B. Kirchner, personal communication); for the grassland site, we used 10 m wind velocity data from 1991–1993 collected at the Rocky Flats Environmental Technology Site for use in a risk assessment study (Rood *et al*., 2002), and for the forest site we used 12 m wind velocity data from 1997–1999 collected by Los Alamos National Laboratory (LANL Web Site, 2003). For the shrubland and grassland sites, we normalized these multi-year wind velocity data to a height of 3 m, the height for which study measurements were obtained, assuming a lognormal wind profile with height and measured roughness lengths (Stull, 1988). Site-specific roughness lengths (medians of 12 cm at the shrubland and 5 cm at the grassland) were derived from friction velocities based on wind velocity measurements that we made with a sonic anemometer (Whicker *et al*., 2002a; J. J. Whicker and D. D. Breshears, unpublished data).

For our event-based extrapolation, a wind erosion event was assumed to occur when the wind velocities exceed a threshold velocity (Table II). Measurements of threshold velocities were made by simultaneously monitoring aerosol concentrations and wind velocities at two of the sites: at the shrubland site based on 1 minute measurements at a height of 3 m, and at the forest site based on 30 minute measurements at 12 m (the height differences are appropriate given plant height differences; we were constrained to 30 minute data at the forest due to existing data collection methods). The threshold velocity for wind erosion in both of these systems was estimated to be about 6–7 m s^{-1} (Whicker *et al*., 2002a, b). We were not able to make threshold velocity measurements at the grassland site, therefore we assumed that the threshold at this site was the same as the other sites. Therefore, the threshold value of 6 m s^{-1} was selected for all three sites, corresponding to a 3 m height for shrubland and grassland sites and a 12 m height for the forest site. We recognize that these threshold values are

Table II. Data used for annual erosion projections for wind and water among each ecosystem type. Projected annual erosion mass flux for wind erosion is the product of the mass flux per wind event and the mean number of wind events per year, and for water erosion is the product of the sediment yield and the average rainfall per year greater than the infiltration rate. Ratios of wind to water erosion for each of the sites are 32·5 at the shrubland, 0·3 at the grassland, and 9·0 at the forest

	Erosion rate measurements	Erosion thresholds	Frequency of erosion events and erosion rates per event			Projected annual erosion mass fluxes
Wind	Median mass vertical flux[a] ($g\ m^{-2}\ d^{-1}$)	Wind velocity threshold ($m\ s^{-1}$)	Measured rate of wind events >6 $m\ s^{-1}$ during study[a] (events d^{-1})	Mean number of events >6 $m\ s^{-1}$ from multi-year data (events a^{-1})	Median mass flux per wind event ($g\ m^{-2}\ event^{-1}$)	Projected annual mass erosion flux ($g\ m^{-2}\ a^{-1}$)
Shrubland	$1·5 \times 10^{-2}$	6	0·4	366	$3·9 \times 10^{-2}$	14·3
Grassland	$8·3 \times 10^{-3}$	6	0·5	290	$1·6 \times 10^{-2}$	4·5
Forest	$9·1 \times 10^{-3}$	6	0·2	90	$4·4 \times 10^{-2}$	4·0
Water	Sediment yield ($g\ m^{-2}\ mm^{-1}$)	Infiltration rate ($mm\ h^{-1}$)		Average rainfall per year > infiltration rate ($mm\ a^{-1}$)		Projected annual mass erosion flux ($g\ m^{-2}\ a^{-1}$)
Shrubland	0·027	58		16		0·44
Grassland	0·19	15		78		15·0
Forest	0·33	35		36		0·83[b] (12 unadjusted)

[a] Sampling period for vertical flux measurements at the forest site was from Aug. 2000 to Jun. 2002.
[b] Adjusted for large percentage difference in percentage ground cover between open and closed forest (cover factor = 0·07).

Table III. Data used for annual projections of horizontal mass transport through a conceptual 1 m wide gate for wind and water among each ecosystem type. Projected annual horizontal transport mass flux for wind is the product of the mass flux per wind event, the height of the sampler (0·75 m), and the mean number of wind events per year. For water erosion, the annual projection is the product of the sediment yield, plot area (32·4 m) divided by plot width (3·03 m), the length of the plot (10·7 m), and the average rainfall per year greater than the infiltration rate. Ratios of annual wind to water horizontal mass transport for each of the sites are 2200 at the shrubland, 3·8 at the grassland, and 2·0 at the forest

	Mass transport measurements	Erosion thresholds	Frequency of erosion events and erosion rates per event			Projected annual horizontal transport mass fluxes
Wind	Median mass horizontal flux ($g\ m^{-2}\ d^{-1}$)	Wind velocity threshold ($m\ s^{-1}$)	Measured rate of wind events >6 $m\ s^{-1}$ during study (events d^{-1})	Mean number of events >6 $m\ s^{-1}$ from multi-year data (events a^{-1})	Median mass flux per wind event ($g\ m^{-2}\ event^{-1}$)	Projected annual mass flux passing through 1 m wide and 0·75 m high gate ($g\ m^{-1}\ a^{-1}$)
Shrubland	15·0	6	0·39	366	38	$1{\cdot}0 \times 10^4$
Grassland	1·5	6	0·53	290	2·8	610
Forest	0·17	6	0·65	90	0·27	18
Water	Sediment yield ($g\ m^{-2}\ mm^{-1}$)	Infiltration rate ($mm\ h^{-1}$)		Average rainfall per year > infiltration rate ($mm\ a^{-1}$)		Projected annual mass flux passing through 1 m wide gate ($g\ m^{-1}\ a^{-1}$)
Shrubland	0·027	58		16		4·7
Grassland	0·19	15		78		160
Forest	0·33	35		36		8·8[a] (126 unadjusted)

