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Ecological site-based assessments of wind and water erosion: informing accelerated soil erosion management in rangelands

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Abstract. Accelerated soil erosion occurs when anthropogenic processes modify soil, vegetation, or climatic conditions causing erosion rates at a location to exceed their natural variability. Identifying where and when accelerated erosion occurs is a critical first step toward its effective management. Here we explored how erosion assessments structured in the context of ecological sites (a land classification based on soils, landscape setting, and ecological potential) and their vegetation states (plant assemblages that may change due to management) can inform systems for reducing accelerated soil erosion in rangelands. We evaluated aeolian horizontal sediment flux and fluvial sediment erosion rates for five ecological sites in southern New Mexico, USA, using monitoring data and rangeland-specific wind and water erosion models. Across the ecological sites, plots in shrub-encroached and shrub-dominated vegetation states were consistently susceptible to aeolian sediment flux and fluvial sediment erosion. Both processes were found to be highly variable for grassland and grass–succulent states across the ecological sites at the plot scale (0.25 ha). We identified vegetation thresholds that define cover levels below which rapid (exponential) increases in aeolian sediment flux and fluvial sediment erosion occur across the ecological sites and vegetation states. Aeolian sediment flux and fluvial erosion in the study area could be effectively controlled when bare ground cover was <20% of a site or the cover of canopy interspaces >100 cm in length was less than ~35%. Land use and management activities that alter cover levels such that they cross thresholds, and/or drive vegetation state changes, may increase the susceptibility of areas to erosion. Land use impacts that are constrained within the range of natural variability should not result in accelerated soil erosion. Evaluating land condition against the erosion thresholds identified here will enable identification of areas susceptible to accelerated soil erosion and the development of practical management solutions.

Key words: accelerated soil erosion; aeolian; agriculture; anthropogenic dust; dryland; land use change; state-and-transition models; sustainability; threshold.

Introduction

Land use change and intensification have resulted in accelerated rates of soil erosion and land degradation in many areas of the world's rangelands (Lal 1990, Ravi et al. 2010). Natural (potential) erosion rates vary across rangelands as a function of climate, topography, vegetation composition and structure, and the status of dynamic soil properties of crusting and aggregation (Gillette 1999). These factors operate by determining landscape susceptibility to erosion, and landscape resilience following soil loss (Lal 2001). Accelerated soil erosion, defined here as erosion greater than the natural potential erosion rate at a site, may occur when the controlling factors are modified by anthropogenic processes in excess of their natural variability (Neff et al. 2008, Ginoux et al. 2012). Increasing our understanding of the relative sensitivities of land types to climate and land use pressures by evaluating their rates of wind and water erosion under a range of conditions will enhance our capacity to identify practical management solutions (Neff et al. 2005). Conducting such analyses in the context of ecological models of landscape change may facilitate management to improve the resilience of rangeland systems so that they are sustainable and productive under different land uses, increasing development pressures, and future climate change (van de Koppel et al. 1997, Okin et al. 2009).

There is a growing body of work applying predictive models to assess rates of aeolian sediment flux and water erosion in rangelands. The focus of water erosion research has been to quantify soil loss (fluvial sediment erosion) at the hillslope and catchment scales to determine the impacts of management activities. This has included assessments of the types and intensities of impacts that rangelands can support, such as livestock grazing and military training activities (Vachta and Hutchinson 1990, Grantham et al. 2001, Bartley et al. 2010*a*, *b*). Adaptations of the Universal Soil Loss Equation (USLE; Weltz et al. 1998, Wang et al. 2007),

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Revised-USLE (Fernandez et al. 2003, Spaeth et al. 2003), Water Erosion Prediction Project (Simanton et al. 1991), and SedNet models (Bartley et al. 2007) were developed in these studies for rangeland applications. However, few field studies or model applications have concurrently, or independently, assessed management impacts on rangeland wind erosion (Breshears et al. 2003, Van Donk et al. 2003, Visser et al. 2004, Field et al. 2011*a*). Land management–wind erosion interactions have not been evaluated extensively outside of cultivated systems (Hu et al. 1997).

Consequently, most available wind erosion models have been parameterized to represent processes that are typical of cropping systems (Webb and McGowan 2009). While dust emission models (e.g., Marticorena and Bergametti 1995, Shao et al. 1996) use more generalized process representations, they do not adequately represent the effects of management (Raupach and Lu 2004). Assessments of management impacts on rangeland soil erosion depend on reliable representation of soil and vegetation conditions and the effects of disturbance due to different land uses on soil detachment and sediment transport (Ludwig et al. 2007, Al-Hamdan et al. 2012). For wind erosion, this requires representing different vegetation horizontal and vertical structures and distributions and dynamic soil surface conditions (Webb and Strong 2011). For water erosion, there is a need for models that can represent splash and thin-sheet flow processes that dominate sediment transport at the plot and hillslope scales in some rangelands (Nearing et al. 2011).

Data on the nature of management impacts on soil erosion in rangelands are becoming increasingly available through field studies (e.g., Leys and Eldridge 1998, Belnap et al. 2007, Belnap et al. 2009, Baddock et al. 2011, Field et al. 2011*b*). This has only recently been transferred to a modeling capability for both wind and water erosion assessment. For example, the wind erosion model (WEMO) of Okin (2008) represents vegetation distribution effects on aeolian horizontal sediment flux, and the Rangeland Hydrology and Erosion Model (RHEM) represents erodibility as a function of rangeland-specific vegetation life forms (Nearing et al. 2011). However, these models are yet to be applied broadly or in a structured way across diverse land units to evaluate the relative impacts of different land uses and disturbance on soil erosion.

One of the challenges associated with modeling rangeland soil erosion is the high diversity of soils and types of vegetation occurring within and among landscapes. Ecological sites provide a system for organizing landscapes at management-relevant scales based on properties that are understood to control both wind and water erosion including climate, topography, vegetation, and soil properties (Bestelmeyer et al. 2009). Because ecological sites are correlated with National Cooperative Soil Survey soil map unit components (in the USA), which are the individual soils that occur within a delineated soil map polygon, they can be predicted spatially using soils maps (Duniway et al. 2010). Ecological sites therefore provide a potential conceptual and spatially explicit structure for assessing erosion and for managing and restoring eroding landscapes.

Ecological sites can be used to organize information about vegetation cover, composition and structure, and dynamic soil properties in state-and-transition models (Westoby et al. 1989). Vegetation communities in ecological sites that are subject to land use change or intensification may undergo transitions to alternative states, defined by alterations in ecological processes and feedbacks from which recovery may be difficult on management-relevant time scales (Bestelmeyer et al. 2009). Such state changes that result from land degradation often include increases in soil erosion (Ash et al. 1994, Bestelmeyer et al. 2009). Varying degrees of accelerated erosion may, therefore, occur within and among ecological sites and their states depending on the land degradation status and the intensity of land use and climate stressors (Chartier and Rostagno 2006, Ravi et al. 2010, Miller at al. 2011). Management systems (Fig. 1) that structure assessments of soil erosion in the context of ecological sites have the potential to enhance our capacity to better manage accelerated wind and water erosion.

In the United States, the National Resources Inventory (NRI) has included ecological site information for the ~2000 plots measured each year on nonfederal lands since 2003. This number has doubled since 2011 with the inclusion of Bureau of Land Management (BLM) lands. These extensive data, which include vegetation composition, cover, and structure, have the potential for driving assessments of soil erosion (e.g., Hernandez et al. 2013) and provide opportunities to evaluate erosion rates among ecological sites and states, test site sensitivities to climate and management pressures, and identify vegetation cover levels for controlling erosion (herein thresholds) that transcend the diverse soil types and plant communities found across rangelands.

