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Application of a rangeland soil erosion model using National Resources Inventory data in southeastern Arizona

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Abstract: Rangelands comprise a large portion of the western United States. They are important for providing ecosystem services such as sources of clean water and air, wildlife habitat, ecosystem biodiversity, recreation, and aesthetics. The National Resources Inventory (NRI) is a primary data source for ongoing assessment of nonfederal land in the United States, including rangelands, and the data collected during an NRI assessment is typical of rangeland monitoring conducted by managers. This study outlines a methodology for using that type of monitoring data to run a rangeland hydrology and erosion model in order to estimate the relative soil erosion rates across ecosystems located in the American Southwest. The model was run on 134 NRI rangeland field locations with data collected between 2003 and 2006 in Major Land Resource Area 41, the Southeastern Arizona Basin and Range, which is a diverse ecological area of 40,765 km^2 (15, 739 mi^2) in the transition zone between the Sonoran and Chihuahuan deserts. Results of the study showed that the data collected was adequate to run the model and effectively assess the influence of foliar cover, ground cover, plant life forms, soils, and topography on current soil erosion rates. Results suggested that the model could be further improved with additional measured experimental data on infiltration, runoff, and soil erosion within key ecological sites in order to better quantify model parameters to reflect ecosystem changes and risk of crossing interdependent biotic and abiotic thresholds.

Key words: conservation—hydrology—landscapes—rangeland health—Rangeland Hydrology and Erosion Model—soil erosion

Rangelands are estimated to cover approximately 31% of the United States (Havstad et al. 2009), and developing tools for assessment of those lands is a critical resource management need. Rangelands are often characterized by dry climates and highly variable precipitation and sparse vegetation comprised mostly of grasses, forbs, and shrubs (Mitchell 2000). Historically these lands were used primarily for livestock production. However, in recent years, the broader value of these lands has been recognized and demands for multiple-use management have increased. In addition to livestock production, rangelands and rangeland watersheds are now being managed for wildlife and fishery habitat, ecosystem biodiversity, recreation, air and water quality, and aesthetics. With nearly 364 million ha (899 million ac) of rangeland in the United States (Havstad et al. 2009), the proper management of these lands is critical to issues facing many urban and agricultural areas throughout the 17 western states. Soil erosion is a key variable in determining sustainability of rangeland ecosystems and management practices. In arid and semiarid rangeland ecosystems, responses to management are usually slow and often require a decade or more to evaluate due to interannual variability in weather.

The basic landscape unit for range management in the United States by the Natural Resource Conservation Service (NRCS) is the ecological site (USDA 2003), which is considered in conjunction with associated state-and-transition models to describe vegetation dynamics and critical thresholds (May 1977; Westoby et al. 1989; Laycock 1991). An ecological site is "a distinctive kind of land with specific characteristics that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation" (USDA 2003). State-and-transition models are conceptual models that describe the long-term dynamics of an ecological site, wherein the "states" are identifiable and relatively stable groupings of plant species and "transitions" are pathways from one state to another (Westoby et al. 1989). Soil erosion has been recognized as the key process associated with degraded states (National Resource Council 1994), and since interpretations of hydrologic function and erosion history of states within ecological sites have generally been made on a subjective basis (Pyke et al. 2002), scientifically based tools and data are needed to provide a basis for understanding and quantifying erosion rates within the context of the ecological site concept.

Several legal mandates, such as the Public Rangelands Improvement Act of 1978, the Forest and Rangeland Renewable Resources Planning Act of 1974, and the Soil and Water Resources Conservation Act of 1977, require federal land management agencies to periodically report on the status of the rangelands they administer (USDA 2011). The National Resources Inventory (NRI) on nonfederal rangeland consists of on-site, random segments that are sampled with a statisti-

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cally sound approach to determine trends and conditions (Herrick et al. 2010; Nusser et al. 1998). The NRI provides a nationally consistent database that allows for resources assessment over time (Spaeth et al. 2003).

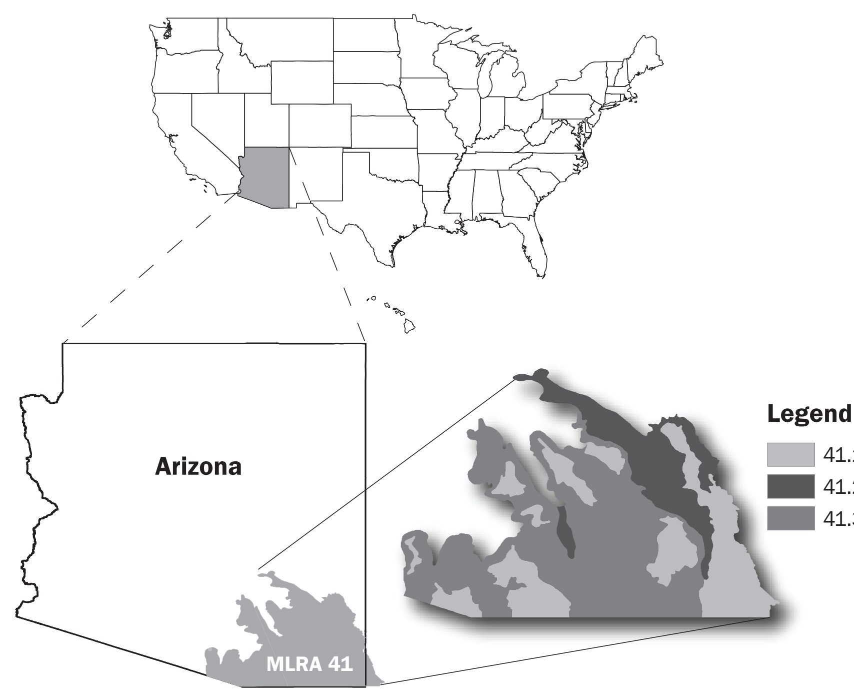
The complex problem of managing rangelands for an increasing intensity of multiple uses has made it necessary for natural resource managers to seek out new tools to assist them in making management decisions (Weltz et al. 2008, 2011; Herrick et al. 2012). Over the past 30 years, researchers have built many mathematical computer simulation models for different agricultural production systems, but only a few models have been developed specifically for rangelands (Wight 1983; Pierson et al. 1996, 2001; Carlson et al. 1993; Lane et al. 1992; Singh 1995; and Wei et al. 2009). As a consequence, modelers have attempted to apply concepts developed for more mesic cropland systems to western rangelands with little change to accommodate the unique aspects of arid and semiarid rangeland systems (Wei et al. 2009).

In response to this need, the USDA Agricultural Research Service developed a new process-based model for assessing soil erosion rates on rangelands (Nearing et al. 2011; Wei et al. 2009). The Rangeland Hydrology and Erosion Model (RHEM) is a newly conceptualized model adapted from relevant portions of the Water Erosion Prediction Project Model (Flanagan and Nearing 1995; Nearing et al. 1989; Laflen et al. 1997) and modified to specifically address rangelands conditions. It predicts soil loss based on the simulation of hydrologic and erosion processes unique to rangelands.

The main objective of this study was to develop a general strategy to utilize ground-measured NRI rangeland site data for running RHEM. This was done using an example analysis of the application of NRI and RHEM in the Major Land Resource Area (MLRA) 41 located in the Southeastern Basin and Range region of the southwestern United States. To illustrate how the information from the model can be used, we examined the influence of plant and soil characteristics on soil erosion and hydrologic function in the study area using parametric and multivariate analyses to determine differences between RHEM-simulated annual erosion rates and NRI-reported soil and vegetation conditions.

Figure 1

Boundaries of Major Land Resource Area (MLRA) 41, including Common Resource Areas (CRA) 41.1, comprising precipitation zone 406 to 762 mm, 41.2 comprising precipitation zone 203 to 305 mm, and 41.3 comprising precipitation zone 305 to 406 mm.