[a] Adjusted value for large difference in percentage ground cover between open and closed forest (cover factor = 0·07).

rough approximations of annual averages, that the values are subject to soil and vegetation conditions that vary throughout the year (Selah and Fryrear, 1995; Marticorena *et al*., 1997; Stout and Zobeck, 1996; Stout, 2001), and that threshold determination is subject to the averaging times over which the measurements are taken (Stout, 1998). However, the selected threshold values seem reasonable based on other published field studies in a variety of ecosystems which report threshold values in the range of 4 to 8 m s^{-1} (Helgren and Prospero, 1987; Godon Todhunter, 1998; Stout, 2001; Arimoto *et al*., 2002; Whicker *et al*., 2002a, b). Long-term wind velocity data were limited at highest temporal resolution to 2 hour means, and hence we defined a wind erosion event as having a 2 hour mean ≥6 m s^{-1}. Our results about the relative rank of wind erosion among the three sites or the relative importance of wind versus water erosion within a given ecosystem is not highly sensitive to our definition of a wind erosion event: estimates using threshold values of 4 and 8 m s^{-1} yielded the same trends as those for 6 m s^{-1}.

Equation 1 shows the formula for the event-based estimates of annual horizontal and vertical fluxes. For this, we first calculated the flux per event by dividing the rate of erosion (RE) from wind events that occurred during the entire sampling period (RE_{study}, units of events per day, where an event has a 2 hour average >6 m s^{-1}) into the median fluxes measured during the study (F_{study}, units of g m^{-2} d^{-1}), where F_{study} can be either the horizontal or vertical flux medians. The annual flux estimate (F_{year}, units of g m^{-2} a^{-1}) was then calculated as the product of the flux on a per-event basis multiplied by the average rate of events per year from the longer-term meteorological data sets ($\overline{RE}_{\text{year}}$, units of events per year).

$$F_{year} = \frac{F_{study}}{RE_{study}} \times \overline{RE}_{year} \tag{1}$$

Annual water erosion extrapolation. Measures of water transport and erosion were based on the results of previously published rainfall simulation studies conducted at each of the three study sites (grassland and shrubland: Johansen *et al*., 2001a; forest: Johansen *et al*., 2001b). A rotating boom rainfall simulator was used to apply simulated rainfall at a rate of 60 mm h^{-1}, a relatively high rainfall intensity, onto 3·03 m × 10·7 m bordered plots located on slopes ranging from 4 to 10 per cent. There were three replicate simulation plots at the grassland site, three at the shrubland site and two at the forest site. For each site, three sequential rainfall simulations were conducted: an initial *c*. 60 min simulation referred to as 'dry', a second *c*. 30 min simulation referred to as 'wet', and a final *c*. 30 min simulation referred to as 'very wet'. By knowing the amount of rainfall applied and measuring runoff from the plot, the soil infiltration can be estimated.

Our estimate of a threshold for water erosion is based on the site-specific estimates of soil infiltration rate. This value varies among the dry, wet, and very wet runs. We selected the wet runs for our extrapolation. We believe that the high soil moisture conditions associated with the very wet runs occur infrequently and hence are unrealistic for extrapolating annually. In using the wet runs instead of the dry runs, our estimates of water erosion were biased upwards slightly: 11 per cent for the grassland, 9 per cent for the forest, and 2 per cent for the shrubland.

Estimates of annual transport by water erosion were extrapolated from rainfall simulation results for each system by: (1) assuming no transport occurred when natural rainfall rates (i.e. mm h^{-1}) were less than the measured soil infiltration rate (Table II); (2) estimating the amount of time per year when rainfall rates exceeded infiltration rates; and (3) extrapolating the annual rate of transport from the measured rate during simulation studies and the time per year when rainfall rate exceeded infiltration rate. This approach was selected as most analogous to that for extrapolating wind erosion. The perhaps more typical procedure of using rainfall simulation data to parameterize a model such as WEPP and using the model to predict rates was precluded because of our objective of using parallel approaches for both wind and water erosion.

