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Quantifying decadal-scale erosion rates and their short-term variability on ecological sites in a semi-arid environment

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ABSTRACT

Soil erosion rates on six semi-arid loamy upland rangeland sites located in southeastern Arizona were measured using a rainfall simulator and ^{137}Cs fallout methods. Site characteristics that have the greatest effects on soil erosion and runoff were identified. Long term (50 years) soil erosion rates as estimated using ^{137}Cs method varied between 5.1 and 11.0 Mg ha^{-1} y^{-1} and showed significant differences between Historic Climax Plant Community and Mesquite/Native states within the State and Transition Model. Erosion rates under simulated rainfall were measured between 0.9 and 17.2 g m^{-2} min^{-1} at 175 mm h^{-1} precipitation across all sites and varied as much as 8-fold at the same location, depending on the time of the simulation. Temporal variability of erosion rates within a site was in some cases much greater than inter-site differences. This variability was attributed to natural or management driven changes in plant community and soil characteristics. Bare soil area, an aggregate indicator for all types of cover combined, was the main controlling factor of erosion process across ecological sites. For meaningful interpretation rainfall simulation, results must be placed in the context of the range of possible vegetation and surface conditions within a given ecological site.

1. Introduction

Soil erosion affects the functioning of rangeland plant communities and has in many areas negatively impacted their ability to produce forage for stock and wildlife. The lack of quantitative data on soil erosion rates at the hillslope scale in arid rangelands hinders the development of erosion prediction tools and conservation efforts in the western United States. In addition, these data are necessary to better understand the effects of erosion on rangeland sustainability in the context of ecological sites within the State and Transition Models (STM).

An ecological site is a landscape unit defined by a unique set of physical characteristics including climate, soil, and topography, which supports a range of possible plant communities different from those found on other landscape units (USDA, 2003; Briske et al., 2005). Ecological sites are convenient conceptual entities for implementing management decisions. The State and Transition Model (STM) is a conceptual model describing the characteristics and temporal dynamics of ecological sites in response to disturbances (Westoby et al., 1989; Stringham et al., 2003). An ecological site within the STM can either remain stable when disturbances are minor, or transition into other states when disturbances are significant.

There are structural and functional thresholds that exist between different states within the STM (Briske et al., 2005). Functional thresholds defined by ecohydrological processes have been studied using rainfall simulation (Chartier and Rostagno, 2006; Petersen and Stringham, 2008) but have not been quantified. It is unknown whether these thresholds necessarily exist between all states of an STM (Stringham et al., 2003) or whether erosion rates necessarily differ between different states. Once the threshold is exceeded, an ecological site may undergo a transition to another state. Soil erosion is recognized as one of the key factors of this process; however the extent of its influence in comparison with other factors is not well understood. Erosion has the potential to remove organic matter and other nutrients from the soil, and also reduces its water holding capacity and fertility. Experimental data are needed to quantify erosion thresholds and provide understanding of the role of soil erosion within STM dynamics.

Rainfall plot experiments on rangelands have shown that state transition within ecological sites may be triggered by wildfire, drought, or invasive species (Chartier and Rostagno, 2006; Petersen and Stringham, 2008; Chartier et al., 2011). In addition, these disturbances may lead to changes in basic erosion mechanisms. Namely, raindrop detachment and short transport distance on undisturbed rangelands (Parsons et al., 2006) is succeeded as a dominant process by sheet and concentrated flow detachment on degraded sites (Petersen and Stringham, 2008).