Materials and Methods

Study Area Description. This study was conducted in MLRA 41, Southeastern Arizona Basin and Range, which encompasses approximately 41,000 km^2 (15,830 mi^2) of southeastern Arizona (89%) and a small portion of southwestern New Mexico (11%) (figure1). MLRA 41 is a diverse ecological area in the transition zone between the Sonoran and Chihuahuan deserts, with distinctive combinations of topographies, soils, climate, water resources, and land uses that include a series of isolated mountain chains and arid river basins. There are three Common Resource Areas (CRAs) defined by the USDA (2006) within MLRA 41. The Mexican Oak-Pine Forest and Oak Savannah (CRA 41-1) occupies the higher elevations, with oak savannah and perennial grasses dominating at elevations ranging from 1,300 to 1,700 m (4,265 to 5,577 ft), where the precipitation ranges from 406 to 508 mm (16 to 20 in) per year. At elevations above 1,700 m with greater than 508 mm of precipitation, vegetation is dominated by conifer woodland. The vegetation of the Chihuahuan-Sonoran desert shrubs mix (CRA 41-2) is typified by sparse cover of perennial grasses and shrubs and only a few trees. This relatively smaller area of land consists primarily of river valleys, with elevation ranging from 880 to 1,440 m (2,887 to 4,724 ft) and annual precipitation ranging from 203 to 305 mm (8 to 12 in). The third CRA (CRA 41-3) is the Chihuahuan–Sonoran Semidesert Grassland. It is the largest and covers the midrange of elevation, from 975 to 1,500 m (3,298 to 4,921 ft), with precipitation from 305 to 406 mm (12 to 16 in) (USDA 2006).

About one-third of the study area is federally owned, and most of the area is used for livestock grazing. Relatively minor areas of irrigation are used for cotton (*Gossypium hirsutum* L.), corn (*Zea mays* L.), alfalfa (*Medicago sativa* L.), small grains, or other farm crops. The major soil resource concerns are maintenance of soil organic matter and productivity and wind and water erosion. The dominant conservation practice on rangelands is generally prescribed grazing, and supporting practices (fencing and water development) help control the distribution and intensity of grazing (USDA 2006).

National Resources Inventory Field Measurements and Data Description. MLRA 41 is characterized by 60 rangeland ecological sites. In this study, we only considered

Table 1
Ecological sites located in Major Land Resource Area 41.

IDN	Ecological site ID	Ecological site name	Soil properties	Number of NRI plots
1	XA103AZ	Limestone Hills 16 to 20 in p.z.	Yarbam-Rock outcrop complex, 15% to 65% slopes	1
2	XA111AZ	Volcanic Hills 16 to 20 in p.z.	Magoffin-Budlamp-Rock outcrop complex, 5% to 70% slopes	2
3	XA104AZ	Limy Slopes 16 to 20 in p.z.	Hathaway gravelly sandy loam, 20% to 50% slopes	1
			Carbine very gravelly loam, 3% to 30% slopes	1
4	XC301AZ	Basalt Hills 12 to 16 in p.z.	Graham-Rock outcrop complex	2
5	XC323AZ	Volcanic Hills 12 to 16 in p.z. Loamy	Rock outcrop-Atascosa-Graham complex, 9% to 70% slopes	4
			Rock outcrop-Lampshire complex, 15% to 50% slopes	2
6	XC306AZ	Granitic Hills 12 to 16 in p.z.	Romero-Rock outcrop-Oracle complex, 10% to 45% slopes	1
			Lampshire-Chiricahua association, steep	2
7	XC307AZ	Limestone Hills 12 to 16 in p.z.	Mabray-Chiricahua-Rock outcrop complex, 3% to 45% slopes	1
8	XC314AZ	Loamy Slopes 12 to 16 in	Caralampi gravelly sandy loam, 10% to 60% slopes, eroded	2
			No map unit	2
9	XC303AZ	Clayey Slopes 12 to 16 in p.z.	Limpia-Graham-Rock outcrop complex, 9% to 50% slopes	9
			Signal very cobbly clay loam, 10% to 40% slopes	2
10	XC308AZ	Limy Slopes 12 to 16 in p.z.	Powerline-Kimrose family complex, 10% to 35% slopes	2
			Hathaway soil, 1% to 40% slopes, eroded	1
11	XC309AZ	Limy Upland 12 to 16 in p.z.	Luckyhills-McNeal complex, 3% to 15% slopes	1
			Karro loam	1
12	XC322AZ	Granitic Upland 12 to 16 in p.z.	Romero-Oracle-Rock outcrop complex, 5% to 20% slopes	1
13	XC313AZ	Loamy Upland 12 to 16 in p.z.	Bernardino-Tombstone association, 5% to 16% slopes	1
			White House-Caralampi complex, 5% to 25% slopes	1
			White House gravelly loam, 0% to 10% slopes	6
			White House-Caralampi complex, 10% to 35% slopes	2
			McAllister loam, 1% to 3% slopes	2
			McAllister-Stronghold complex, 3% to 20% slopes	2
			No map unit	2
14	XC320AZ	Loam Upland 12 to 16 in p.z. Proposed ESD	Elfrida silty clay loam	1
15	XC305AZ	Clay Loam Upland 12 to 16 in p.z.	Selvin-Tombstone-Saddlebrook complex, 3% to 45% slopes	3
			Bernardino-Hathaway association, rolling	1
			White House gravelly loam, 0% to 10% slopes	1
			Libby-Gulch complex, 0% to 10% slopes	2
			Bonita-Forrest complex, 1% to 8% slopes	2
			Tubac sandy clay loam, 0% to 2% slopes	4
			No map unit	2
16	XC304AZ	Clayey Upland 12 to 16 in p.z.	Bonita very cobbly silty clay, 2% to 8% slopes	2
			Graham-Pantak complex, 2% to 15% slopes	1
			Outlaw-Epitaph-Paramore complex, 0% to 15% slopes	2

Table 1 continued

the 31 ecological sites that are represented with at least one NRI point. Table 1 lists the 31 ecological sites with their corresponding site characteristics. The NRI rangeland data used in this study represent conditions based on the data collected at 134 NRI rangeland field locations between 2003 and 2006. Two sets of indicators were extracted from the NRI database to accomplish two objectives. The first set of indicators conveys the current rangeland condition in the context of three rangeland health attributes: soil and site stability, hydrologic function, and biotic integrity (table 2). The second set of indicators provides information to estimate parameters to run RHEM and estimate erosion using the pedotransfer functions for the model (Nearing et al. 2011). These indicators include percentage foliar cover, percentage ground cover, percentage basal, percentage cryptogams, percentage litter, percentage rock fragment, and slope gradient.

Ground and foliar cover characteristics were determined from the line-point intercept protocol at 91 cm (3 ft) intervals along two intersecting 46 m (150 ft) transects. Bare ground was defined as soil that was not protected by plant bases (including lichens and moss), litter, gravel, or rocks. Intercanopy gaps were measured using the line intercepts tran-

Table 1 continued

Ecological sites located in Major Land Resource Area 41.

IDN	Ecological site ID	Ecological site name	Soil properties	Number of NRI plots
17	XC319AZ	Sandy Loam Upland 12 to 16 in p.z.	Sasabe gravelly sandy loam, 0% to 2% slopes	1
			Continental soils, 1% to 10% slopes	1
18	XC318AZ	Sandy loam 12 to 16 in p.z. Deep	Oracle-Romero-Combate complex, 1% to 20% slopes	1
			Sonoita sandy loam, 0% to 2% slopes	1
			Sonoita sandy loam, 2% to 5% slopes	2
19	XC311AZ	Loamy swale 12 to 16 in p.z.	Grabe-Comoro complex, 0% to 5% slopes	2
			Forrest silt loam, 0% to 1% slopes	2
			Forrest clay loam, 1% to 3% slopes	2
			Forrest-Bonita complex, 0% to 3% slopes	1
			River Road and Ubik soil, 0% to 1% slopes	1
			Pima loam	3
20	XC316AZ	Sandy Wash 12 to 16 in p.z.	Santo Tomas soils	2
			Keysto-Riverwash complex, 1% to 5% slopes	2
			Comoro soils, 0% to 5% slopes	1
21	XC315AZ	Saline Bottom 12 to 16 in p.z.	Cogswell clay loam, alkali	2
			Gothard fine sandy loam	2
22	XB216AZ	Clayey Slopes 8 to 12 in p.z.	Topawa-Rillino-Eba complex, 3% to 50% slopes	2
23	XB207AZ	Limy Slopes 8 to 12 in p.z.	Calcigypsids-Contention-Redo complex, chihuahuan, 5% to 45% slopes	1
			Peloncillo-Orthents-Pinaleno complex, 20% to 90% slopes	2
24	XB215AZ	Sandy Loam Upland 8 to 12 in p.z.	Tres Hermanos-Continental-Nickel complex, 2% to 45% slopes	1
25	XB204AZ	Clay Loam Upland 8 to 12 in p.z.	Tapco-Peloncillo association, 2% to 15% slopes	2
26	XB219AZ	Gypsum Upland 8 to 12 in p.z.	Contention-Whitecliff complex, eroded, 0% to 5% slopes	1
27	XB212AZ	Saline Upland 8 to 12 in p.z.	Hondale silty clay loam	4
28	XB229AZ	Limy Upland 8 to 12 in p.z. Deep	Pinaleno-Continental gravelly sandy loams, 0% to 10% slopes	1
			Pinaleno-Cave complex, 0% to 5% slopes	3
			Pinaleno-Bitter Spring complex, 0% to 5% slopes	1
29	XB206AZ	Sandy Loam Upland 8 to 12 in p.z. Proposed ESD	Tres Hermanos-Continental-Nickel complex, 2% to 45% slopes	7
			Wampoo gravelly loam, 2% to 10% slopes	4
30	XB213AZ	Sandy Bottom 8 to 12 in p.z.	Pinaleno-Continental gravelly sandy loams, 0% to 10% slopes	1
			Arizo gravelly sandy loam, 2% to 5% slopes	2
31	XB226AZ	Loamy Bottom 8 to 12 in p.z. Subirrigated	Glendale loam	1