The objectives of this research were to (1) explore how aeolian sediment flux and fluvial sediment erosion assessments structured across ecological sites can inform systems for managing accelerated soil erosion in rangelands (Fig. 1), and (2) determine how soil erosion models could be improved to take advantage of the information associated with ecological sites and states. We conducted an assessment by applying recently acquired monitoring data on soil and vegetation attributes to drive models of aeolian horizontal sediment transport and fluvial sediment erosion across five ecological sites in southern New Mexico, USA. We evaluated the transport and erosion assessments in the presence and absence of an anthropogenic disturbance (graded roadways) to identify underpinning patterns in sediment transport and erosion controls across ecological sites and among their states, and how the findings

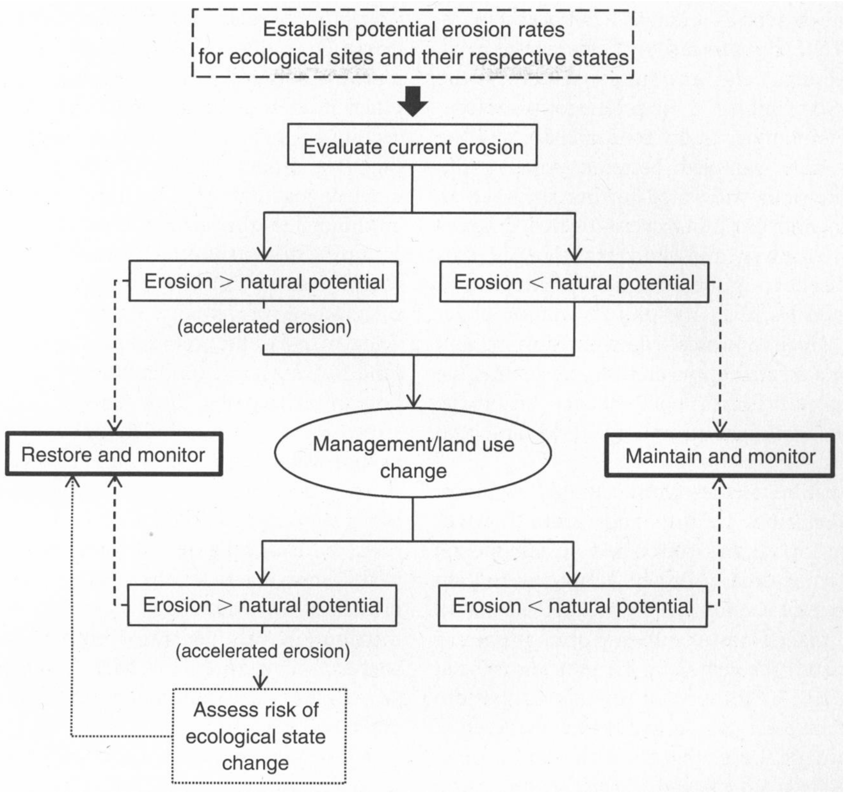

FIG. 1. Illustration of a generalized schema for assessing and managing accelerated soil erosion. Line styles and arrows help separate process in the framework. The dashed box signifies the starting point of the analysis. Solid lines and boxes represent the soil erosion evaluation process. The oval represents the management or land use change that will be evaluated in the framework. Bold boxes and dashed arrows represent management responses to outcomes of the soil erosion evaluation. The dotted box and arrow signifies an additional step that may be required if erosion rates are found to be significantly large, even after management change.

can be used to improve systems of accelerated erosion management in rangelands.

METHODS

Study area description

The study area is located in the semiarid rangelands of southern New Mexico, USA (centered on 105°52′6.0″ W, 32°29′8.6″ N; Fig. 2). The area covers a variety of landforms and associated soils including limestone hills, alluvial fans, valley bottoms, alluvial plains, and aeolian sands. The elevation ranges from 1440 to 2160 m above sea level. Plant community composition in the study area varies with soils, elevation, and landscape position. The historic plant communities in the study area are warm-season (C_4) grasslands or mixed communities of warm-season grasses, shrubs, and half-shrubs (McClaran and Van Devender 1997). Currently, dominant species include mesquite (*Prosopus glandulosa* Torr.) and black grama (*Bouteloua eriopoda* Tor.) on sandy sites, creosote (*Larrea tridentata* DC.) and black grama on gravelly sites, and tarbush (*Flourensia cernua* DC.) and tobosa (*Pleuraphis mutica* Buckley) on loamy sites. The mean annual precipitation (1960–2012) for the study area within the Tularosa Basin is 260 mm (range 124–468 mm), and on Otero Mesa is 305 mm (range 165–578 mm) with 50% falling in the summer months, from June through September. The study area includes three BLM grazing allotments in areas jointly managed by the Fort Bliss Army Base. Both cattle grazing and military training activities occur in the study area.

Field data collection

Soil and vegetation attributes were sampled at 120 plots as part of a broader study to evaluate the sensitivity of a standard rangeland assessment protocol to road impacts (Duniway and Herrick 2013). The plots were stratified across five types of ecological site with similar potential vegetation composition and vegetation production (Table 1; Appendix: Table A1). Stratification with soil maps was used, and at each plot, the ecological site soil texture was verified in the field using either soil pits or auger holes. The plots were located through a geographic information system (GIS) analysis of digitized soils maps (USDA Natural Resources Conservation Service [NRCS], *available online*)[4] and

[4] http://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx

historical Landsat-derived Normalized Difference Vegetation Index (NDVI) data to capture the range of plant community structures and productivity across the ecological sites. Sixty plots, 12 in each ecological site, were sampled in September and October 2008, and the remaining plots were sampled between August and October 2009. The plots were then further stratified so that in each year, half (30 plots) were located in areas without graded roadways, and the remaining 30 plots were located adjacent to graded roads. The presence of roads was expected to affect the plot vegetation cover and distribution, which influence sediment transport and erosion rates. This stratification enabled us to test the effects of an important disturbance mechanism on the condition of the ecological sites and their potential erosion rates.

Quantitative estimates of plant canopy gap-size distribution, vegetation height, and ground cover (including vegetation, litter, rock, and lichen crusts) were calculated from data collected along three 25-m transects at each plot (following Herrick et al. 2005). Transects were placed in a hub-and-spoke pattern originating 5 m from the center of the plot and spaced at 120° intervals. The orientation of the transect pattern was randomized for each plot to avoid systematic error at plots where roads were aligned with the cardinal directions. Vegetation cover and species composition were estimated using line–point intercept (LPI) sampling with a point spacing of 50 cm. Canopy height and gap size were measured along the same transects following Herrick et al. (2005). In addition to the soil sampling in pits/auger holes to verify the ecological sites, top soil (1 cm and 5 cm depth) samples were collected from each plot for texture analysis using the hydrometer method (Klute 1986).

Modeling approach

We applied models to assess aeolian horizontal sediment flux and fluvial sediment erosion across the study ecological sites. Here we define aeolian horizontal sediment flux as the horizontal mass transport of sediment by wind over the land surface, expressed with the units grams per centimeter per day. We define fluvial sediment erosion as the sediment loss per unit area (Mg/ha) along a hillslope via splash and thin-sheet flow processes. While sediment transport (for example, aeolian horizontal sediment flux) does not equate to sediment erosion (soil loss), we note that the limited data on the two processes suggest that they are correlated (Breshears et al. 2003).

Wind-driven sediment transport.—Assessments of aeolian horizontal sediment flux ($g \cdot cm^{-1} \cdot d^{-1}$) were made using the wind erosion model (WEMO) of Okin (2008). The WEMO core components include representation of vegetation roughness effects and a sand flux equation. Model inputs include an estimate of the soil threshold friction velocity (u_{*t}) for mobilization by wind, a wind speed probability distribution, and measures of the vegetation canopy gap distribution and average canopy height for the study area of interest. The model representation of surface roughness effects differs from other models (e.g., Raupach et al. 1993) in that it accounts for the distribution of vegetation and assumes that roughness elements affect the wind erosivity incident on the land surface, rather than the bulk erodibility of the surface itself (Okin 2008). The input canopy gap distribution, scaled by the average canopy height, is used with the input wind speed distribution to calculate a probability distribution of wind friction velocity (u_*) at the soil surface. The model then applies a sand flux equation (Gillette and Passi 1988) to calculate horizontal sediment flux for the distribution of shear stress that is in excess of the independent soil threshold friction velocity (u_{*t}). Li et al. (2013) describe the model calibration, validation, and performance relative to other wind erosion models.

Scaled canopy gap distributions as input to WEMO were generated from the canopy gap and height data measured at each plot. A wind speed probability distribution was generated from 5-min average wind speed data measured in 2009 at the nearby Holloman Air Force Base automatic weather station (located: 106°04′53″ W, 32°50′14.71″ N).

A constant soil threshold friction velocity (u_{*t}) of 25 cm/s was used for the WEMO simulations in the absence of estimates for the individual study plots. The value of the nominal u_{*t} is consistent with that of an erodible fine sand (Gillette et al. 1980). Our use of a constant u_{*t} was due to the fact that WEMO does not provide a way to estimate the threshold from soil data. Estimation of u_{*t} is further limited by the fact that it also depends on the level of soil surface disturbance, for which there is currently no standard measurement method. This limitation is critical for explaining the results.