It has been shown that in an arid or semi-arid grassland community a rapid increase in erosion rates due to decline in canopy and litter cover may occur during transition from native to invasive species (Polyakov et al., 2010b) after which the equilibrium is restored. The use of soil tracers such as ^{137}Cs may provide information on whether these short transition periods make significant contribution to overall soil loss at

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longer temporal (~60 years) scales. ^{137}Cs is an artificial radionuclide with half-life of 30.2 years introduced into the environment through fallout of atmospheric atomic weapons tests from the mid-1950s through 1963 (Walling and He, 2000). The technique is based on the property of ^{137}Cs to strongly and irreversibly adsorb onto clay particles in the upper soil layer. As a result, soil redistribution due to erosion can be estimated from a ^{137}Cs budget relative to a reference inventory using a variety of conversion models (Walling and He, 1999). Erosion and deposition rates calculated by this method represent time-integrated average rates since the peak of ^{137}Cs fallout in 1963. The ^{137}Cs activity in soil has been widely used to study soil redistribution (Ritchie and McHenry, 1990) including that on semiarid croplands where vegetation was one of the main controlling factor of soil erosion (Sadiki et al., 2007) and on arid rangelands (Chappell, 1999; Nearing et al., 2005; Ritchie et al., 2005; Foster et al., 2007).

Despite the evidence that important interactions and feedbacks exist between vegetation change and erosion at different spatial (Chartier and Rostagno, 2006; Polyakov et al., 2010b) and temporal scales, quantitative information on the erosion dynamics, thresholds and transitions between different states within STMs is limited. The purpose of this study was to: a) quantify and compare runoff and erosion rates, as measured with a rainfall simulator and estimated using ^{137}Cs, on several semi-arid loamy upland rangeland sites located in southeastern Arizona in the context of ecological states and transition thresholds; b) determine which site characteristics have the greatest effects on soil erosion rates at these sites.

2. Methods

2.1. Location and site characteristics

Six ecological sites (R041XC313AZ) that belong to Major Land Resource Area (MLRA) 41-3 Loamy Upland (NRCS, 2013) were selected for the study (Fig. 1). Five sites are located at the historic Empire Ranch northeast of Sonoita, AZ and one site in the San Rafael Valley east of Patagonia, AZ (Table 1). The Historic Climax Plant Community (HCPC) encompasses three of the sites (Willow, ER2, and ER5). HCPC is comprised of plants adapted to natural disturbances including fire, while able to maintain natural equilibrium. Plant community in this state is represented by grasses of genera *Bothriochloa*, *Bouteloua*, *Ergrostis* and *Aristida* and native forbs. ER4S, ER4G, and ER3 sites are classified as Mesquite Natives Community. ER4S and ER4G sites are located in close proximity to each other and are very similar with respect to ecological characteristics.

Historically, the Empire Ranch has been heavily grazed, although the timing and extent are poorly documented. The ER5 site has been excluded from grazing since the mid-1980s but was grazed prior to that time. The ER2 was heavily grazed until the mid-2000's, and a wildfire swept through the area in 2000. The ER4S has established mesquites on the plots and the mesquites on ER4G had been mechanically removed. The Willow grassland location was burned by wildfires in 2005 and has regained vegetation cover.

The study area has a semi-arid climate dominated by the North American Monsoon (Sheppard et al., 2002). Precipitation is highly spatially and temporally variable with a pronounced peak in July through mid-September and a lesser increase in December through March. The annual precipitation at the Empire Ranch sites ranges between 300 and 400 mm y^{-1} and at San Rafael Valley is 450 mm y^{-1}. The average daily temperatures are 24 °C in July and 10 °C in January. The soils at all of the sites are gravely loams and belong to the White House (fine, mixed, thermic, Ustollic Haplargids) soil series (McGuire and Robinett, 2003). They were formed on alluvial fans and are characterized by a shallow A horizon underlain by deep argillic and calcic horizons.