A correction was necessary for the forest site to account for large differences in tree and ground cover between the water and wind plots. The forest wind erosion site had *c*. 98 per cent ground cover, whereas the water erosion plots had 42–62 per cent cover. A difference of this amount in ground cover is known to have a large effect on runoff and water erosion and can be corrected for by using the cover factor of the Universal Soil Loss Equation (Wischmeier and Smith, 1978; Nyhan and Lane, 1986). Applying this correction to the water erosion measurements reduces the water erosion by a factor of 0·07. We used this adjusted value of water erosion in our extrapolations. We estimated the amount of time that precipitation exceeded the infiltration rate using results from 'intensity–duration–frequency' analysis (Dunne and Leopold, 1978; Dingman, 1994; Bonan, 2002). We obtained intensity–duration–frequency relationships from NOAA weather stations at Denver, Colorado, for the grassland site, Roswell, New Mexico, for the shrubland site, and Sante Fe, New Mexico, for the forest site (US Weather Bureau, 1955). We verified that the Santa Fe curve was similar to a longer-term, more detailed analysis for Los Alamos (McLin, 1992), but used the Santa Fe data for the extrapolation so that the level of detail was similar for all three study sites. We used curves for 2-year storm frequencies, which are most appropriate for developing estimates of annual averages. To extrapolate the measured water transport to an annual transport estimate, we multiplied the measured rates by the amount of time that precipitation intensity exceeded infiltration rate.

Horizontal transport rates for wind and water erosion

Our third objective was to contrast the mass of material being horizontally transported by wind and water through the conceptual gate, as proposed above. This projection was based on erosion-driving events in the same manner as described above. We assumed that the threshold values for wind and water erosion described in the previous section also applied to horizontal transport. The number of events at each site was also the same as above, being derived from the multi-year meteorological data. Hence, the only difference for these extrapolations was estimates of transport rates per event (Table III).

To calculate the horizontal transport rates for wind, the amount of material collected at each of the sampling heights was summed up to the top of the sampler (0·75 m) and divided by the width of the sampler opening (0·01 m) and the sampling time interval (units of g m^{-1} d^{-1}). To keep the projections of annual transport rates as data-based as possible, we chose not to extend the height of the gate beyond the height of the sampler as is often done, as this requires a model to extend transport rate predictions at heights beyond the measurement data, and the greatest horizontal mass fluxes generally occur below 1 m (Stout and Zobeck, 1996; Gillette *et al*., 1997). This procedure resulted in lower wind transport rates compared to values based on gates with increased heights. To evaluate the effect of this assumption, we estimated the potential amount of transport above 0·75 m up to 1 m (the assumed height for the start of suspended particles to be available for long-distance transport; Gillette *et al*., 1997) by fitting single-component exponential functions to the horizontal flux (units of kg a^{-1} m^{-2}) as a function of sampling height (m; $r^2 = 0{\cdot}86$ for the shrubland, 0·79 for the grassland, and 0·33 and for the forest). We then integrated the functions from 0 m to 0·75 m and from 0 m to 1 m and compared the values. Across the three sites, the average amount of transport between 0·75 m and 1 m was less than 10 per cent of the amount from the ground surface to 0·75 m. We did not adjust our results by this amount, but rather used this value as an indicator of the degree to which our projected wind transport rates are conservatively low compared to an infinitely tall gate.

To estimate horizontal transport rates for water, we divided the sediment yield from the plot by the length of downslope edge the plot (3·03 m). In these calculations we also corrected data from the forest site for differences in cover between wind and water locations, as described above.

RESULTS

Measured vertical and horizontal wind-driven transport among ecosystems

Vertical flux appeared to be greater in the shrubland than the grassland or forest: the median vertical flux (g m^{-2} d^{-1}) across sampling periods was $1{\cdot}5 \times 10^{-2}$ for the shrubland site, $8{\cdot}3 \times 10^{-3}$ for the grassland site, and $9{\cdot}1 \times 10^{-3}$ for the forest site. However, due to the large temporal variation between sampling periods, median vertical flux did not differ significantly among the three sites ($p > 0{\cdot}05$; Figure 3a). The median fluxes for all three sites were positive, indicating upward fluxes corresponding with site losses, as opposed to downward fluxes and site gains, indicated by negative values. As vertical flux was measured at only one location at each site, we are unable to evaluate within-site spatial variation for vertical flux.

In contrast to vertical fluxes, temporal site-to-site variation in the horizontal flux rates did differ significantly among the three ecosystems ($p < 0{\cdot}01$; Figure 3b), with median values (g m^{-2} d^{-1}) greatest in the shrubland