Keeping both the wind speed and soil threshold constant enabled an assessment of how the vegetation characteristics of the plots may influence sediment transport rates, without the confounding influences of changing wind erosivity and soil erodibility. To demonstrate the effect of the soil threshold friction velocity on aeolian sediment flux we conducted a model sensitivity test in which we incrementally increased u_{*t} from 25 cm/s to 40 cm/s and 70 cm/s. The test enabled us demonstrate the effect of a change in soil crusting or surface disturbance on aeolian sediment flux, independent of disturbance impacts on vegetation. WEMO was run to estimate aeolian horizontal sediment flux for each of the plots across the five ecological sites sampled in 2008 and 2009.

Water erosion.—Fluvial sediment erosion assessments were made using the Rangeland Hydrology and Erosion Model (RHEM). RHEM is an empirical, process-based model that simulates soil loss (Mg/ha) along a hillslope via splash and thin-sheet flow erosion (Nearing et al. 2011). Concentrated flow is calculated as a function of the erosivity of overland flow, the sediment transport

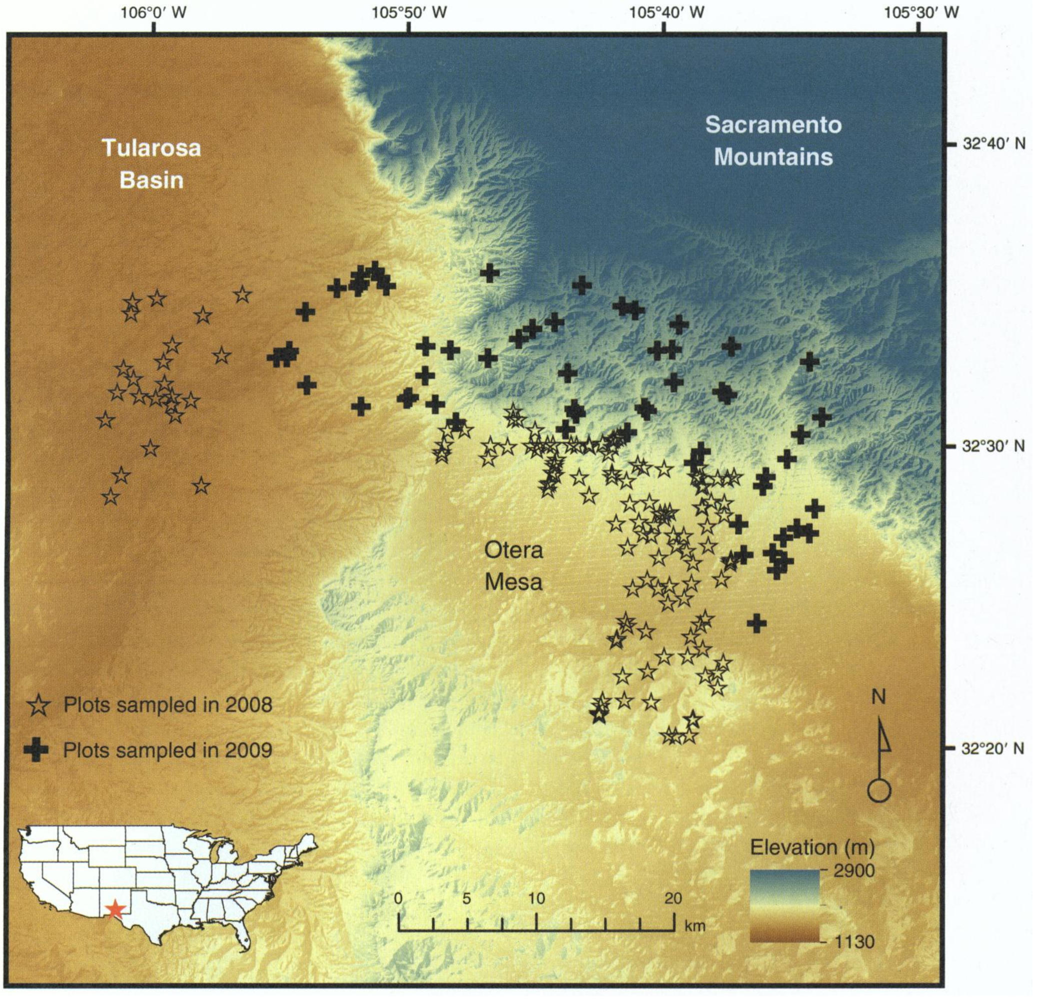

FIG. 2. Study area map showing the location of plots sampled in 2008 and 2009 in southern New Mexico, USA. Plots were located across five ecological sites on Otero Mesa and the eastern fringe of the Tularosa Basin.

capacity, and the existing sediment load in the flow (Nearing et al. 2011). The RHEM model structure, validation, and application are described in Nearing et al. (2011) and Hernandez et al. (2013).

RHEM requires inputs describing rainfall volume, duration, and intensity, which were obtained as predefined inputs from the model's application website (*available online*).[5] The data were produced from nearby Orogrande (located at 106°05′30.27″ W, 32°22′37.79″ N) precipitation records using the Cligen V5.101 stochastic weather generator (Nicks et al. 1995). Unfortunately, rainfall and wind speed data were not available from the same station for consistency between the RHEM and WEMO simulations. This was a limitation of our study, as annual precipitation is likely to vary among the locations. However, the approach enabled us to conduct the model assessments and serves to highlight the need for local measurements of weather variables to improve the accuracy of model applications.

The model slope gradient input was obtained via inclinometer at each of the study plots. In the absence of data on slope length, we kept the parameter constant at 250 m for all simulations. As slope length can be a source of variability in erosion rates, it will be important to include this parameter in future ecological-site descriptions. Fractional cover estimates obtained from the LPI measurements of total ground, plant canopy, litter, plant basal, lichen and surface rock cover, and measures of the site soil clay contents from the texture analyses, were used to calculate the RHEM friction factors for runoff and erosion and the splash and sheet erosion coefficients following Nearing et al. (2011). RHEM was run to simulate a synthetic 300-year time series of fluvial sediment erosion events at each plot sampled across the five ecological sites in 2008 and 2009. The long series was used to ensure stabilization of the

[5] http://dss.tucson.ars.ag.gov/rhem/

TABLE 1. Ecological sites in southern New Mexico, USA, that were evaluated in this study and summary of their physical characteristics.

Ecological site group	Clay content (%)	ESD slope range (%)†	Number of plots
Gravelly	12–20	3–10	24
Limestone hills	18–27	35–65	24
Loamy	7–26	1–3	24
Draw	18–35	1–5	24
Sandy	2–8	5–15	24

† Slope ranges provided in the Ecological Site Descriptions (ESD) for Major Land Resource Area 42 (https://esis.sc.egov.usda.gov/).

average annual soil loss, which was computed for each of the 120 plots (following Hernandez et al. 2013).

Data analysis

The simulated aeolian horizontal sediment flux and soil loss due to fluvial sediment erosion were initially evaluated by visually comparing their magnitudes across the ecological sites. For all analyses, we combined data from plots sampled in 2008 and 2009 on the basis that their geomorphic and local climate characteristics were highly variable and more influential than differences in precipitation between the two years. We could not make quantitative comparisons between wind and water erosion, or their rates among the ecological sites, because of the significant differences in how the models represent soil erodibility effects on erosion and because WEMO simulates aeolian sediment flux ($g \cdot cm^{-1} \cdot d^{-1}$), while RHEM simulates soil loss (Mg/ha). This was unavoidable, as no wind erosion model has been developed to estimate soil loss in rangelands and which also accurately represents the spatial distribution and structure of vegetation in the study area. While the soil erodibility representations and outputs of WEMO and RHEM are not comparable, the models provide the most reliable process representations of wind and water erosion processes available for the study area.