2.2. Rainfall plots and simulation

Four 6 by 2 m long-term runoff plots were established on each experimental site. The plots had sheet metal borders installed at the top and sides to prevent lateral flow. Plot surface and vegetative cover were measured at 400 points on a 15 × 20 cm grid using the line-point intercept method (Herrick et al., 2005). Surface cover was

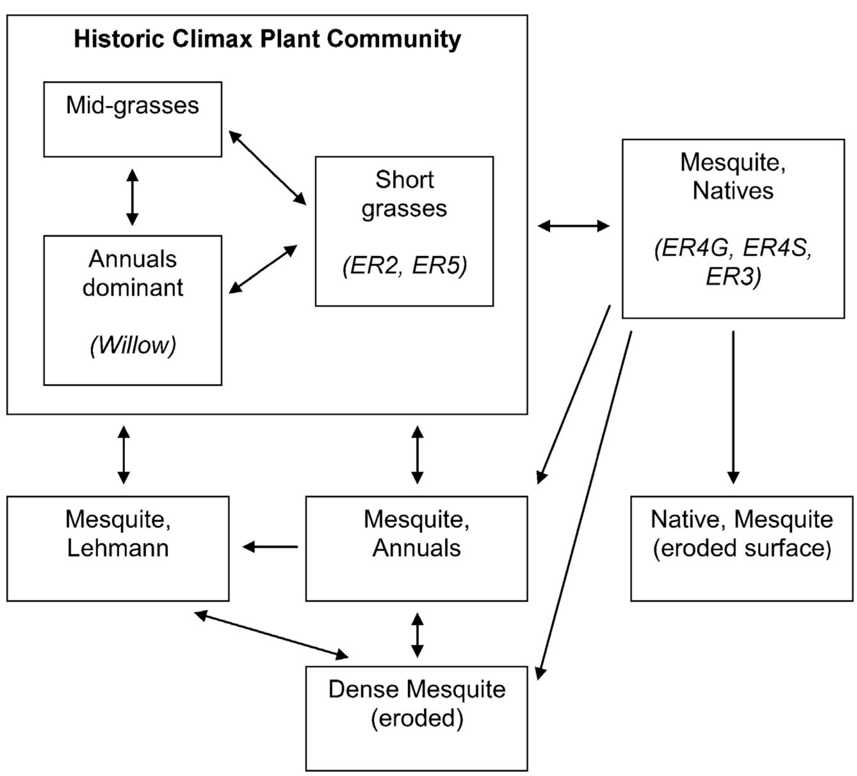

Fig. 1. Experimental sites within State and Transition Model for MLRA 41-3, loamy uplands (R041XC313AZ) (NRCS, 2013).

Table 1
Location and key characteristics of experimental sites.

Location	Latitude longitude	Elevation	Slope	STM state	Management notes	Rainfall simulation years	Clay/silt/sand
		m	%				%
ER2	31.7086 −110.588	1418	12.8	HCPC	Wildfire 2000 Drought, heavy grazing until mid 2000s, light grazing after	2003, 2007, 2010, 2013	11/15/14
ER5	31.75639 −110.6792	1480	6.3	HCPC	Exclosure since mid 1980's prior grazing intensity unknown	2010	n/d
Willow	31.45217 −110.634	1535	8.7	HCPC	Wildfire 2005	2006, 2007, 2010	17/28/55
ER3	31.76427 −110.559	1419	13.1	Mesquite-Natives	Wildfire 2005	2005, 2006, 2009, 2013	n/d
ER4S	31.79564 −110.619	1375	4.3	Mesquite-Natives		2006, 2007, 2010, 2013	11/20/69
ER4G	31.79571 −110.618	1371	4.7	Mesquite-NATIVES	Brush removal 2006 (post simulation that year)	2006, 2010, 2013	11/20/69

characterized as rock, litter, basal, bare soil, and canopy and vegetation classified by species. Soil aggregate stability samples were collected along three transects for a total of 18 samples per plot. Each sample was subjected to the slake test (Herrick et al., 2001; Holifield Collins et al., 2015) and assigned a 1 through 6 stability class (Table 2).

A portable, computer-controlled, variable intensity rainfall simulator, referred to as the Walnut Gulch Rainfall Simulator (WGRS) was used in the experiment. The WGRS has a single oscillating boom with four V-jet nozzles that can deliver rainfall rates ranging between 13 and 178 mm h^{-1} with kinetic energy of 204 kJ ha^{-1} mm^{-1} and coefficient of variability of 11% across a 2 by 6.1 m plot. Detailed description and design of the simulator are available in Stone and Paige (2003).