Notes: IDN = identification number associated with the ecological site number shown in figure 9. NRI = National Resources Inventory. ESD = ecological site description.

sect protocol, an on-site method to record all foliar gaps of at least 91 cm in length along two intersecting 46 m transects. The protocol allows entry of up to six foliar and one ground cover layers. At least one entry must be made for each of these two data categories at 91 cm intervals along the transect. The NRI ground hits may include any plant species (living or dead); lichen crust on the soil; moss; rock fragments (rock fragments must be greater than 6.25 mm [0.25 in diameter], if smaller they are considered to be soil); and the term NONE, which is soil visibly unprotected by any of the above. For purposes of RHEM application, ground cover is the cover of the soil surface that essentially is in contact with the soil, as opposed to canopy or foliar cover, which is cover above the ground surface. Ground cover may be present in the form of plant litter, rock fragments, cryptogams, and plant basal areas. Note that any given point on the surface or line transect may have either, neither, or both foliar cover and ground cover present.

A comprehensive review of the NRI inventory sampling strategy is presented in Goebel (1998), Nusser at al. (1998), and Nusser and Goebel (1997). A review of new proposed NRI protocols on non-federal rangelands is presented in the National Resources Inventory Handbook of Instructions for Rangeland Field Study Data Collection (USDA 2005), and a summary of NRI results on rangeland is presented in Herrick et al. (2010).

Rangeland Hydrology and Erosion Model Concepts. RHEM computes soil loss along a slope and sediment yield at the end of a hillslope (Nearing et al. 2011). Splash and sheet erosion is described as a process of soil detachment by raindrop impact and surface water flow, transport by shallow sheet flow and small rills, and sediment delivery to larger concentrated flow areas such as arroyos. Sediment delivery rate from hillslopes is computed by using an improved equation developed by Wei et al. (2009) using rangeland runoff and erosion data from rainfall simulation experiments. Concentrated flow erosion

Table 2
National Resources Inventory measurement and indicators relevant to the rangeland health attributes.

Measurements	Indicator	Attributes: Soil and site stability	Hydrologic function	Biotic integrity
Line–point intercept	Foliar cover (%)	X	X	X
	Basal cover (%)	X	X	X
	Bare ground (%)	X	X	X
	Ground cover (%)	X	X	—
	Rock fragment (%)	X	X	—
	Cryptogams (%)	—	X	X
	Litter cover (%)	—	X	X
Canopy and basal gap intercept	Soil surface in canopy gaps (%)	X	X	X
	Soil surface in basal gaps (%)	X	X	X

is conceptualized as a function of the flow's ability to detach sediment, sediment transport capacity, and the existing sediment load in the flow. The appropriate scale of application is for hillslope profiles. Details of the model have been published (Nearing et al. 2011).

Model Parameter Estimation. The RHEM model requires 13 input parameters grouped in three categories: slope profile, soils, and climate. A list of the input parameters referred to in this paper and definitions are provided in table 3. The soils parameter group in table 3 are calculated using the pedo-transfer (parameter estimation) equations as derived by Nearing et al. (2011). An important aspect of the model relative to application by rangeland managers is that RHEM is parameterized based on four plant life form classification groups (annual grass and forbs, bunchgrass, shrubs, and sodgrass). Nearing et al. (2011) developed the equations to estimate effective hydraulic conductivity (k_e) and the splash and sheet erosion coefficient (k_{ss}) for each of the life form groups.

A computer program was developed to query the NRI database and then compute percentage foliar cover, percentage basal cover, and percentage ground cover, which were calculated based on the number of plant and basal hits recorded along the transect. In this study, foliar cover is defined as the percentage of ground covered by vertical projection of the plant canopy. Only the upper most plant species intercepted in the canopy layer was considered for calculating percentage foliar cover.

The National Plant Data Center of the USDA NRCS (USDA NRCS 2013a) was consulted to organize each plant species into the four plant life forms described above. The NRI database provides a four-letter code based on the first two letters of the genus and species, or the common name for each plant intercepted in the top canopy layer. First, we classified each plant species into plant growth forms following the 10-plant growth habitat definitions described in the PLANTS Database of the USDA NRCS (2013b). All plant species in our study area fall into two main plant growth habitat groups: shrub and graminoid. Second, we further broke down the graminoid group into bunch, annuals, and sod grasses based on the description of 71 range grasses in the state of Arizona carried out by Humphrey (1970). Third, we determined the dominant plant growth habitat at NRI sampling plots by adding the number of point intercepts of each plant growth form and dividing by the total point intercepts of vegetation in the plot. The largest proportion (%) of the various plant growth forms characterized the dominant plant growth habitat group at each NRI plot. Then the appropriate equation as described by Nearing et al. (2011) was selected to estimate k_e and k_{ss}.

Based on the map unit information available in the NRI database, a link was established to the NRCS Soil Survey Geographic Database (SSURGO), which is available online at http://soildatamart.nrcs.usda.gov (USDA NRCS 2012). The soil database was queried (on June 9, 2012) to calculate weighted average estimates of sand and clay percentage over the soil layer depth at 5 cm (2 in). The soil survey area symbol and map unit symbol information from the NRCS SSURGO database was used to obtain weighted average estimates of sand and clay at 0 to 5 cm (0 to 2 in) soil depth, depth of soil surface layer, and the Universal Soil Loss Equation slope length. The basic structure of the SSURGO database tables is a one-to-many hierarchy, where a survey area has multiple map units, a map unit can have multiple soil components, and a soil component can have multiple soil horizons.

The Climate Generator model (CLIGEN V5.101) was used to generate a synthetic, statistically representative series of precipitation data for a period of 300 years based on measured data from the nearest weather station to a given NRI point in MLRA 41 (Nicks et al. 1995). Such a long series of data was used to make certain that average annual soil loss stabilized to a relatively constant value (defined in terms of the cumulative average

Table 3
Descriptions of the primary Rangeland Hydrology and Erosion Model inputs.

Input group	Parameter name	Units	Description
Slope profile	Slp	%	Slope gradient
	Slplen	m	Slope length
Soils	k_e	mm h^{-1}	Effective hydraulic conductivity
	k_r	s m^{-1}	Concentrated flow erosion coefficient
	Tc	N m^{-2}	Critical shear stress
	k_{ss}	ND	Splash and sheet erosion coefficient
	Fr	ND	Friction factor for runoff
	Fe	ND	Friction factor for erosion
	Psd	ND	Particle size distribution
Rainfall	Rain	mm	Rainfall volume
	Dur	hr	Rainfall duration
	Ip	ND	Normalized peak rainfall intensity
	Tp	ND	Normalized time to peak intensity

not fluctuating more than plus or minus 10% with continued duration of the simulation) (Baffaut et al. 1996). The CLIGEN simulations produced values for rainfall amount (mm), rainfall duration (h), relative time to rainfall peak (dimensionless), and relative maximum intensity (dimensionless).

Data Analysis. The data was divided in several ways for comparative analysis. Average erosion values were computed and reported for each of the 31 ecological sites represented in the data. Comparisons were made between groupings of ecological sites by the 4 life form groups. In addition, for 3 of the ecological sites that were represented by a relatively larger number of samplings, more detailed analysis was possible.