The large variability in simulated aeolian sediment flux and fluvial sediment erosion within ecological sites led us to hypothesize that (1) the vegetation community states (that is, the vegetation life forms and their distributions) within ecological sites have a significant effect on the erosion processes, and, (2) by reducing vegetation cover, the presence of graded roadways will result in a significant change (increase) in sediment transport and erosion. To test the hypothesis that vegetation state is important for aeolian sediment flux and fluvial sediment erosion, we classified the 120 plots (60 adjacent to roads) into six vegetation groups based on their dominant plant community life forms and their horizontal and vertical structures in the landscape. These were determined from species composition data and photographic records that were collected during the field sampling. The groups included grasslands, natural grass–shrub mix, grass and juniper mix, grass and succulent mix, shrub-encroached, and shrub-dominated plots. The vegetation structures within these generalized groups are typical of the dominant ecological states that occur in the region (USDA NRCS; *available online*).[6]

We used mixed-model analysis of variance (ANOVA) to test for differences in simulated aeolian sediment flux and fluvial sediment erosion rates between the vegetation groups (herein states) and between sites without and with adjacent roadways. All statistical analyses were conducted using SAS version 9.3 software (SAS Institute 2009). We assessed all variables for normality and homogeneity of variance prior to the analysis. As expected, the simulated aeolian horizontal sediment flux and fluvial sediment erosion displayed skewed distributions. This was due to the nonlinear response of erosion (and the models) to differences in ground cover among the plots. We therefore conducted a log-transformation of the simulated data for input to the ANOVA, which was run separately for the aeolian sediment flux and fluvial sediment erosion. We calculated post hoc pairwise comparisons of the vegetation community states using Fisher's least significant difference (LSD) tests. More rigorous significance tests were not applied given the uncertainty in the simulated flux and erosion assessments.

Finally, we evaluated patterns in the aeolian sediment flux and fluvial sediment erosion responses to the site vegetation characteristics through an analysis of the relationships between the model outputs and the measured ground cover and canopy gap size data. While this analysis was circular in nature, and assumes that the models reflect reality, it facilitated the identification of general responses and vegetation thresholds that transcend the ecological sites and that could be used to make practical management recommendations.

RESULTS

Ground cover attributes of the ecological sites

Measured vegetation cover and the area of erodible bare ground were highly variable within and among the five ecological sites at the 0.25-ha plot scale (Fig. 3). The limestone hills sites had the largest total ground cover at plots without and with adjacent roads. The loamy and sandy sites had the smallest total ground cover at sites without roads, while total ground cover was similar at

[6] https://esis.sc.egov.usda.gov/

the gravelly, loamy, draw, and sandy sites with roads. Foliar cover and plant basal cover showed little variability between the ecological sites. The percent cover of rock fragments, which influences surface roughness and provides a protective layer over the soil surface, was highly variable between the ecological sites. Rock cover was greatest for plots in the limestone hills site and smallest in the loamy and sandy sites. The percent cover of bare ground was more variable at plots sampled without adjacent roads. Plots in the loamy, sandy, and gravelly sites consistently had the largest areas of exposed bare ground.

The effect of roads on soil erosion processes among ecological sites

The presence of roads did not have a significant effect on either the simulated aeolian horizontal sediment flux ($P = 0.6032$) or rates of fluvial sediment erosion ($P = 0.6817$). Variation in the simulated aeolian sediment flux and fluvial erosion was large among the ecological sites, reflecting patterns of measured vegetation cover (Fig. 4). However, the magnitude of aeolian sediment fluxes and fluvial sediment erosion was similar for plots located away from and adjacent to graded roads. The patterns in the simulated aeolian sediment flux and fluvial erosion across the ecological sites can be explained by their ground cover attributes, and largely reflects the percentage of exposed bare ground (Fig. 3). This is particularly true for the aeolian sediment fluxes, which appear to be largest in the loamy and sandy ecological sites. However, the lack of site-specific soil threshold friction velocity (u_{*t}) information that could be applied in the model simulations (see *Methods*) means that real patterns of aeolian sediment flux among the ecological sites could be quite different to those found here.

While patterns of the simulated aeolian sediment flux reflect the measured ground cover attributes, slope appears to have had a significant effect on the simulated soil loss due to fluvial sediment erosion (Fig. 4). The simulated fluvial sediment erosion rates were largest for the limestone hills ecological site (slope range 35–65%), and smallest at the draw ecological site (slope range 1–5%). Differences in simulated fluvial sediment erosion among the gravelly, loamy, and sandy ecological sites appear to reflect more their ground cover attributes.

Soil erosion processes between vegetation states

Vegetation state had an overall significant effect on simulated aeolian horizontal sediment flux ($P < 0.0001$) and rates of fluvial sediment erosion ($P < 0.0001$). Shrub-dominated and grassland plots had the largest mean aeolian sediment fluxes (Table 2). The aeolian sediment flux for shrub-dominated plots was significantly larger than for shrub-encroached plots ($P = 0.0016$) and plots with a natural mix of grasses and shrubs ($P < 0.0001$) or grasses and succulents ($P = 0.0329$). Shrub-encroached plots and plots with a natural mix of grasses and shrubs did not have significantly different aeolian

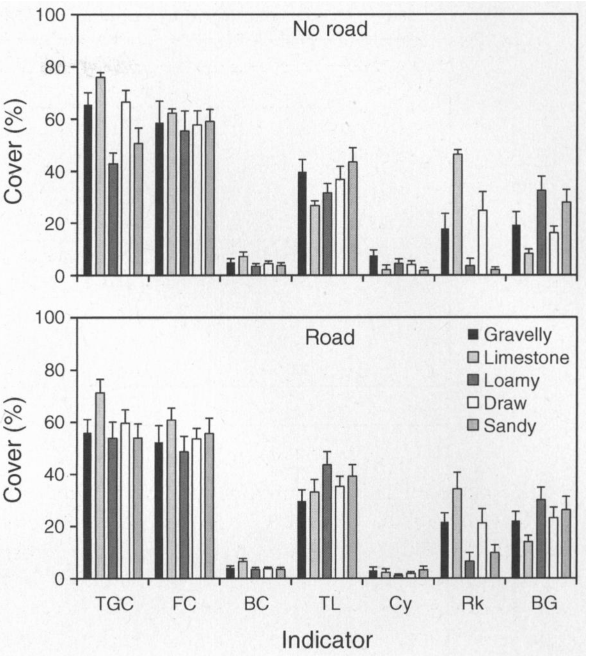

FIG. 3. Summary of ground cover characteristics (2008 and 2009 combined) for the five ecological sites. Data were collected along transects using the line–point intercept (LPI) method (see *Methods* section). Ground cover characteristics include: total ground cover (TGC, including foliar and basal cover, litter, cyanobacteria and rock cover), foliar cover (FC, of plant canopies), plant basal cover (BC), total litter cover (TL), cyanobacteria crust cover (Cy), rock cover (Rk), and bare ground cover (BG, including that beneath overhanging plant canopies).

sediment fluxes to plots with a mix of grasses and succulents ($P > 0.05$), but did have significantly larger mean aeolian sediment flux than plots with a mix of grasses and oneseed juniper (*Juniperus monosperma* (Engelm.) Sarg.) (Table 2).

Patterns of simulated fluvial sediment erosion between the vegetation states differ from those of aeolian horizontal sediment flux (Table 3). Overall, plots with a mix of grasses and succulents had the largest simulated fluvial sediment erosion, which was significantly larger than for the other vegetation states ($P < 0.05$). Shrub-dominated and shrub-encroached plots had significantly larger fluvial erosion rates than plots with a natural mix of grasses and shrubs (Table 3). Grassland plots had the smallest simulated fluvial erosion rates and, unlike aeolian sediment flux, were significantly smaller than for shrub-dominated ($P = 0.036$) and shrub-encroached plots ($P = 0.0179$).