A sequence of rainfall simulations was conducted between 2003 and 2013. The experiment was repeated in different years on the same plots: four times on ER2, ER3, and ER4G, three times on Willow and ER4G, and once on ER5 (Table 1) in order to capture the evolution of ecological sites and burn recovery. The experimental procedure was as follows. A 45-min dry run at 65 mm h^{-1} intensity, followed by a 45 min pause and a wet run with varying intensity. The purpose of the dry run was to saturate the soil and create initial hydrological conditions for the wet run. The wet run consisted of a sequence of application rates (65, 100, 125, 150, and 180 mm h^{-1}) that were increased after runoff had reached steady state for at least 5 min and 3 runoff samples were collected. Runoff rate at the outlet was measured using a calibrated flume with an electronic gage. Overland flow velocities at each rainfall rate increment were obtained by measuring the travel time of a salt solution applied in the middle of the plot. For this purpose the flume was equipped with resistivity sensors.

Table 2
Range of values of soil cover and aggregate stability measured on experimental plots over a 10 year period.

Location		Surface cover, %					Aggregate
		Rock	Litter	Basal	Bare soil	Canopy	stability
ER2	Min	12	17	1	7	32	4.1
	Average	24	45	6	25	53	5
	Max	38	72	17	55	70	6.0
ER3	Min	12	5	0	3	4	2.8
	Average	28	39	5	28	38	4
	Max	50	84	17	60	63	5.3
ER4G	Min	9	15	3	7	21	3.3
	Average	13	53	8	30	37	5
	Max	23	80	23	67	54	5.8
ER4S	Min	0	15	0	3	18	2.2
	Average	11	51	4	35	38	5
	Max	34	87	13	71	51	6.0
ER5	Min	0	85	1	4	71	5.6
	Average	0	91	3	6	76	6
	Max	1	95	6	9	83	5.9
Willow	Min	0	27	0	1	0	4.0
	Average	3	77	5	16	37	5
	Max	12	99	15	52	76	6.0

2.3. Sample collection and ^{137}Cs analysis

Soil samples were collected in April 2013 using a soil auger with 34 cm^2 sampling area (6.6 cm diameter) to 15 cm depth. Prior reference sampling in the area showed this depth to be sufficient to capture the entire ^{137}Cs inventory on undisturbed or eroded sites (unpublished data). Sixteen samples were collected on each ES following a grid pattern in close proximity and between the runoff plots while avoiding any area that might have been disturbed by rainfall simulation or other equipment. Soil samples were dried and sieved through a 2 mm sieve to determine the rock fraction. For spectral analysis, soil and rock fractions were combined to ensure that cesium that adhered to rock fragments (Auerswald and Schimmack, 2000) was accounted for. The samples were then ground, weighed, and placed into 170 ml polypropylene jars with air-tight lids.

The analysis for ^{137}Cs was performed using the gamma ray spectrometry system consisting of three high-purity germanium detectors (Canberra GC4019) with > 30% relative efficiency and multi-channel analyzer (DSA-2000). The detectors were enclosed in 100 mm thick lead shield. The system was calibrated using mixed radionuclide reference material IAEA-327 (Dekner, 1996) certified by International Atomic Energy Agency. The gamma emission spectrum was obtained over 0–2 MeV range with the resolution of 0.24 keV (8192 channels). Measurement and spectrum analysis were conducted using Genie-2000 Spectroscopy software (Canberra, 2009). The samples were counted for at least 80,000 s or until < 10% peak area uncertainty was achieved. Activity of ^{137}Cs was calculated from the 661.6 keV photopeak. The analysis included correction for self-attenuation due to variation of sample density (Quindos et al., 2006).