Pearson's correlation coefficients were used to assess the linear correlation association of the variables and one way analysis of variance (ANOVA) for testing the overall nonlinear relationship. The testing procedure used for this null hypothesis involves the overall F-test, that is, probability of accepting the null hypothesis $p < 0.05$ and probability of rejecting the null hypothesis $p > 0.05$. The significance of the differences in mean among the four plant life form types was tested using ANOVA. Mean separation with the Fisher's Protected Least Significant Difference test was used ($\alpha = 0.05$). The uncertainty in the determination of the mean was estimated as

$$SE = \frac{standard\ deviation}{\sqrt{N}}, \quad (1)$$

where N is the number of data points.

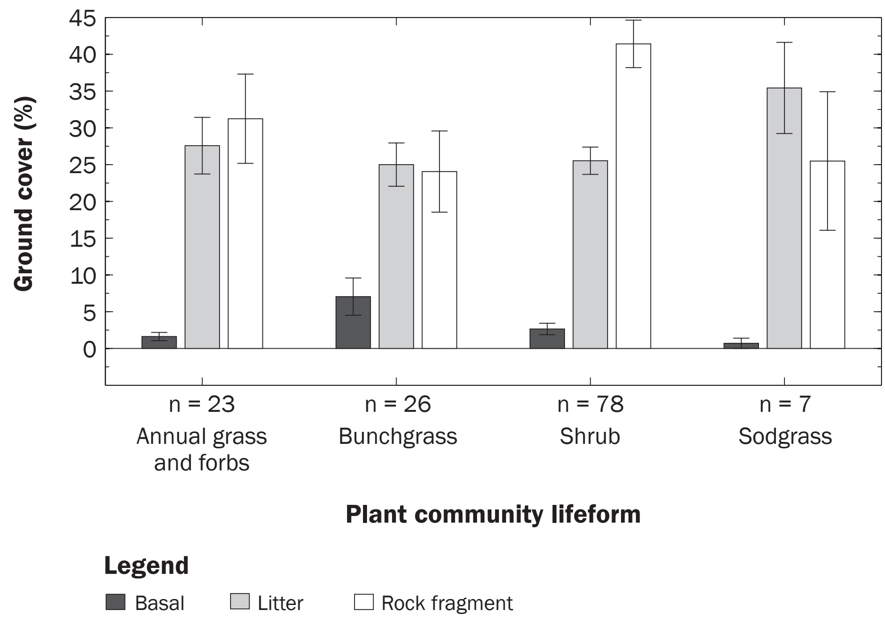
Results and Discussion

Interpretation of the National Resources Inventory Cover Data. Differences in the means of measured ground cover between the four plant life form types were found with surface rock fragment and basal area cover. Conversely, the means of litter cover for all the four plant life form types were not significantly different from each other ($p > 0.05$) (figure 2). Average rock fragment cover was significantly greater ($p < 0.05$) for the shrub life form group compared to the others. This is consistent with previous findings that indicate the occurrence of greater rates of past soil erosion in degraded shrub areas (Nearing et al. 2005). Where soils contain rock fragments within their profile, the fragments become concentrated on the surface as a result of the preferential removal of fine material, resulting in the formation of a lag gravel pavement (Cooke and Warren 1973; Simanton et al. 1994). Runoff flow velocities, and thus erosion and transport capacities, tend to increase with the increasing slope gradient. An increase in sediment transport capacity increases the amount and size of soil material that are transported by runoff, which leads to the relationship of increased surface rock fragment cover with increasing slope gradient. Simanton et al. (1994) and Nearing et al. (2005) found positive correlations between slope gradient and soil profile rock fragment along catenas on the Walnut Gulch Experimental Watershed, Arizona, and Parsons and Abrahams (1987) reported that the mean diameter of soil surface particles on Mojave Desert debris slopes was positively correlated with slope gradient. In the current study, we also found that there was a significant correlation ($r = 0.73$, $p < 0.0001$) between slope gradient and surface rock fragments from the 134 NRI plots (figure 3).

The bunchgrass had the largest average percentage basal cover among the four plant life forms. Grasslands are characterized by having a diffuse distribution of basal areas causing overland flow to be slower and less concentrated. Sediment is thus deposited behind the bases of the plants, often forming a series of microtopographic terraces, which in turn enhance the diffusion of the overland flow (Parsons et al. 1996). In addition, it has been reported in the literature that the rate of infiltration into the soil is often higher around plant bases, due to root channels and the activity of soil organisms (Weltz et al. 1998; Whitford 2002). Hence, we might argue that soil erosion should be low in ecological sites primarily dominated by bunchgrass, and conversely, higher erosion rates should be expected in ecological sites primarily dominated by shrubs, which is consistent with erosion data from the area (Nearing et al. 2005).

The mean values of foliar cover for bunchgrass and annual grass and forbs were significantly greater than the mean for shrubs ($p < 0.05$) (figure 4). The means of bare ground percentages for all plant groups were not significantly different ($p > 0.05$). However, the means of basal gap of the perennial grasses were significantly less than for the shrub and annual grass and forbs (figure 4). Under conditions with similar plant types, a reduction in total plant canopy cover will usually increase the area encompassed by larger gaps. The distance between plant bases generally increases when basal cover declines, which usually occurs when shrubs replace bunchgrass. The spatial pattern of vegetation is thought to be correlated with soil and site stability, hydrologic function, and biotic integrity (Pellant et al. 2005; Okin et al. 2009).

Figure 2
The mean percentage of ground cover (± standard error) for the different plant life forms: annual grass and forbs, bunchgrass, shrubs, and sodgrass.



While the measured canopy gap percentage does not completely characterize the spatial patterns, it does provide an indication of the extent to which plant cover is aggregated (forming a few large gaps) or dispersed (forming many small gaps). Thus, the proportion of a transect line covered by canopy gaps exceeding 30 cm (1 ft) is considered to be a useful indicator to help determine the status of the hydrologic function and biotic integrity attributes and, consequently, soil erosion (Pellant et al. 2005). This indicator can vary across sites within the same total foliar cover, depending on how the vegetation is arranged, that is, large canopy gaps or small canopy gaps. Larger gaps also generally indicate greater spatial variability in soil organic matter since organic matter generally decreases further from vegetation. This means that soil structure is typically poorer in large gaps than in small gaps, and consequently, soil in the gaps may be more erodible.

There is an argument for the idea that soil erosion may be accelerated in areas with large gaps between vegetation because these gaps tend to be more highly connected, with less vegetation obstructions to water flow (Okin et al. 2009). This would result in greater erosive forces and also means that once a soil particle is detached, there is little to prevent it from continuing to move downslope. This process may result in an island effect in which excessive soil loss occurs in the interspace area where runoff is concentrated. It has been argued that the soil erosion–site degradation process can be accelerated in these situations and result in loss of biotic integrity, desertification, and sustainability of the site (Schlesinger and Pilmanis 1998; Schlesinger et al. 1990, 1996). Examples of areas that have large gaps in vegetation and patchiness are seen in shrub dominated landscapes, which have formed coppice dunes (i.e., sagebrush, creosotebush [*Larrea tridentate* (DC.) Coville], and mesquite [*Prosopis juliflora* (Sw.) DC.]) in woodlands where juniper and pinyon pine have invaded sagebrush steppe communities in arid and semiarid rangelands (Pierson et al. 1994, 2011; Davenport 1998; Spaeth et al. 1994) and degraded bunch/tussock grasslands. Tongway and Ludwig (1997) found that on degraded grasslands flow was concentrated in long straight paths between the grasses. In the good condition grassland, water flow was tortuous and uniformly distributed.

In the Loamy Upland 305 to 406 mm (12 to 16 in) precipitation zone (p.z.) ecological site, represented by 16 NRI plots,

Figure 3

Relationship between surface rock fragment cover and slope gradients for all the National Resources Inventory points.