The effect of vegetation cover and spatial patterns on soil erosion processes

Aeolian horizontal sediment flux and fluvial sediment erosion display threshold-type responses to changes in ground cover and the proportion of exposed and potentially erodible soil surface (Fig. 5). These thresholds describe the vegetation cover levels at which rapid

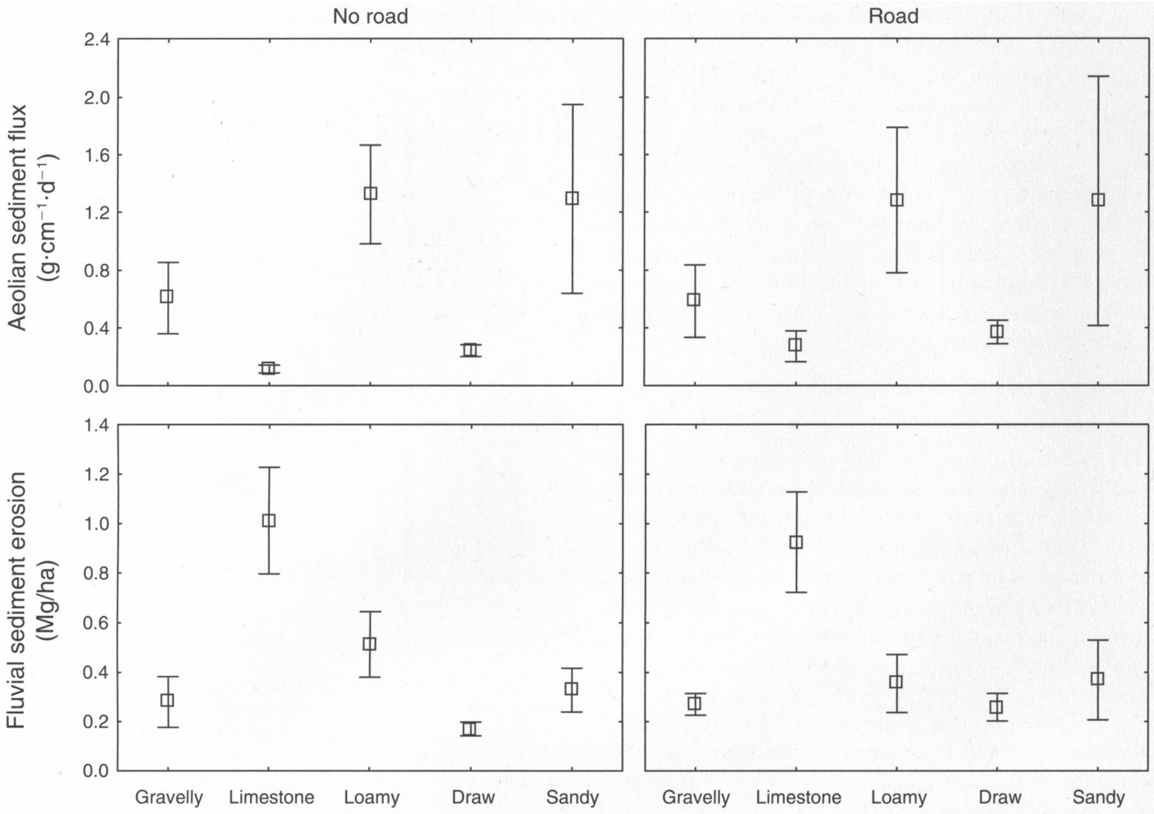

FIG. 4. Simulated aeolian horizontal sediment flux (top) and soil loss due to fluvial sediment erosion (bottom), averaged across plots within the ecological sites without (left) and with (right) adjacent roadways. Data show means ± SE.

TABLE 2. ANOVA results showing the presence/absence of significant differences in simulated aeolian horizontal sediment flux between plant community structural groups (states), and for plots with and without adjacent roads.

Comparison	Mean_1	Mean_2	t	SE	P
GRASS–GRJU	0.87	0.04	5.69	5.69	<0.0001*
GRASS–GRSHM	0.87	0.25	2.59	2.59	0.0108*
GRASS–GRSU	0.87	0.37	1.34	1.34	0.1823
GRASS–SHDO	0.87	1.51	−1.27	−1.27	0.2056
GRASS–SHEN	0.87	0.39	1.53	1.53	0.1280
GRJU–GRSHM	0.04	0.25	−3.73	−3.73	0.0003*
GRJU–GRSU	0.04	0.37	−2.98	−2.98	0.0035*
GRJU–SHDO	0.04	1.51	−6.98	−6.98	<0.0001*
GRJU–SHEN	0.04	0.39	−4.76	−4.76	<0.0001*
GRSHM–GRSU	0.25	0.37	−0.29	−0.29	0.7745
GRSHM–SHDO	0.25	1.51	−4.30	−4.30	<0.0001*
GRSHM–SHEN	0.25	0.39	−1.26	−1.26	0.2106
GRSU–SHDO	0.37	1.51	−2.16	−2.16	0.0329*
GRSU–SHEN	0.37	0.39	−0.45	−0.45	0.6504
SHDO–SHEN	1.51	0.39	3.24	3.24	0.0016*
No road–road	0.72	0.76	−0.52	0.09	0.6032

Notes: Groups include: grasslands (GRASS), natural grass–shrub mix (GRSHM), shrub-encroached (SHEN), shrub-dominated (SHDO), grass-succulent mix (GRSU), and grassland containing oneseed juniper (*Juniperus monosperma* (Engelm.) Sarg.; GRJU). The number of samples (n) in each group is: GRASS (20), GRSHM (21), SHEN (28), SHDO (38), GRJU (8), and GRSU (5). Mean_1 and Mean_2 refer to the first and second, respectively, ecological state listed in the first column of the table. ANOVA results for comparison of simulated aeolian horizontal sediment flux ($\text{g·cm}^{-1}\text{·d}^{-1}$); df = 112.

* Indicates a significant difference ($P < 0.05$) between groups.

TABLE 3. ANOVA results showing the presence/absence of significant differences in simulated soil loss due to fluvial sediment erosion between plant community structural groups (states), and for plots with and without adjacent roads.

Comparison	$Mean_1$	$Mean_2$	t	SE	P
GRASS–GRJU	0.21	0.24	−0.12	0.15	0.9085
GRASS–GRSHM	0.21	0.25	0.22	0.11	0.8290
GRASS–GRSU	0.21	0.24	−4.55	0.18	<0.0001*
GRASS–SHDO	0.21	0.48	−2.12	0.09	0.0360*
GRASS–SHEN	0.21	0.60	−2.40	0.11	0.0179*
GRJU–GRSHM	0.24	0.25	0.28	0.15	0.7819
GRJU–GRSU	0.24	1.49	−3.90	0.21	0.0002*
GRJU–SHDO	0.24	0.48	−1.38	0.14	0.1699
GRJU–SHEN	0.24	0.60	−1.64	0.15	0.1036
GRSHM–GRSU	0.25	1.49	−4.73	0.18	<0.0001*
GRSHM–SHDO	0.25	0.48	−2.42	0.09	0.0171*
GRSHM–SHEN	0.25	0.60	−2.70	0.10	0.0080*
GRSU–SHDO	1.49	0.48	3.56	0.17	0.0005*
GRSU–SHEN	1.49	0.60	3.22	0.17	0.0017*
SHDO–SHEN	0.48	0.60	−0.50	0.09	0.6161
No road–road	0.46	0.44	−0.41	0.07	0.6817

Note: See Table 2 for abbreviations and sample numbers. ANOVA results for comparison of simulated fluvial sediment erosion (Mg/ha); df = 113.
* Indicates a significant difference ($P < 0.05$) between groups.

(exponential) increases in sediment flux and soil loss begin to occur. The thresholds lie at ~20% bare ground and ~50% total ground cover for aeolian sediment flux, and ~20% bare ground and ~70% total ground cover for fluvial sediment erosion. Beyond the thresholds, aeolian horizontal sediment flux and fluvial sediment erosion increase exponentially with declining ground cover and an increasing proportion of bare ground. The form of the relationship was consistent between plots with and without road disturbances (Fig. 5). Impor-