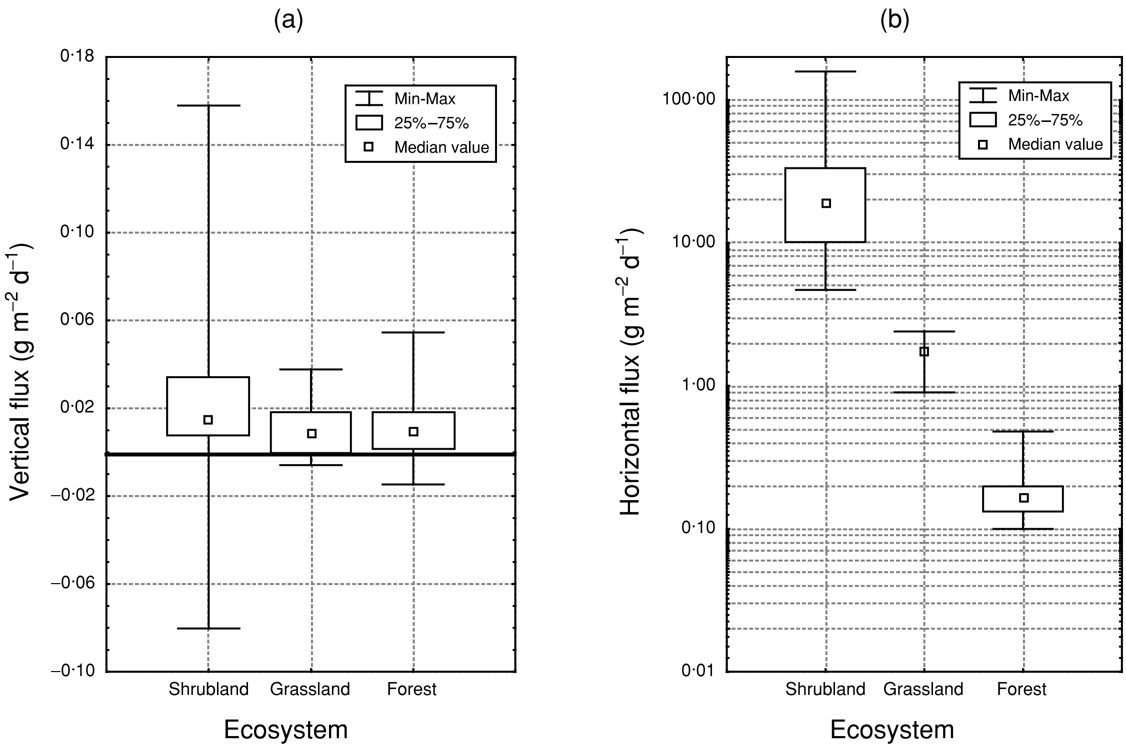


Figure 3. Distribution of wind-driven vertical (a) and horizontal (b) flux reflecting temporal variation across sampling periods in each ecosystem type

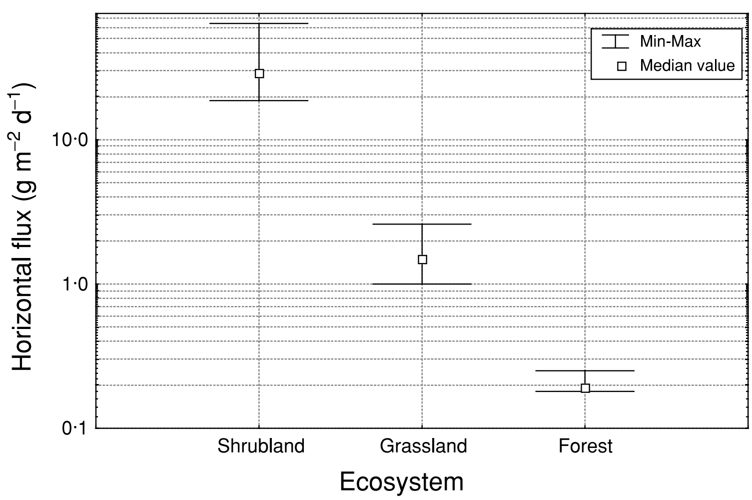


Figure 4. Distributions of wind-driven horizontal flux reflecting only spatial variation across samplers ($n = 3$) in each ecosystem type

(15·0), intermediate in the grassland (1·5), and least in the forest ($1{\cdot}7 \times 10^{-1}$). Within each of the three ecosystems, horizontal flux exceeded vertical flux. The measurements also were evaluated with respect to spatial variation within each ecosystem, by averaging horizontal flux across sampling periods for each sampler ($n = 3$ per site). This analysis also quantified significant differences in horizontal fluxes among ecosystems ($p < 0{\cdot}05$; Figure 4), and indicated that spatial variation within each ecosystems was small compared to variation among ecosystems independent of temporal variation.

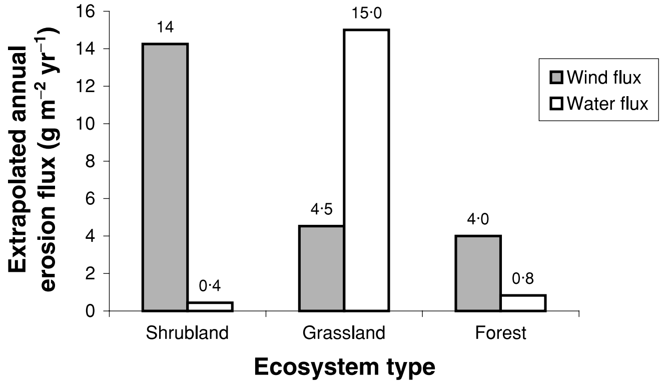


Figure 5. Annual erosion flux for wind and water among the ecosystem types

For comparative purposes, we report published values of water erosion (Johansen *et al.*, 2001a, b) that will be used in the extrapolations below. Water erosion measured as sediment yield from rainfall simulation, expressed on a per-rainfall amount basis (g m^{-2} mm^{-1}), exhibited a different trend across ecosystems from that for wind erosion, with sediment yields of $2{\cdot}7 \times 10^{-2}$ for the shrubland site, 0·19 for the grassland site, and 0·33 for the forest site.