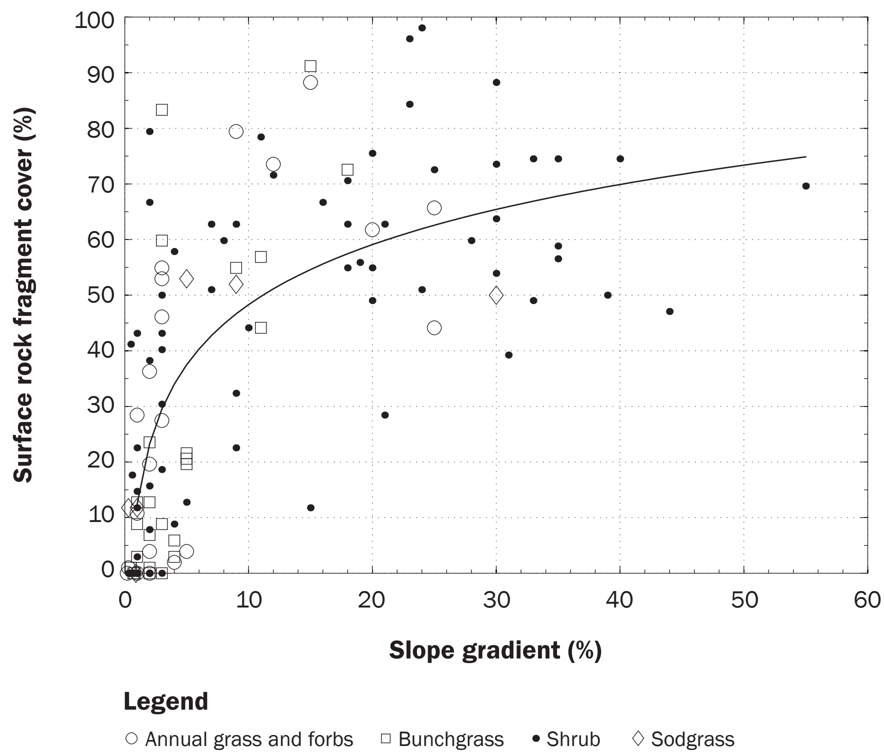
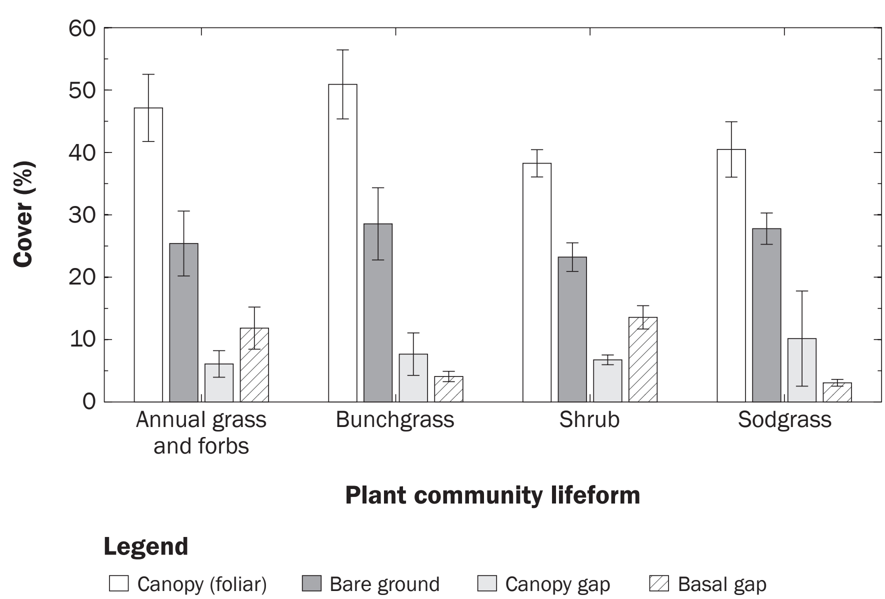


Figure 4

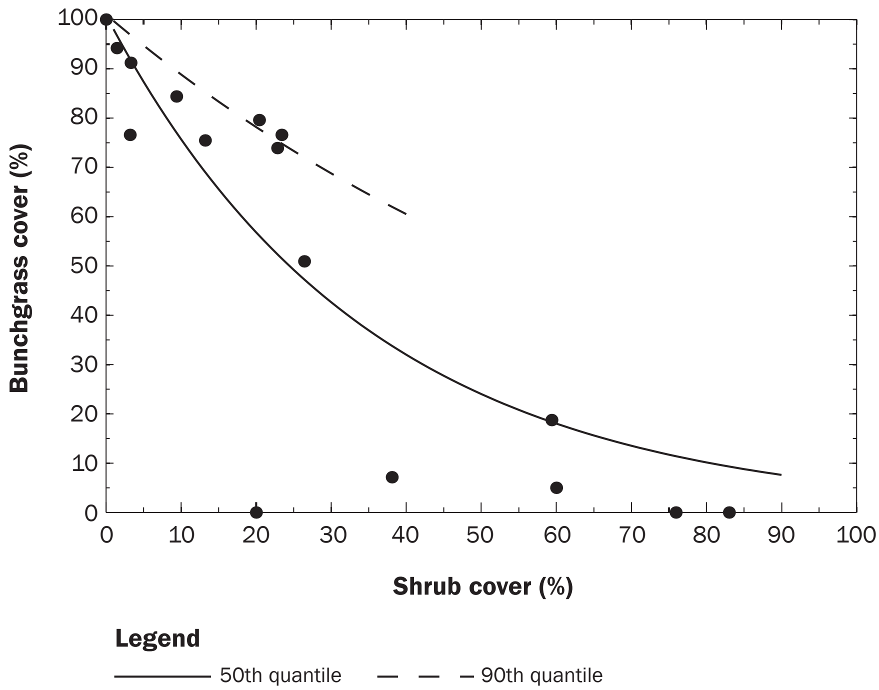
The mean percentage of line foliar cover, bare ground, canopy and basal gaps (± standard error) for the four plant life form communities.



quantile regression analysis revealed that dominant perennial bunchgrass cover had a negative exponential relationship with the amount of shrub species cover at the 50th (median) quantile (figure 5). The perennial dominant bunch grasses included blue grama (*Bouteloua gracilis*), lehmann lovegrass (*Eragrostis lehmanniana* Nees), slender grama (*Bouteloua repens* (Kunth) Scribn. and Merr.), and Rothrock's grama (*Bouteloua rothrockii* Vasey), and the dominant shrub species included honey mesquite (*Prosopis glandulosa* Torr.) and mesquite. Beyond approximately 30% shrub cover, bunchgrass cover was not observed to be higher than approximately 20%. Moreover, the quantile regression analysis revealed that the upper bound on bunchgrass cover (>70%) and lower bound on shrub cover (<30%) also were correlated at the 90th quantile. Points falling far below the upper bound at the low shrub cover values might represent plant communities at risk of degradation. The points that define the upper bound, on the other hand, might represent the upper potential for bunchgrass cover for the site. Too strong a conclusion is not justified considering the small sample size (n = 16) and a large sampling variation for the upper quantiles.

Figure 5

Relationship between percentage foliar cover of shrubs and perennial bunchgrass cover in the Loamy Upland (12 to 16 in p.z.) ecological site. The solid line is fit to a negative exponential function for the 50th quantiles and the dashed line to the 90th quantiles.



Interpretation of the National Resources Inventory Soil Data. Soil aggregate stability is recognized as a key indicator of soil quality and rangeland health (USDA 1996). Increases in stability are thought to reflect increased soil erosion resistance. Sites with average values of five or above generally are considered to be very resistant to erosion, particularly if there is little bare ground and there are few large vegetation gaps. Maximum possible soil stability values may be less than six for very coarse sandy soils. High soil stability values usually reflect good hydrologic function. This is because stable soils are less likely to disperse and clog soil pores during rainstorms. High stability values also are strongly correlated with soil biotic integrity (Herrick et al. 2005).

Reported soil stability values ranged from 1 to 6 over the 134 NRI points. The median soil stability varied among the four plant life form types between 3.5 and 4; the median lowest value was observed in the sodgrass and annual grass and forbs sites and the median highest value was observed in the shrubs and bunchgrass (figure 6). Furthermore, the high variability of soil stability, which is typical of rangeland settings, reflects spatial variability in organic matter inputs and aggregation and degradation processes. A positive linear correlation (r = 0.45, p < 0.0001) was observed between percentage basal cover plus percentage litter cover and soil stability, suggesting that soil stability was high where organic matter tended to concentrate and raindrop impact was less. Values were relatively lower (many below 4) in bare areas. A negative correlation (r = –0.26, p = 0.0024) was observed between percentage bare ground and soil stability. This indicates that erosion susceptibility is likely to increase in the plant interspaces. There was 1 value of 6 from the shrub plant community sites. This value is high for shrub-dominated ecosystems; however, the point is situated in the Saline Upland 203 to 305 mm (8 to 12 in) p.z. ecological site, which comprises the soil map unit Hondale silty clay loam, and the reported percentage clay in the soil was 35%. The second point of interest is the value of 1 in the bunchgrass sites, which was low for the bunchgrass dominated grasslands. In this case, the percentage clay in the plot was 10%, and the plot was situated in the Loamy Swale 305 to 406 mm (12 to 16 in) p.z. ecological site, which includes the soil map unit Riveroad and Ubik soils and 0% to 5% slopes. In these two plots, plant foliar gap and basal gap in the shrub site were greater than in the bunchgrass; however, the percentage clay difference between these two sites was 25%. These two plots illustrate the importance of interpreting soil stability values in the context of vegetation cover, foliar canopy and basal gap distance, and soil texture.