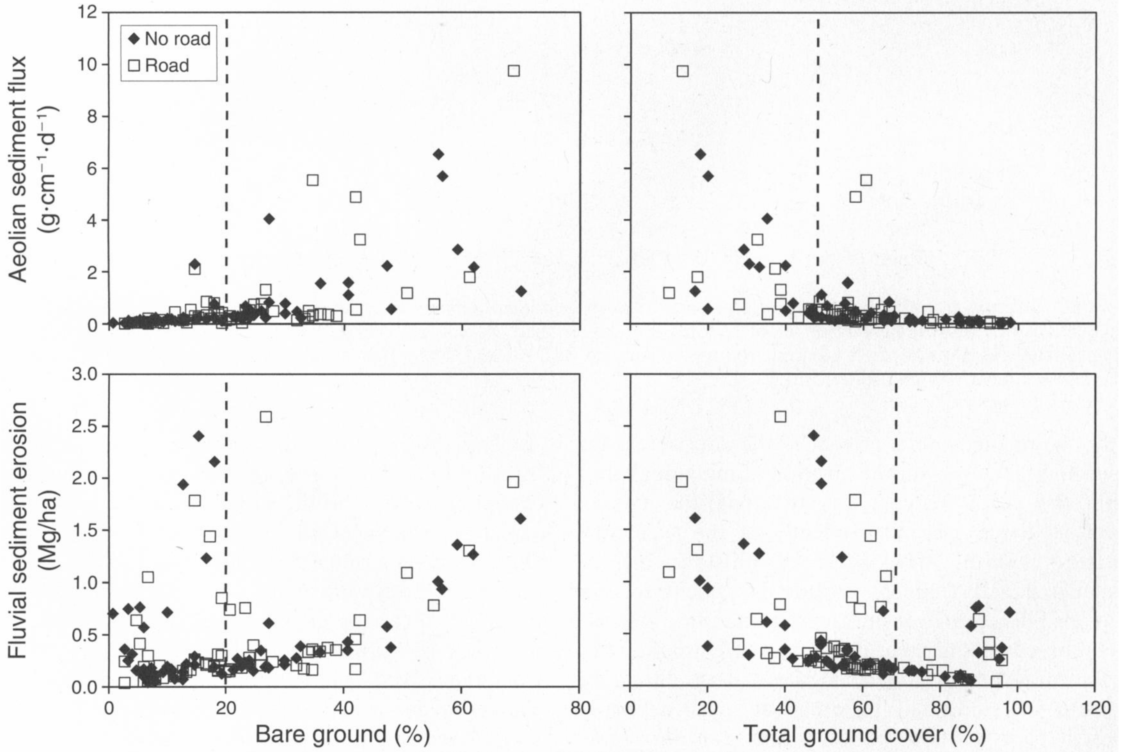

FIG. 5. Graphs showing the relationship between simulated aeolian horizontal sediment flux (top), soil loss due to fluvial sediment erosion (bottom), and the fractional cover of bare ground and total ground cover. Data are shown from all ecological sites, at plots without and with adjacent roadways in 2008 and 2009. Dashed lines indicate proposed thresholds for horizontal sediment flux and soil erosion.

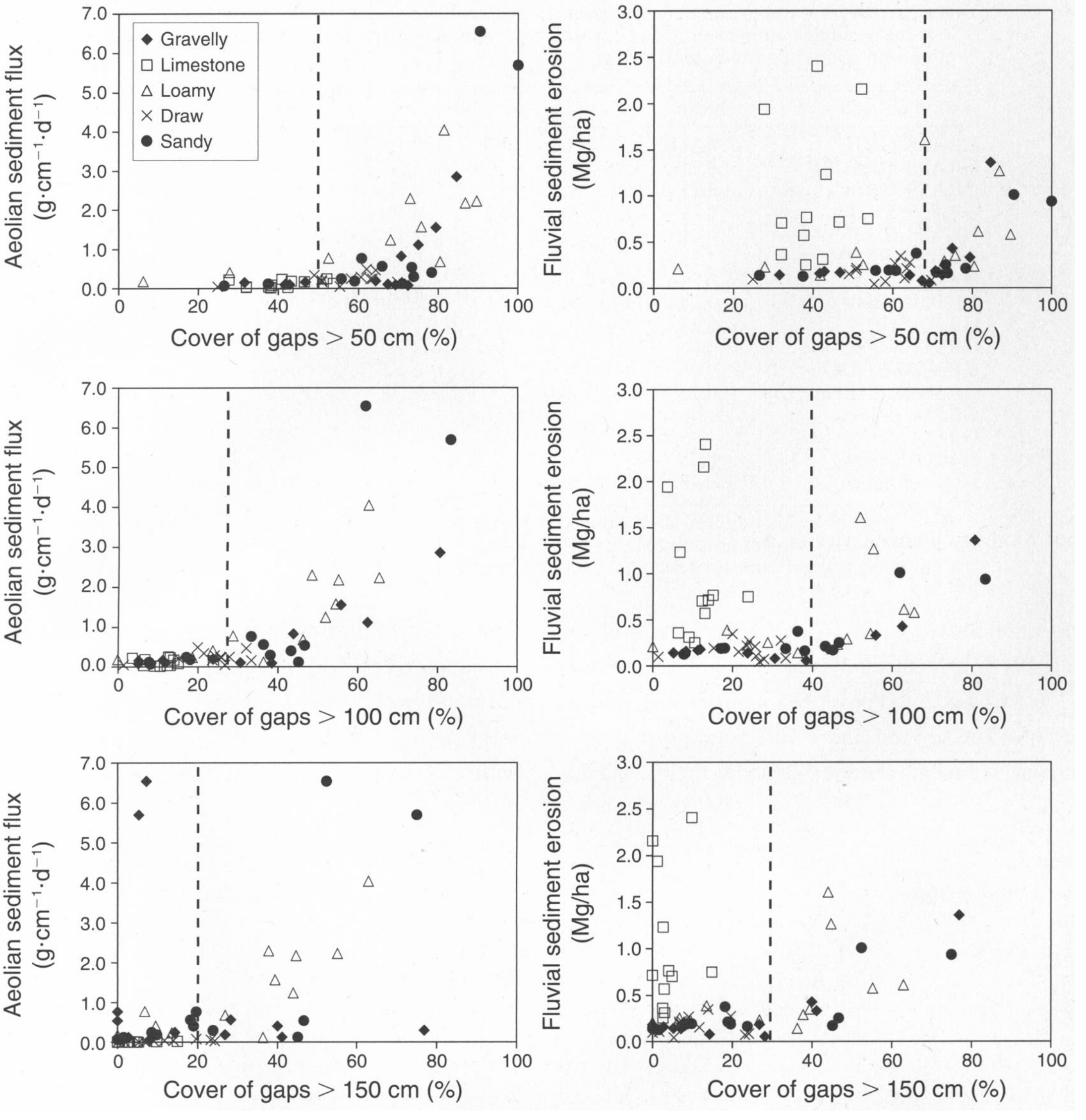

FIG. 6. Selected graphs showing the relationship between simulated aeolian horizontal sediment flux, soil loss due to fluvial sediment erosion, and the percent cover of canopy gaps larger than 50 cm, 100 cm, and 150 cm. Data are shown for all five ecological sites, at plots without adjacent roadways sampled in 2008 and 2009. Dashed lines indicate proposed thresholds for horizontal sediment flux and soil erosion.

tantly, the relationship is present in the data across the five ecological sites, with the position of individual plots within the graph varying largely along the axis of vegetative cover and independently of the vegetation community states. The effect of hillslope on the simulated fluvial sediment erosion is evident for the limestone hills ecological site and outweighs the effect of vegetation cover and area of exposed bare ground (Fig. 5). Fluvial sediment erosion at plots in the limestone hills site was indicated to occur at lower levels of exposed bare ground (<20%) that for the other ecological sites.

The size distribution of vegetation canopy gaps also has an effect on the magnitude of the simulated aeolian horizontal sediment flux and fluvial sediment erosion. Aeolian horizontal sediment flux displays a clear threshold-type response to the size distribution of canopy gaps, and this threshold appears to be consistent across the ecological sites and vegetation states (Fig. 6). Our modeling results show that the aeolian sediment flux increases exponentially above 0.5 $g \cdot cm^{-1} \cdot d^{-1}$ when the cover of canopy gaps >50 cm reaches ~50%, when the cover of canopy gaps >100 cm reaches ~30%, or when the cover of canopy gaps >150 cm reaches ~20%. The results indicate that plots that were patchy and had many large canopy gaps would be likely to experience more aeolian sediment transport than sites with many small or few large canopy gaps.

The simulated fluvial sediment erosion response to increasing canopy gap size was not as strong as for aeolian sediment flux. We found considerably more variability in the relationship (Fig. 6). This variability results from between-site differences in hillslope, which have a strong effect on the erosivity of runoff and sheet flow in RHEM (Nearing et al. 2011). The effect of hillslope is again particularly evident for limestone hills plots (Fig. 6). Our results indicate that this slope effect can be moderated by the ground cover characteristics of the plots, which appear to have a stronger and more consistent impact on the simulated fluvial sediment erosion. When not confounded by hillslope influences, fluvial sediment erosion had an apparent threshold-type response; increasing exponentially when sites had >70% cover of canopy gaps > 50 cm, >40% cover of canopy gaps > 100 cm, and >30% cover of canopy gaps > 150 cm. However, the simulations indicate that fluvial sediment erosion may occur at a lower cover of these canopy gap sizes in any of the ecological sites, with or without roadway disturbances (Fig. 6).