The Profile Distribution Model (Walling and He, 1999) was utilized to convert measured ^{137}Cs inventories into soil erosion and deposition rates on the study sites. The model assumes that total ^{137}Cs fallout occurred in 1963 (fallout peak) and, in the absence of cultivation, the nuclide is concentrated near the surface while its depth distribution exhibits an exponential decline (Walling and Quine, 1990):

$$A_h = A_{ref}\left(1-e^{-x/h_0}\right) \quad (1)$$

where A_h is the amount of ^{137}Cs (Bq m^{-2}) above depth h (cm), A_{ref} is the ^{137}Cs total inventory of the profile (Bq m^{-2}), x is the mass depth (kg m^{-2}), and h_0 is the profile shape coefficient. The shape of the profile distribution (h_0) is determined experimentally at an undisturbed reference site.

Table 3
^{137}Cs activity and estimates soil erosion rates on the experimental sites.

Site	State	^{137}Cs inventory			CV	Estimated erosion
		Average[1] Bq m^{-2}	Min Bq m^{-2}	Max Bq m^{-2}	%	rate[2] t ha^{-1} y^{-1}
Willow	HCPC	631	324	1017	30	5.1[a]
ER5	HCPC	547	238	1056	46	6.7[ab]
ER2	HCPC	495	215	858	29	7.1[ab]
ER4S	Mesq./Natives	569	182	1280	65	7.4[ab]
ER3	Mesq./Natives	432	133	936	45	8.7[bc]
ER4G	Mesq./Natives	335	127	595	46	11.0[c]

[1] n = 16 for each site.
[2] Numbers with the same letter are not significantly different at $\alpha = 0.05$.

The net erosion and deposition are then determined by comparing measured inventory to the ^{137}Cs inventory on undisturbed local site. The erosion rate is than expressed as:

$$Y = -10/(t - 1963)\ln(1 - X/100)h_0 \quad (2)$$

where t is the year of sample collection, and X (%) is the reduction of total ^{137}Cs inventory.

Reference inventory was determined by sampling carefully selected undisturbed sites (27 cores partitioned at 2 cm increments) in the vicinity of the study area and, in addition, verified using a global ^{137}Cs distribution model (Walling and He, 2000). This model is designed to estimate likely total inventory for a specific location taking into account longitudinal and latitudinal variations in fallout input, secondary inputs (Chernobyl accident), precipitation patterns, and nuclide decay.

Stepwise selection method (SAS, 2008) was used to identify variables that best predicted runoff and erosion rates from the plots. Linear regression was employed to describe the relationships.

3. Results

3.1. The ^{137}Cs inventory and erosion estimates

The ^{137}Cs inventories and long term (50 years) erosion estimates for the six experimental sites are shown in Table 3. The average ^{137}Cs inventory varied between 335 Bq m^{-2} (ER4G) and 631 Bq m^{-2} (Willow). ER4S exhibited seven-fold difference between minimum and maximum inventory for an individual sample and the highest CV (65%) among all sites. ER2 had the lowest CV of 29%. ^{137}Cs is considered to be a moderately variable (CV 15-35%) soil property (Sutherland, 1996) where random spatial variability is the most influential factor affecting the overall uncertainty of its measurement (Owens and Walling, 1996). Random spatial variability is caused by variations in soil bulk density, rock fraction, microtopography, plant cover, and root distribution and it is also a function of sampling area. The values of CVs for a given sample size (n = 16) presented in Table 3 agree well with values commonly reported in the literature (Owens and Walling, 1996; Sutherland, 1996). High variability of ^{137}Cs inventory (CV <45%) is typically found at forest sites (Wallbrink et al., 1994), which may be attributed to canopy effects. In our study similarly high CV was observed on sites dominated by shrubs (ER4S). On HCPC sites with denser grass cover (Willow, ER2) the CV was lower (~30%).

The estimated reference inventory (A_{ref}) obtained from undisturbed reference sites was 1057 Bq m^{-2} at the Empire Ranch and 1162 Bq m^{-2} at San Rafael. The reference profile exhibited peak of ^{137}Cs concentration at 0–2 cm, which declining exponentially with depth (h_0 = 45). The entire ^{137}Cs inventory was contained in the top 12 cm soil layer, which is comparable with other locations in the region (Crouvi et al., 2015) but sallower than arid sites with similar soils elsewhere (Quine et al., 1994). The A_{ref} was in good agreement with the value (1250 Bq m^{-1}) calculated using global ^{137}Cs distribution model (Walling and He, 2000).