Simulated Soil Erosion by Life Form. The greatest average annual simulated erosion values were estimated in the annual grass and forbs and shrub plant community groups, but average annual soil erosion was not significantly different (p > 0.05) among the four plant life forms (figure 7). The lack of overall differences is not surprising given the high variability in other factors, such as slope gradient and soil properties. In the shrub case, where litter cover was observed to be low on average (figure 2), we might have expected to see higher mean erosion values. However, the greater rock fragment cover (figure 2), as represented in the erosion model, is effectively protecting the underlying soil from erosive forces and accounts for the lower than otherwise expected average erosion in the shrub. As discussed above, this erosion pavement is a product of high rates of erosion in the past, but the pavement itself now is protecting

Figure 6
The distributions of the site-median rangeland health soil stability indicator of annual grass and forbs, bunchgrass, shrubs, and sodgrass plant community life forms.

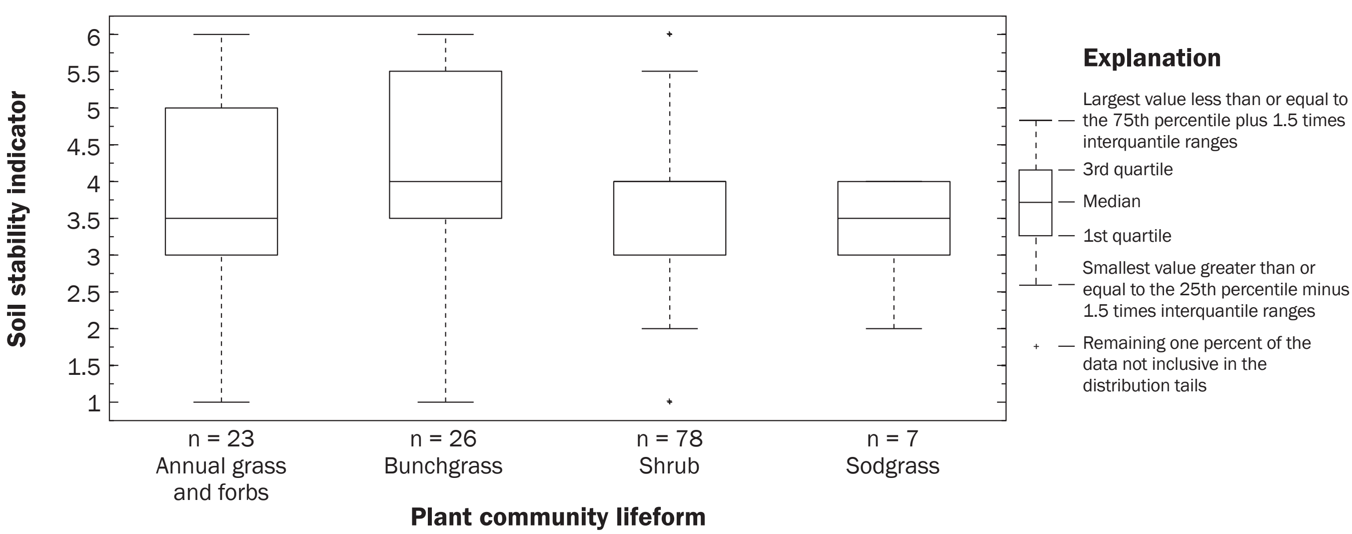
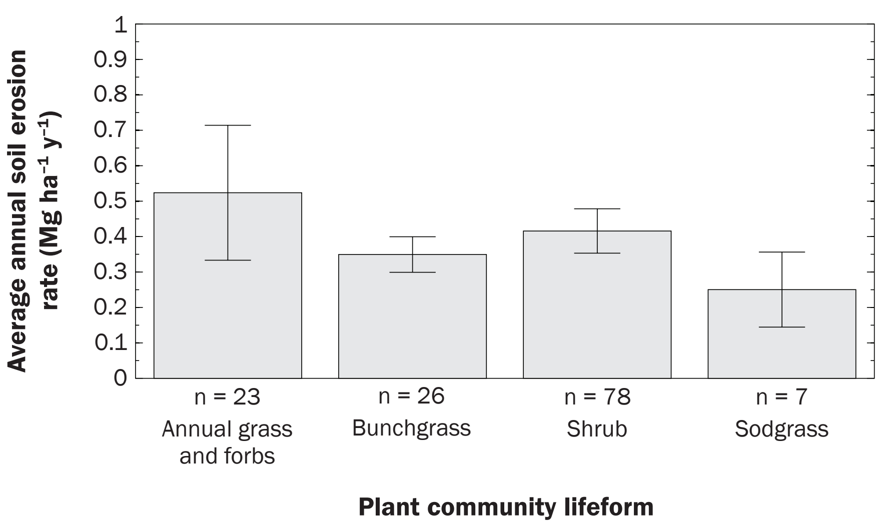


Figure 7
The mean of average annual soil erosion rate (± standard error) analyzed by plant community life form.



the surface and has reduced the erosion rates from their previous levels. This effect is represented in the model results by way of the surface rock cover term.

Simulated Soil Erosion by Ecological Sites. The variability of average annual erosion among ecological sites is shown in figure 8. The maximum value was found in the Granitic Hills 305 to 406 mm (12 to 16 in) p.z. ecological site (figure 8, identification number = 6) with 3 NRI plots. The NRI plot that produced the largest average annual erosion has soil-plant characteristics as follows: a soil stability test at the 50th quantile value of 4, foliar cover of 47%, ground cover of 76%, sand percentage of 46%, clay percentage of 9%, and slope gradient of 39%. The plot contained velvet mesquite (17%) (*Prosopis velutina*), catclaw acacia (15%) (*Acacia greggii* var. greggii), and turpentine bush (8%) (*Ericameria laricifolia*). This observation is consistent with description of alternative states in the state-and-transition model for this ecological site. The Granitic Hills 305 to 406 mm p.z. features one state with mimosa and mesquite with 10% to 35% foliar cover. Hence, the model results suggest that this NRI plot might be characterizing one of the states in the state-and -transition model of the Granitic Hills 305 to 406 mm p.z. ecological site. Furthermore, the high variability in the Granitic Hills 305 to 406 mm p.z. is attributed to the low slope gradient in 2 of the 3 NRI plots within this ecological site, which were reported having values of 7% and 9%. Based on the ecological site description report, these slope values are outside the slope range that characterizes the hills landform (15% to 70%). The second and third highest average annual erosion values occurred in NRI plots located within the ecological site Granitic Upland 305 to 406 mm p.z. (figure 8, identification number = 12) with 1 NRI plot and Clay Loam Upland 305 to 406 mm p.z. (figure 8, identification number = 15) with 15 NRI plots, respectively. Both NRI plots were dominated primarily by shrubby plants and exhibited steep slope gradients, 33% and 35%, respectively.

Overall, the simulated average annual erosion among the 31 ecological sites conveys the typical probability distribution pattern as characterized by landform and topographic features. The average annual erosion plotted on log-normal probability coordinates shows

Figure 8

Average annual soil erosion rates and their variabilities for the ecological sites. Ecological sites were ordered based on precipitation zones (p.z.) and landform. See ecological site properties based on the identification number (IDN) shown in column 1 of table 1.