Contrasts of projected annual wind and water erosion flux within ecosystems

For both wind and water erosion, estimated median annual rates varied among ecosystems, as did the relative magnitude of wind to water erosion (Figure 5). Median annual wind erosion flux (g m^{-2} a^{-1}), as measured by the vertical mass flux, was estimated to be 14·3 for the shrubland, 4·5 for the grassland, and 4·0 for the forest.

Annual water erosion rates (g m^{-2} a^{-1}) were 0·44 for the shrubland, 15·0 for the grassland, and 0·83 for the forest (after adjusting the forest estimate down by a factor of 0·07 to correct for ground cover; Figure 5). Therefore, when water erosion differences across sites were converted from a per-rainfall amount basis to an annual basis and differences in forest ground cover were accounted for, water erosion for the grassland exceeded that in either the shrubland or the forest by an order of magnitude or more.

Our projections of annual erosion flux (as estimated by the vertical flux for wind and the sediment yield from the rainfall simulation plots) suggest that wind erosion flux is greater than water erosion at the shrubland by a factor of *c*. 33 and at the forest by a factor of *c*. 5. In contrast, the median water erosion flux at the grassland site is greater than that for wind erosion flux by a factor of *c*. 3.

Contrast of annual wind and water horizontal transport within ecosystems

For both wind and water horizontal transport, corresponding to the 'gate' concept outlined above, the estimated annual rates (g m^{-1} a^{-1}) varied by orders of magnitude among ecosystems, as did the relative magnitude of wind to water erosion (Figure 6). Annual horizontal wind transport (g m^{-1} a^{-1}) was greatest at the shrubland ($1{\cdot}0 \times 10^{4}$),

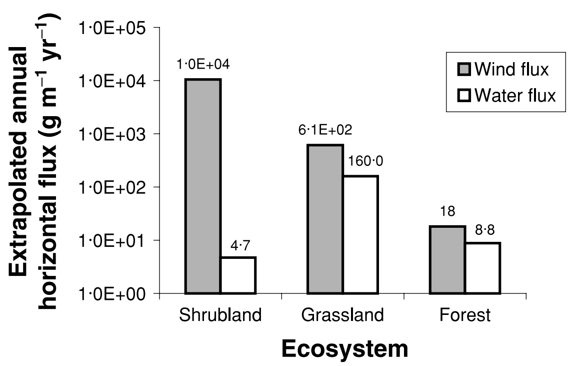


Figure 6. Annual horizontal flux for wind and water among the ecosystem types

intermediate at the grassland (610), and least at the forest (18). Annual horizontal water transport (g m^{-1} a^{-1}) was greatest in the grassland (160), intermediate in the forest (8·8), and least in the shrubland (4·7; note that this trend is the same as for erosion flux [g m^{-2} a^{-1}] because the conversion is a linear one). Notably, at all three sites, wind-driven mass flux through the gate was greater than water-driven mass flux, the ratio of wind to water flux through the 1 m gate varying greatly among the sites: *c*. 2200 for the shrubland, *c*. 4 for the grassland, and *c*. 2 for the forest. Hence, these results suggest that horizontal wind-driven transport process dominates over water-driven transport for local redistribution of soil at all three sites, most dramatically at the shrubland site.

DISCUSSION

Comparison of wind and water erosion and transport among ecosystems

Our results indicate that there are large, important differences in wind erosion and associated transport among the three ecosystems studied. Wind erosion rates are highly variable temporally (e.g. Whicker *et al*., 2002a) and hence, for any given sampling period wind erosion may not be greater in one ecosystem than another (Figure 3a). We observed periods of both soil gain (reflected as negative values) and soil loss (reflected as positive values) at each of the three sites over the measurement period at each (Figure 3a). When extrapolated to an annual basis, we estimate that the vertical fluxes associated with wind erosion are greatest in the shrubland and lower in the forest and grassland (Figure 5).

The differences among ecosystems in horizontal fluxes (Figure 3b) were much greater than those for vertical fluxes (Figure 3a). Differences in horizontal flux were significantly different even given the large temporal variation among sampling periods. Notably, the shrubland had the greatest fluxes in both the vertical and horizontal dimensions. This is consistent with the viewpoint that fluxes in the horizontal dimension are relevant to understanding vertical fluxes. The large horizontal flux at the shubland probably contributes to the vertical flux, as well as to redistribution of particles between shrub canopy patches and intercanopy patches.

A strength of our study is the ability to pair measurements and extrapolations of wind erosion and transport with estimates of water erosion and transport at each of the three sites. We found that water erosion was exceeded by wind erosion in both the shrubland and the forest but that water erosion exceeded wind erosion in the grassland by a factor of *c*. 3. In contrast, when we evaluated horizontal fluxes using the ‘gate’ concept (Figure 1), horizontal wind transport was dominant over water transport in all three ecosystems: by *c*. 2200 at the shrubland, *c*. 4 at the grassland, and *c*. 2 at the forest (Table III).