The estimated soil erosion rate varied between 5.1 t ha^{-1} y^{-1} and 11.0 t ha^{-1} y^{-1} on Willow and ER4G sites, respectively (Table 3). These rates, with the exception of that on ER4G, compare well with erosion measured locally on small watersheds (Nichols, 2006; Polyakov et al., 2010a) or calculated using the ^{137}Cs method on hillslopes (Nearing et al., 2005; Ritchie et al., 2005). Statistical analysis shows significant difference only between Willow (5.1 t ha^{-1} y^{-1}) and ER3, ER4G (8.7, 11.0 t ha^{-1} y^{-1}). The rock fraction on the experimental sites ranged between 5% (Willow) and 25% (ER2), and slope gradient between 4.3% (ER4S) and 13.1% (ER3); however, there was no significant relationship between these abiotic factors and erosion rate. The former finding is consistent with previous observations (Nearing et al., 2005) in similar ecological conditions.

3.2. Erosion plots and sediment yield

Stepwise selection of predictor variables (SAS, 2008) for regression of the form:

$$S_y = \beta_0 + \beta_1 \times P \quad (3)$$

showed that rainfall rate, P (mm h^{-1}), was the best predictor of steady state sediment yield, S_y, (g m^2 min^{-1}), explaining between 81% (ER3, 2013) and 99% (ER2, 2003) of its variability. The relationship was statistically significant in all cases (Table 4). Fig. 2 shows the relationship between P and measured S_y on the rainfall simulation plots, and its temporal (year to year) dynamic. Erosion response to rainfall varied greatly depending on the ecological site and year. The highest sediment yield (17.2 g m^2 min^{-1} at P = 173 mm h^{-1}) was observed on ER4G and the lowest (0.9 g m^{-2} min^{-1} at P = 175 mm h^{-1}) on Willow site (Fig. 3). The range of S_y values within one site under the same rainfall intensity also showed large variation from year to year

Table 4
Regression coefficients for steady state sediment yield equation ($S_y = \beta_0 + \beta_1 \times P$) from the rainfall simulation plots.

Sites	Parameter	Year of simulation						
		2003	2005	2006	2007	2009	2010	2013
Willow	β_0			−2.81	−0.24		0.35	
	β_1			0.045	0.006[a]		0.007[a]	
ER5	β_0						−0.58	
	β_1						0.012	
ER2	β_0	−1.61			−0.71		0.40	−1.19
	β_1	0.051			0.028[a]		0.014	0.029[a]
ER3	β_0		−4.15	−2.24		−0.12		−0.45
	β_1		0.105	0.078		0.013[a]		0.028[a]
ER4S	β_0			−0.74	−1.50		−1.16	−0.88
	β_1			0.031	0.042[a]		0.047[a]	0.016
ER4G	β_0			0.14			−5.81	−1.14
	β_1			0.019			0.127	0.030

All parameters are significant at $P = 0.05$. Values with the same letter within a row do not significantly differ (ANCOVA).