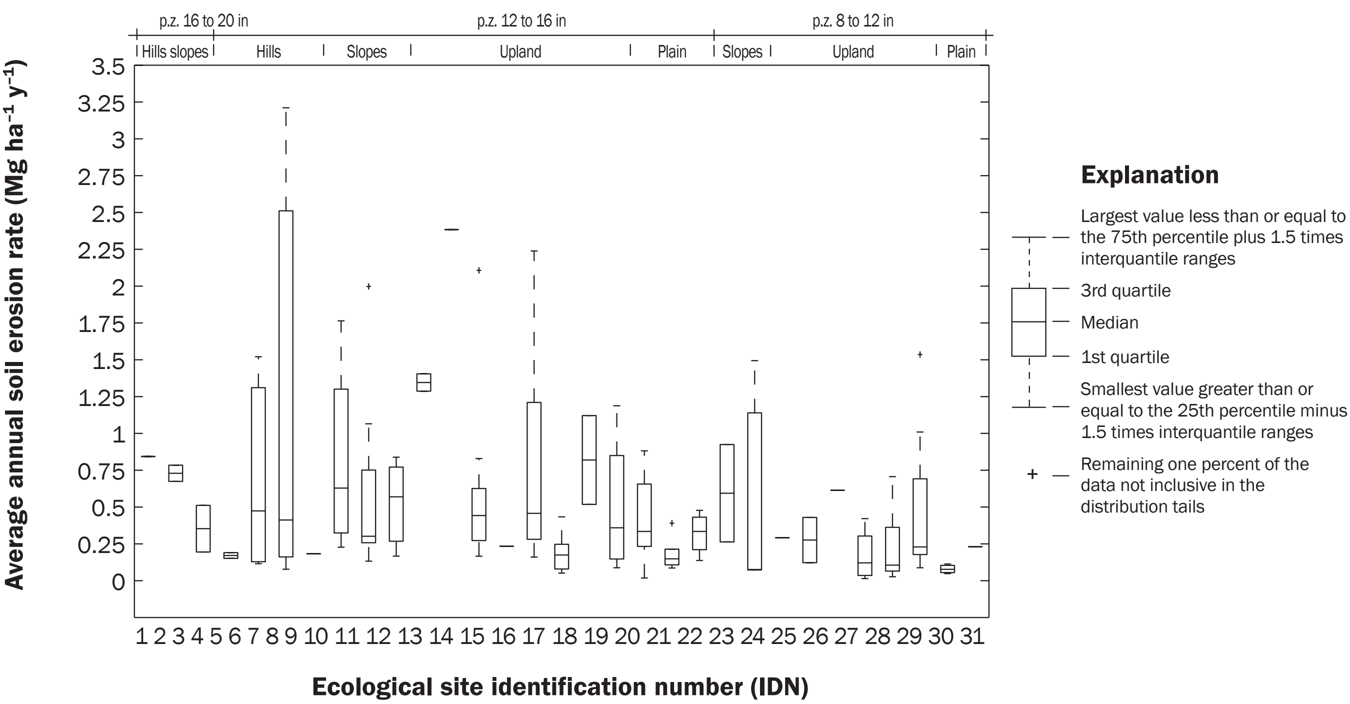
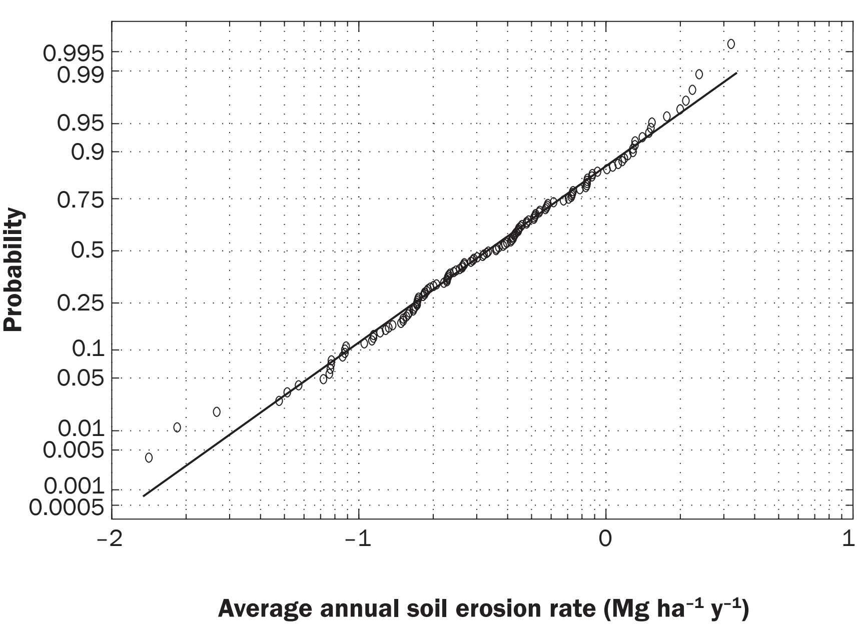


Figure 9

Simulated average annual soil erosion rate data for the 31 ecological sites in Major Land Resource Area 41 on lognormal probability coordinates.



a linear relationship (figure 9). The geometric mean and the multiplicative standard deviation of the lognormal fitted distribution are 0.33 Mg ha^{-1} and 2.74 Mg ha^{-1} (0.15 tn ac^{-1} and 1.22 tn ac^{-1}), respectively. The one standard deviation interval from 0.12 Mg ha^{-1} to 0.9 Mg ha^{-1} (0.05 tn ac^{-1} to 0.40 tn ac^{-1}) covers approximately 70% for the probability density data. The highest average annual erosion value is outside of the area represented by the two standard deviations upper boundary (2.5 t ha^{-1} [1.12 tn ac^{-1}]). Both the Kolmogorov-Smirnov and the Shapiro-Wilk (Shapiro and Wilk 1965) tests were consistent with the hypothesis that the average annual erosion is lognormally distributed. Hence we conclude that the average annual erosion for MLRA 41 can be represented by a two-parameter lognormal probability distribution.

Simulated Soil Erosion within Ecological Sites. The effect of the indicators relevant to the rangeland health attributes on average annual erosion rate was investigated within 3 ecological sites with more than 10 NRI plots. The correlation coefficients (ρ) are reported in table 4. The analysis revealed that the effect of an increase in the indicator on erosion was not the same across the selected ecologi-

Table 4

Correlation (ρ) and *p*-values between the nine indicators calculated for this study and soil erosion rates for the three ecological sites with more than 10 NRI points.

Indicators	Loamy upland 12 to 16 in p.z. R041XC313AZ IDN = 13 *n* = 16 ρ	*p*-value	Clay loam upland 12 to 16 in p.z. R041XC305AZ IDN = 15 *n* = 15 ρ	*p*-value	Clayey slopes 12 to 16 in p.z. R041XC303AZ IDN = 9 *n* = 11 ρ	*p*-value
Basal cover (%)	−0.33	0.2155	−0.08	0.7744	−0.45	0.1667
Foliar cover (%)	−0.31	0.2394	−0.27	0.3397	−0.15	0.6498
Ground cover (%)	−0.49	0.0538	+0.21	0.4441	−0.81	0.0026
Litter cover (%)	−0.12	0.6659	−0.68	0.0049	−0.32	0.3407
Rock cover (%)	−0.26	0.3230	+0.74	0.0016	−0.31	0.3596
Canopy gap at the 95th quantile (%)	+0.42	0.1072	+0.43	0.1085	+0.07	0.8287
Basal gap at the 95th quantile (%)	+0.72	0.0017	+0.29	0.3030	+0.24	0.4832
Clay (%)	−0.36	0.1744	−0.17	0.5420	−0.10	0.7754
Slope (%)	+0.55	0.0271	+0.75	0.0012	+0.45	0.1628

Notes: IDN = identification number. See table 1 for ecological site properties. *n* = number of National Resources Inventory plots in ecological sites.

cal sites. For example, an atypical result was that ground cover and rock fragment were related positively to average annual erosion in the Clay Loam Upland 305 to 406 mm (12 to 16 in) p.z. ecological site (identification number = 15 in figure 8). Though the relationship between rock fragment cover and erosion can be complex (Poesen et al. 1994; Poesen and Ingelmo-Sanchesz 1992), in this case the reported erosion rates are model results, indicating that the factors most affecting the model results for this ecological site were something different than rock fragment cover. In this case, the high correlations were with slope gradient and litter cover.

The analyses revealed that the strongest correlations occurred among average annual erosion and the following indicators: ground cover, litter, rock fragment, slope steepness, and basal gap at the 95th quantile. Basal gap at the 95th quantile varied in the three ecological sites studied as follows: from 1 to 24 m (3 to 79 ft) in the Loamy Upland 305 to 406 mm (12 to 16 in) p.z., from 1 to 45 m (3 to 148 ft) in the Clay Loam Upland 305 to 406 mm p.z., and from 6 to 39 m (20 to 128 ft) in the Clayey Slopes 305 to 406 mm p.z. The result relative to basal gap is particularly interesting because, as we have seen, the model parameters themselves are not directly calculated based on basal gap (table 5); hence this indicates a secondary relationship via other parameters rather than a direct effect on the calculation of the model inputs.