The differences that we report in our extrapolations are confounded by two factors, one of which we were able to correct for and a second of which we were not. At the forest ecosystem, the location where the horizontal fluxes were measured had greater ground cover than the locations for the vertical flux measurements (which was tied to a permanent meteorological site) and for the rainfall simulation measurements (which was constrained to an area between trees sufficiently large for two side-by-side plots). Using the Universal Soil Loss Equation (Wischmeier and Smith, 1978), we were able to make a simple correction for the water erosion, based solely on the differences in ground cover at the two sites. This correction was essential for a reasonable comparison as it reduced the water erosion estimates to a factor of 0·07 of the measured value. We were unable to apply a similar correction to the vertical fluxes for wind. However, we suspect that our estimates of vertical flux in the forest would be substantially lower (and possibly negative or in a downward direction) if measured at the same location as the horizontal flux. Additional data on horizontal flux at the location where vertical flux was measured suggest that horizontal flux may be several times greater there than at the location where horizontal flux was measured for the study. Hence, we believe that the difference we report between vertical flux and horizontal flux for the forest (Figure 3) is smaller than it would be if ground cover and canopy characteristics were both held constant.

Our results, which represent one of the few, if not only, data-based estimates to date of wind versus water erosion and transport in dryland ecosystems, indicate dominance of horizontal wind-driven transport over horizontal water-driven transport in all three ecosystems studied, and suggest dominance of wind erosion over water erosion in two of the ecosystems. The differences in magnitude of wind and water erosion and transport, both among and within the three ecosystems, result potentially from several factors. These factors include: site climate drivers (precipitation and wind velocity distributions); surface characteristics, including height and density of woody vegetation, amount of ground cover, and patchiness of ground cover; and soil properties, such as soil moisture distribution and particularly soil texture. Our three study sites varied with respect to these factors (Table I). The soil texture differences may be particularly important to consider in explaining the dominance of water erosion over wind erosion at the grassland sites (Figure 5). At the grassland, the soil was a clay (Table I), which probably resulted in the low infiltration rates at this site, even though there was a high amount of ground cover. Relative to the forest, the grassland site produced less water erosion per millimetre of precipitation in the rainfall simulations (Table II), but when extrapolated to an annual basis there was more precipitation exceeding the infiltration rate at the grassland site (78 mm) than at the forest site (36 mm; Table II), and hence water erosion on an annul basis is greater than wind erosion at the grassland site.

Differences in soil conditions, in concert with vegetation amount and pattern, are probably important in determining the differences in wind erosion and transport among ecosystems. The shrubland vegetation is highly patchy, with a mosaic of shrub canopy patches and intercanopy patches that are relatively bare (e.g. 28 per cent shrub cover; Whicker *et al*., 2002a), a pattern frequently observed in degraded shrublands (Pickett and White, 1985; Schlesinger *et al*., 1990; Aguiar and Sala, 1999; Okin and Gillette, 2001). This pattern contrasts with that in the grassland site, where bare patches are much smaller (Hook and Burke, 1991; Schlesinger *et al*., 1996). This difference in the average length of bare patches may help explain how a small difference in ground cover could produce the large difference in wind erosion observed between the shrubland and grassland site. In addition, the large amount of ground cover at the forest site, in conjunction with the reduced near-surface wind velocities resulting from the high density of tree canopies, largely explain the greatly reduced wind erosion within the forest (Table I). Indeed, given the high ground cover at the forest site, it is likely that the small amounts of vertical and horizontal transport measured correspond to airborne particles transported over larger distances. Our estimates of wind-driven horizontal mass flux rates compare reasonably well with literature values for other dryland ecosystems (Gallegos, 1978). Our estimates are much lower than those found during high-wind events in agricultural fields (Stout and Zobeck, 1996; Buschiazzo *et al*., 1999) and in dry lake playas (Gillette *et al*., 1997), which have much lower amounts of ground cover. For both wind and water erosion and transport, differences in climate among the sites probably produce different soil moisture conditions, which, in turn, could affect erosion and transport. We are unable to evaluate this factor in our study. However, all three sites are semi-arid and therefore likely to have dry soil surfaces for much of the year.

Applications and implications

An improved understanding of wind erosion and its role relative to water erosion is needed to address a diverse set of scientific and land management issues. Water and wind erosion are fundamental geomorphic forces, so better quantification of their relative magnitudes across a variety of dryland systems is required to better understand landscape evolution. Wind erosion is thought to be integrally linked to the desertification process where dryland grasslands are converted to shrublands (Schlesinger *et al*., 1990; Havstad *et al*., 2000). There can be both large net soil loss from shrubland systems over long periods (Gibbens *et al*., 1983; Hennessy *et al*., 1986) and redistribution of soil between intercanopy and canopy patches (Havstad *et al*., 2000; Whicker *et al*., 2002a), both of which affect distributions and cycling of carbon and nutrients. Hence, erosional issues need to be considered for effective land management (Miles and McTainish, 1994). Similarly, contaminant transport is of concern for many dryland ecosystems such as US Department of Energy facilities, military lands, and disposal facilities (Grantham *et al*., 2001; Arimoto *et al*., 2002; Whicker *et al*., 2002a, b). Wind erosion is a likely mechanism for transporting offsite contaminants at many of these sites, (e.g. Anspaugh *et al*., 1975; Breshears *et al*., 1992; Rood *et al*., 2002; Whicker *et al*., 2002a), with the airborne contaminant concentrations dependent on the competitive interactions between both wind and water erosion.