The effect of soil surface disturbance on aeolian sediment flux

The disturbance of rangeland soils, for example, by livestock trampling, disrupts surface crusts and increases the susceptibility of exposed soil surfaces to wind erosion. This increase in erodibility manifests as a decline in the soil threshold friction velocity (u_{*t}) for mobilization by wind (Gillette et al. 1980). Fig. 7 reports sensitivity test results, which show the aeolian horizontal sediment flux response to a reduction in u_{*t} relative to the fraction of bare ground at the study plots. At around the threshold of 20% bare ground simulated horizontal sediment flux increases exponentially and by an order of magnitude with a reduction in u_{*t} from 70 cm/s to 25 cm/s. The effect of the soil threshold change is consistent across the ecological sites and treatments. An increase in the soil threshold above 70 cm/s would apparently reduce aeolian horizontal sediment flux in the study area to a very low level (<0.1 $g \cdot cm^{-1} \cdot d^{-1}$).

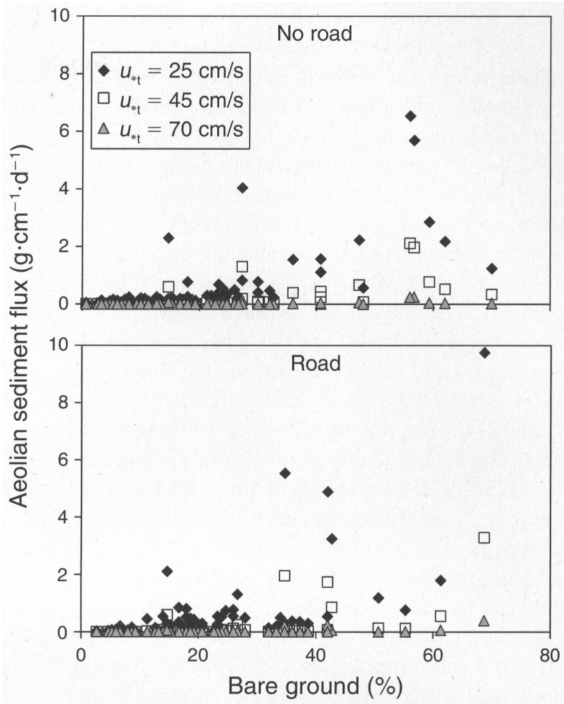

FIG. 7. Simulated effects of the threshold friction velocity (u_{*t}) on aeolian horizontal sediment flux relative to the percent cover of bare ground at plots from all five ecological sites sampled without and with adjacent roadways in 2008 and 2009.

DISCUSSION

Structuring soil erosion assessments within an ecological site framework has potential to improve the utility of assessments for monitoring and managing accelerated soil erosion in rangelands. The approach enabled us to resolve the effects of vegetation community life form and spatial structure on erosion rates, indicative of the effects of an ecological state change, identify the potential impacts of a site disturbance (graded roadways) on erosion, and elucidate underpinning patterns and thresholds in erosion responses and how they manifest across ecological sites and their respective states. The assessment also demonstrates that accounting for spatial variability within ecological sites is important for erosion monitoring and management strategies.

Spatial patterns of measured erosion controls and simulated aeolian horizontal sediment flux and fluvial sediment erosion were highly variable at the plot scale (0.25 ha) within and among the five ecological sites. This finding reflects differences in biophysical characteristics of the study plots sampled in 2008 and 2009 which, given their scale of variability (<1 ha), is feasibly assessable for management. Our inability to identify a robust pattern in aeolian horizontal sediment flux between the ecological sites does not suggest that the site stratification by ecological site is unimportant for wind and water erosion studies. Rather, it reflects the absence of information in our model assessments defining the soil threshold friction velocity (u_{*t}) characteristics of the ecological sites (see *Methods*). Differences in u_{*t} between ecological sites will result in significant differences in actual and relative sediment transport rates (Gillette 1999). Including u_{*t} information in model assessments is likely to produce more consistent patterns and a large segregation of the susceptibility of ecological sites to erosion over time. For example, the sandy ecological site will typically have $u_{*t} \approx 25$ cm/s while the loamy site may have $u_{*t} \approx 200$ cm/s, potentially dropping to ~25 cm/s under heavy disturbance (Gillette et al. 1980). These differences in soil threshold friction velocity would strongly modify the apparent patterns in simulated aeolian sediment flux among the ecological sites due to vegetation cover and distribution. Differences in soil properties among ecological sites will also influence their relative sensitivities to land use and disturbance, which differentially affect rates of aeolian sediment flux and fluvial sediment erosion (Ravi et al. 2010, Field et al. 2011*b*). Developing reliable estimates of the threshold friction velocity of soils will be critical for framing model assessments of aeolian sediment flux in the context of ecological sites and their states.

While patterns in erosion rates among ecological sites could not be established with confidence, our results demonstrate the potential effects of ecological state change. Locations that were shrub encroached or shrub dominated were consistently susceptible to both aeolian sediment flux and fluvial sediment erosion. This result supports previous observations that the degradation of grasslands and their transition to shrub-dominated states can increase their susceptibility to erosion (Ash et al. 1994, Tongway and Ludwig 1996, Breshears et al. 2003, Chartier and Rostagno 2006, Okin et al. 2009). Increases in aeolian sediment flux and fluvial sediment erosion in degraded landscapes arise largely as a consequence of changes in their functional and structural connectivity (Field et al. 2009). For example, an increase in shrub density in rangelands is typically associated with a reduction in grass cover, which creates gaps between plants that are more susceptible to wind- and water-driven sediment transport (Ravi et al. 2010). Feedbacks between the ecologically induced vegetation change (e.g., management) and geomorphic processes drive soil biogeochemical changes (nutrient loss) that may reinforce vegetation state changes and further accelerate soil erosion (Okin et al. 2009).

The large simulated aeolian horizontal sediment flux for grassland plots and large fluvial sediment erosion for plots with a grass–succulent mix suggests that their vegetation cover can at times make them more susceptible to erosion than shrublands (Throop et al. 2012). Variability in cover is a characteristic of mixed grass–shrublands globally and likely results from differences in phenology and resource use between grasses and shrubs in response to spatial variations in soil properties, interannual climate variability, and drought (Ogle and Reynolds 2004, Scanlon et al. 2005, Pennington and Collins 2007). Thus, while shrub-encroached and shrub-dominated sites may be consistently erodible, our results indicate that grasslands can also experience high rates of erosion. Evaluating aeolian sediment flux and fluvial sediment erosion from repeated measurements of plots would reveal the magnitude of temporal variations in erosion between ecological sites and states and enable an analysis of the mechanisms driving the differences in space and time.

Disturbance of sites due to graded roadways did not have a consistent or significant ($P < 0.05$) effect on the simulated wind-driven sediment flux or water erosion. Our results indicate that we should, therefore, reject our hypothesis that by reducing vegetation cover the presence of graded roads will result in a significant change (increase) in modeled aeolian sediment flux and fluvial sediment erosion. The effect of landscape disturbance on erosion rates is dependent on how disturbance processes influence the physical controls on erosion. In the models applied here, graded roadways can have the effect of reducing ground cover by adding a large canopy interspace (gap) to the landscape. In theory, this would locally increase exposed bare ground, wind fetch, and wind erosivity (Okin et al. 2006). However, our results show that the road disturbances did not alter vegetation cover levels outside the large variability measured at the other plots, and this was reflected in the aeolian sediment flux and fluvial sediment erosion rates.

Importantly, our results suggest that the models may not have adequately represented the road disturbance impacts in accounting for vegetation attributes alone. This outcome is noteworthy, as the road effects may have been larger had our modeling approach accounted for soil disturbance impacts on aeolian sediment flux and landscape setting-road interactions and their hydrological impacts on runoff (Duniway and Herrick 2011, 2013). For example, RHEM does not simulate rill and gully erosion, nor does the model account for landscape-scale hydrologic connectivity (e.g., concentration of up-slope runoff by roads). Road surface type and orientation with respect to the direction of erosive winds are also known to have a more significant effect than shown here on local dust emissions (Goossens and Buck 2009).