More detail regarding the model response and sensitivity can be understood by looking at the data for the Loamy Upland 305 to 406 mm (12 to 16 in) p.z. ecological site, which we looked at in detail previously with regard to foliar cover (figure 5). Note that one data point was not used in this analysis because the slope was particularly high compared to the other data, which caused the model to calculate a high erosion rate due to rill flow detachment. That was not an unreasonable model response, but it masks the relationships of cover on simulated erosion relative to the other data points. The relationships between simulated erosion and foliar and ground cover were strong (figure 10), as expected. Both ground cover (either as a total or parts thereof) and foliar cover are variables that are directly used to calculate the primary model input parameters k_e and k_{ss} (table 5). It is worthwhile to note that both the level of fit (r^2) and the slope coefficient for ground cover were greater than those for foliar cover. This is a good model response, as we certainly expect erosion to be more responsive to ground cover than to foliar cover.

On the other hand, there were no statistically significant relationships between simulated erosion rates and either basal or canopy gap for the data within the Loamy Upland site (figure 11). This reflects the fact that these variables are not used directly to calculate model input parameters (table 5). We currently do not have the measured data for soil erosion and runoff relative to that transition that allows for incorporation of basal and canopy gap into the model parameter estimation equations.

Table 5

Multiple regression equations for estimating K_e (mm h^{-1}) and K_{ss} (dimensionless) for bunchgrass (*bg*), annuals and forbs (*af*), shrub (*sh*), and sodgrass (*sg*) communities as a function of canopy cover (*cancov*), ground cover (*gcover*), rock cover (*rokcov*), litter (*litter*), and clay (*clay*). All independent variables are in decimal fraction. Equations taken from Nearing et al. (2011).

Life form groups	Equation
Bunchgrass	$log(k_{ebg}) = 0.174 - (1.450clay) + (2.975gcover) + (0.923cancov)$ $k_{ssbg} = 10^{(3.13 - 0.506litter - 0.201cancov)}$
Annuals and forbs	$log(k_{eaf}) = \log(k_{ebg})$ $k_{ssaf} = k_{ssbg}$
Shrub	$log(k_{esh}) = 1.2\ log(k_{ebg})$ $k_{sssh} = 10^{(4.01 - 1.18rokcov - 0.982(litter + cancov))}$
Sodgrass	$log(k_{esg}) = 0.8\ log(k_{ebg})$ $k_{sssg} = 1.5\ k_{ssbg}$

Figure 10

Relationships between average annual soil erosion rate and (a) percentage foliar cover, and (b) percentage ground cover along the line transects for the Loamy Upland 305 to 406 mm p.z. ecological site (identification number = 13).

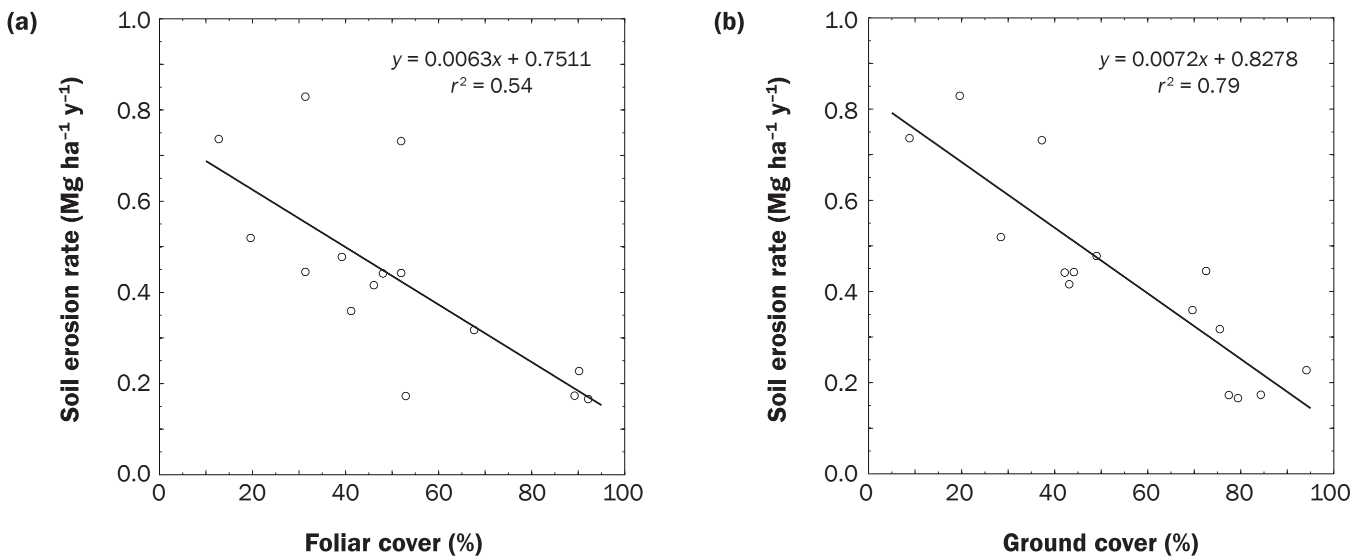
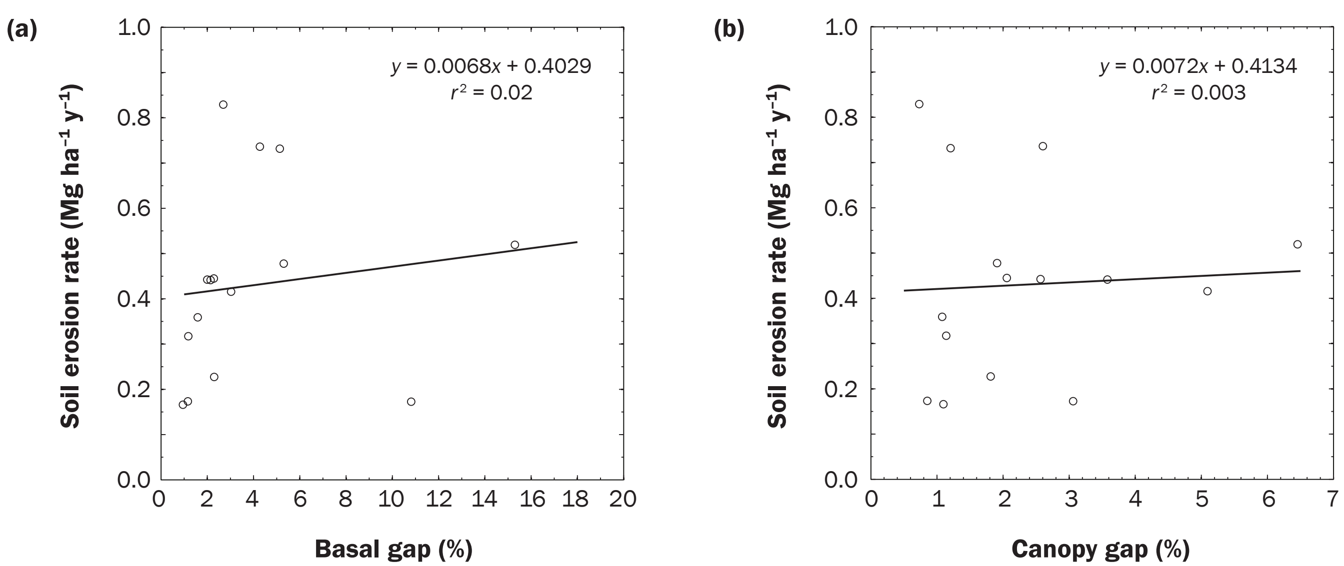


Figure 11

Relationships between average annual soil erosion rate and (a) percentage basal gap, and (b) percentage canopy gap along the line transects for the Loamy Upland 305 to 406 mm p.z. ecological site (identification number = 13).



Summary and Conclusions

This study presents a modeling approach to simulating soil erosion rates for rangeland monitoring data such as that collected for the NRCS NRI rangeland sites. The results of this work have the potential to provide land managers with a tool for predicting ecosystem hydrologic and erosional response using NRI-type field-based measurements and the soil erosion model RHEM. The results point out potential for improvement in the ability of the model, and particularly the model parameterization, to characterize plant community structure for identifying states and predicting thresholds in and between shrub and grass areas. Additional data collection would be required in order to quantify, for example, the effects of changing basal gap on soil erosion rates by water, if or when they exist. Improvements in model response to site ecological changes and identification of sites and phases that are at risk of crossing biotic or abiotic thresholds might also be a priority for improving the modeling of rangeland soil erosion by water. Nonetheless, these results suggest that managers can effectively use data from the NRI on foliar cover, ground cover, plant life form, soils, and topography to use RHEM.

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