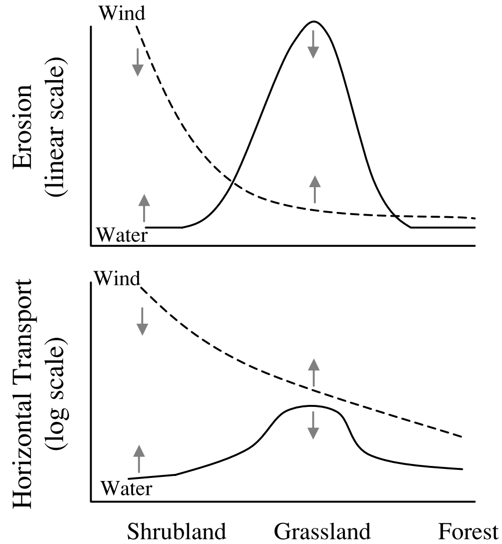
Improving our understanding and predictive capability of wind and water erosion in dryland ecosystems is likely to be of increasing concern as global change progresses, both in terms of land use change and in terms of climate change (Valentin, 1996; Gregory *et al*., 1999; Ingram *et al*., 1999; Gourdriaan *et al*., 1999; Breshears and Allen, 2002). Predicted changes such as an increase in storm frequency and intensity (IPCC, 2001) and associated erosivity of rainfall (Nearing, 2001) have potential to increase water erosion. Similarly, climate changes that effect wind characteristics at a site could have a large impact on wind erosion (Gregory *et al*., 1999). Both wind and water erosion increase dramatically following disturbance, including mechanical disturbances (Sehmel, 1980; Grantham *et al*., 2001), abandoned farmland (Okin *et al*., 2001), heavy grazing (Marticorena *et al*., 1997), fire (Baker and Jemison, 1991; Baker *et al*., 1995; Zobeck *et al*., 1989; Johansen *et al*., 2001a, b, in press; Wilson *et al*., 2001; Whicker *et al*., 2002a), and drought (Allen and Breshears, 1998; Rosenzweig and Hillel, 2000; Clark *et al*., 2002). Addressing these issues will require an improved understanding of erosion in dryland ecosystems.

How applicable are the results reported here to other semi-arid shrubland, grassland, or forest ecosystems? We have no replication within ecosystems, and, as noted, multiple factors can vary among these three ecosystem types. Although we cannot make broad inferences across different ecosystem types from our data, we build on our findings to hypothesize on trends related to wind and water erosion in dryland shrublands, grasslands, and forests. Differences in wind erosion and water erosion for gradients of dryland ecosystems have been hypothesized previously (Kirkby, 1980; Baker *et al*., 1995, drawing from Heathcote, 1983). Two factors have generally been considered in such hypotheses: the amounts of precipitation and ground cover (which, rather than remaining constant over such a gradient, probably increase with precipitation). Kirkby's (1980) hypothesis suggests that for 'natural vegetation cover' (that is, assuming some increase in vegetation cover with precipitation), water erosion exceeds wind erosion for all but the driest of sites (<100 mm annually). Somewhat similarly, Baker *et al*. (1995) hypothesize that wind erosion exceeds water erosion only for sites with <250 mm of annual precipitation. These hypotheses implicitly assume that soil texture does not vary across the gradient. Our results, which are for ecosystems ranging from 300 to 500 mm of annual precipitation, suggest that, in contrast to these previously posed and untested hypotheses, wind erosion can exceed water erosion for ecosystems with >250 mm annual precipitation. Further, our results suggest horizontal wind-driven transport exceeds that for horizontal water transport in all three ecosystems.

Drawing on the previously hypothesized trends described above (Kirkby, 1980; Baker *et al*., 1995) and building from our findings, we pose hypothesized relationships for wind and water erosion and horizontal transport in semi-arid grassland, shrubland, and forest ecosystems (Figure 7). Within and among these ecosystem types, conditions can vary with respect to precipitation regime, near-ground wind characteristics, ground cover, mean bare patch size, and soil conditions (particularly texture). Our hypotheses reflect the differences we observed and how we believe corrections in ground cover affect them. We pose these hypotheses relative to the soil textures we studied – with the shrubland being a sand at one end of the soil texture continuum, the grassland a clay at the other end, and the forest being intermediate in soil texture (silt loam) – and speculate how deviations from these textures impact on our hypotheses. Overall (Figure 7), wind erosion is hypothesized to decrease from shrublands to grasslands to forests, as we observed. In contrast, water erosion is hypothesized to be greater in grasslands than shrublands or forests (water transport exhibits a similar trend among ecosystems). For horizontal transport, wind transport is greater than water transport for all three ecosystems, and greatest in shrublands, intermediate in grasslands, and least in forests.