While roads did not have a significant effect on aeolian sediment flux or fluvial sediment erosion, the impacts of soil surface disturbance for wind erosion can be large. Our sensitivity analysis demonstrated the significant effect of changes in u_{*t} on simulated aeolian horizontal sediment flux (Fig. 7). The changes in u_{*t} represented in the analysis are commensurate with those measured for crusted and disturbed rangeland soils (Gillette et al. 1982, Belnap et al. 2007, Baddock et al. 2011) and illustrate the importance of dynamic soil properties for erosion. It also further demonstrates the importance of representing differences in soil properties and disturbance for assessing aeolian sediment flux between ecological sites and its response to anthropogenic disturbance.

Developing approaches to measure the threshold friction velocity of soils across the vast areas covered by rangelands will, therefore, be central to providing robust evaluations of land use and management impacts on accelerated wind erosion (Webb and Strong 2011). Further testing and refining soil erodibility representations in water erosion models, such as RHEM, is also likely to improve model estimates of soil loss across the diverse rangeland soil textures, chemistries, and for soils with physical and biological crusts.

Assessing aeolian horizontal sediment flux and fluvial sediment erosion across ecological sites enabled us to resolve underpinning patterns in sediment transport and erosion responses to vegetation characteristics. We found that both aeolian sediment flux and fluvial sediment erosion display threshold-type responses to declining ground cover and an increased frequency of large vegetation canopy gaps. Generalizing the results for both aeolian sediment flux and fluvial erosion, we suggest that soil transport and erosion can be effectively reduced or controlled in the study area when bare ground cover is <20% of a site or total ground cover is >50%. Similarly, our results show that aeolian sediment flux and fluvial sediment erosion can be controlled when the cover of canopy interspaces >50 cm in length is less than ~50%, the cover of canopy interspaces >100 cm in length reaches ~35%, or the cover of canopy interspaces >150 cm in length reaches ~20%. These thresholds will be influenced by vegetation height and hillslope gradient, but for the most part, appear to transcend the ecological sites and their respective vegetation states. While the thresholds have been identified on the basis of model input–output comparisons, they reflect the underpinning (measured) physical processes controlling wind and water erosion (Li et al. 2007, Field et al. 2011*b*, Miller et al. 2011, Munson et al. 2011). We therefore have confidence in the vegetation threshold positions. The inclusion of site-specific u_{*t} information in WEMO would have reduced the apparent strength of the vegetation effects on aeolian sediment flux among ecological sites. Nonetheless, our results indicate that the vegetation threshold effects are consistent within ecological sites and among the generalized states. This process knowledge can be applied with knowledge of the differential susceptibility of vegetation states to improve erosion management systems (Fig. 1).

Managing wind and water erosion in rangelands requires accounting for differences in the resilience of ecological sites and their states to land use pressures and their vulnerability to accelerated soil erosion. Land use pressures manifest in rangelands through modifications to soil erodibility, the composition and spatial structure of vegetation, and their temporal variability among years. Vegetation cover levels in rangelands will naturally move above and below the vegetation thresholds that make land most susceptible to soil erosion, producing a range of naturally occurring erosion rates. The vegetation thresholds controlling aeolian sediment transport and fluvial erosion rates transcend vegetation states within ecological sites. They can, therefore, be applied to evaluate where and when sites may become susceptible to soil erosion. Land use and management activities that alter cover levels such that they cross the thresholds will increase the susceptibility of sites to erosion. Similarly, our results indicate that when vegetation state changes occur the potential aeolian sediment transport and fluvial erosion rates of ecological sites may also change. Land use impacts on vegetation and soils that are constrained within the natural variability of sites, respective of their condition under the current climate, should not result in accelerated soil erosion. Ecological sites that are resilient to land use pressures may infrequently cross such thresholds, while less resilient sites are more prone to changes in vegetation and soil erodibility. This could result in more frequent instances of accelerated soil loss and initiate or increase in site degradation (Tongway and Ludwig 1996, Okin et al. 2009).

Structuring aeolian sediment transport and fluvial sediment erosion assessments on an ecological site basis provides a novel approach for identifying areas which are most (least) resilient to land use pressures and most (least) vulnerable to accelerated soil erosion. Soil erosion assessments conducted through field measurements or modeling can be used to determine the range of natural potential erosion as it varies between ecological sites in rangelands (Fig. 1). This requires accounting for the natural variability in both soil and vegetation attributes, differences in potential erosion due to vegetation state, and the range of wind and water erosivity conditions that could be expected to occur at the location(s) in question. The information generated by these initial assessments could then form the basis of subsequent erosion assessments, including the determination of whether current erosion rates are within or in excess of the natural potential erosion rates of a site.

Ecological site vegetation state and thresholds of vegetation cover and distribution could be used to inform this assessment and to answer questions about land use and management change impacts on potential and actual erosion. For example, will a change in stocking rate or management intensity for a particular ecological site (state) increase the frequency at which ground cover falls below the safe thresholds for controlling erosion? Decisions can then be made based on ground cover monitoring data, erosion measurements, and model simulations as to whether the current management should be maintained, and its impacts monitored, whether a desired land use or management change could be safely implemented, or whether restoration and a reduction in land use intensity are required to reduce erosion rates and the potential for site degradation.

Conclusions

The objectives of this research were to (1) explore how aeolian sediment flux and fluvial sediment erosion assessments structured across ecological sites can inform systems for managing accelerated soil erosion in rangelands, and (2) determine how soil erosion models could be improved to take advantage of the information associated with ecological sites and states. Our results show that systems for managing land use impacts on erosion can benefit from structuring wind and water erosion assessments in the context of ecological sites and site dynamics. The benefits for management arise from the types of information that structured analyses can provide, in revealing the following: the variability in natural potential aeolian sediment flux and fluvial erosion rates and current sediment flux and erosion rates in the presence of anthropogenic disturbances across ecological sites; the impacts of vegetation state changes on aeolian sediment flux and fluvial erosion, manifest through changes in vegetation composition and distributions; and the presence of vegetation thresholds that can be used for monitoring and controlling land susceptibility to accelerated erosion and which transcend ecological sites and their respective vegetation states. Such information could be applied to facilitate the development of land use and land management strategies that accommodate differences in ecological site resilience and vulnerability to accelerated soil erosion, and in prioritizing restoration activities.

The global applicability of ecological potential-based classification systems, such as the U.S. ecological site classification of landscapes, gives our approach to assessing accelerated soil erosion broad application potential outside the study area. This includes applications in both rangelands and croplands, in which soil erosion can be evaluated in the context of ecological site and vegetation state with varying levels of management influence. Having the capacity to apply the approach across diverse soil and land cover types will enable assessments of land use and management change impacts on soil erosion in a range of settings. This broad applicability and the utility of the approach for identifying management thresholds is an important advancement for establishing natural potential erosion rates and the timing, magnitude, and location of accelerated soil erosion.

The research has raised some important lessons about quantifying and managing accelerated soil erosion. First, the research has shown that combining field measurements with modeling approaches can provide a means for effectively assessing aeolian horizontal sediment flux and fluvial sediment erosion across diverse rangeland systems. This approach has potential for making use of extensive monitoring data to inform erosion management, in the United States and internationally. The adoption of standard measurement and monitoring protocols (e.g., Toevs et al. 2011) would facilitate such assessments and enable comparisons of management impacts between ecological sites and across broader rangeland systems. The development of wind and water erosion models whose outputs are comparable (e.g., $\mathrm{Mg \cdot ha^{-1} \cdot yr^{-1}}$) will also be important for comparing rates of wind and water erosion among ecological sites and states, and simultaneously accounting for the two erosion processes. Representing the effects of similar soil and vegetation attributes on erosion will likely aid model output comparisons.

Second, the study has shown the importance of ongoing research to represent the effects of dynamic land surface conditions (soil erodibility and vegetation) to improve model assessments of accelerated soil erosion. The absence of data on site-specific soil threshold friction velocities (u_{*t}) limited our capacity to compare aeolian sediment flux between ecological sites, and is critical for such analyses. The development of models to predict u_{*t} should arguably be supported by the development of reliable data on land use and management intensities (e.g., stocking rates), which are required to quantify the magnitude of management impacts on erosion. Land use itself is not always a good indicator of disturbance impact in rangelands as it does not capture spatiotemporal variations in management intensity or landscape responses to management or management change. Finally, linking ecological site-based assessments of soil erosion directly to state-and-transition models of ecological change may provide additional opportunities to evaluate land use and climate change impacts in rangelands and identify options for their sustainable management.

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Supplemental Material

Appendix

Ecological site types used for stratification of the study area plots, listing the ecological site names and numbers associated with each group (*Ecological Archives* A024-084-A1).