Several factors affect the hypothesized trends. Precipitation is generally greater in forests than shrublands or grasslands, thereby creating greater potential for runoff and water erosion. However, increases in ground cover associated with increases in precipitation may offset such effects. The hypothesized reduction in wind erosion from shrublands to grasslands to forests is also related to feedback effects of the vegetation on near-ground wind velocity. We hypothesize that the reduction in near-ground wind velocity is much greater in forests than in shrublands or grasslands, thereby reducing wind erosion and transport in forests. We hypothesize that this decrease is due not only to changes in the amount of vegetation cover (Fryrear, 1985), but also due to the patchy nature of the vegetation cover. In shrublands, not only is ground cover low, but that cover is also highly patchy, leaving large bare areas susceptible to wind erosion (Schelsinger *et al*., 1996; Okin and Gillette, 2001). The patchy nature of the ground cover, in concert with the connectedness of such patches, probably results in increased erosion (Ludwig *et al*., 1997, 2002; Davenport *et al*., 1998).

Soil texture can vary substantially among and within ecosystem type. We hypothesize how the relationships would vary with soil texture by speculating on expected changes as a soil becomes more intermediate in texture (as indicated by the arrows in Figure 7). For the sand-dominated shrubland, becoming more intermediate in soil texture represents an increase in silt corresponding to a decrease in sand, and for the grassland this represents an increase in silt corresponding to a decrease in clay; the forest system is already intermediate in soil texture.



Precipitation	Low	Medium	High
Near-surface wind velocity	Medium	High	Low
Ground cover	Low	Medium	High
Mean bare patch size	High	Medium	Low
Soil-clay content (texture)	Low (Sand)	High (Clay)	Medium (Silt Loam)

Figure 7. Hypothesized relationships for erosion and transport for shrubland, grassland, and forest ecosystems. Factors related to the hypothesized trends are listed below each ecosystem. The hypotheses are specific to the soil textures listed. Arrows indicate the expected direction in which the hypothesized curves would shift if all soil were adjusted to intermediate texture

For transport and erosion in the sand-dominated shrubland, we hypothesize that as the fraction of silt increases, the amount of water erosion increases, due to both reduced soil infiltration and increased soil erodibility. For wind erosion and transport in the shrubland, we hypothesize that as the silt fraction increases, erodibility is reduced relative to fine sands (Skidmore, 1994). Hence, moving toward an intermediate texture in the shrubland dampens the difference somewhat between wind and water erosion and transport, although under most conditions we hypothesize that wind erosion will still be dominant (Figure 7). For the grassland, as the soil texture decreases in clay content and increases in silt, we hypothesize that water erosion and transport will be reduced. Increasing the silt fraction increases the soil erodibility, but simultaneously increases the soil infiltration rate, the latter of which could dominate and produce the reduced erosion and transport. For wind erosion and transport in the grassland, we hypothesize that as the silt fraction increases, soil erodibility increases (Figure 7). Implicit in this hypothesis is the assumption that increases in soil erodibility associated with silt fraction are greater than the effects of antecedent soil moisture: clay is less erodible when wet, but in a semi-arid environment is likely to be dry for much of the year. The antecedent soil moisture needs to be considered further.

Water erosion, of course, depends on slope, and hence the contribution of water erosion to total erosion probably increases with increasing slope. In some cases, increases in water erosion associated with increases in slope could result in dominance of water erosion over wind erosion.

Testing the hypotheses we pose here will require extensive work in the future at other semi-arid ecosystems. In addition, research is needed to better relate small-scale mass flux, as we measured, to larger-scale losses or gains in sediment. Related studies of water erosion document that small-scale erosion per unit area can greatly exceed larger-scale erosion per unit area (Wilcox *et al.*, 1996, 2003; Lane *et al.*, 1997; Reid *et al.*, 1999).

CONCLUSIONS

In summary, we found that annual wind erosion, as estimated from measurements of vertical flux, was dominant over water erosion at the shrubland and forest sites. In contrast, water erosion was greater than wind erosion at the grassland site, probably due not only to vegetation cover and site climate but also to the high clay content at the site. In the forest ecosystem it is likely that other factors such as slope could result in water erosion exceeding wind erosion in some locations. Notably, we estimate that horizontal transport rates greatly exceed more traditional estimates of wind erosion, and further that horizontal transport from wind greatly exceeds that for water. On the basis of our results, we pose a set of hypotheses about cross-ecosystem differences in wind and water erosion rates to be tested further by future studies. Our results provide an initial assessment based on common methodology for comparing wind and water erosion in grassland, shrubland and forest drylands and have fundamental implications for key areas in geomorphology, biogeochemistry and land degradation, contaminant transport, and climate change.

ACKNOWLEDGEMENTS

We thank several people for their contributions to this effort, including Thomas B. Kirchner for providing wind data from the Carlsbad Environmental Monitoring and Research Center, Art Rood for providing wind data for Rocky Flats, Randy Whicker for assistance with data collection for wind erosion at Rocky Flats, and Kristine N. Mack for assistance with data collection at Los Alamos. We also thank F. Ward Whicker, Thomas E. Hakonson, Thomas B. Kirchner, Shawki A. Ibrahim, Susan R. Johnson, Leonard J. Lane, Everett P. Springer, and Cathy J. Wilson for discussions on the importance of competing erosion and contaminant transport processes. We also thank the three anonymous reviewers for helpful comments. This research was supported by the US Department of Energy, through the Environmental Science and Management Program, and by Los Alamos National Laboratory through the Technology Development Enhancement and Application program.

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