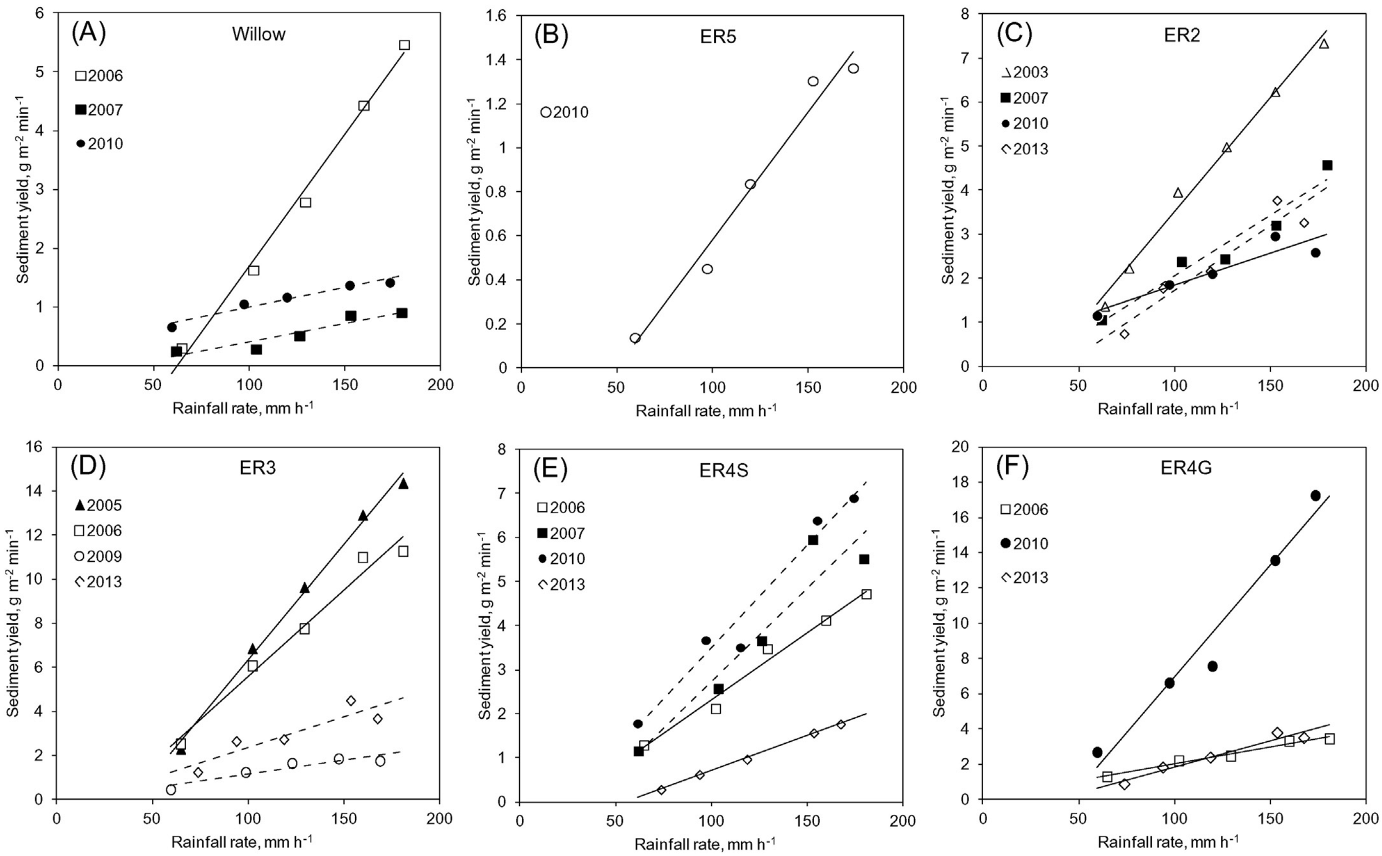

Fig. 2. Steady state sediment yield under simulated rainfall on experimental sites and its temporal (year to year) dynamic. Slopes of dashed regression lines within the same graph are not significantly different from each other $\alpha = 0.05$.

(Fig. 2). For example, on ER3 at $P = 175$ mm/h steady state S_y decreased 8-fold between 2005 and 2009.

In order to identify factors behind temporal changes in erosion response to precipitation within a site, stepwise selection of the second predictor variable was performed. Among the tested variables were biotic and abiotic factors: rock surface cover, litter cover, basal area, bare soil area, canopy cover, and aggregate stability (Table 2). ER5 was excluded from the analysis because rainfall simulation there was conducted only once in 2010. Bare soil area was selected as the second predictor for steady state S_y across all sites. Bare soil serves as an aggregate indicator for all types of soil cover combined. This demonstrates the governing role of surface cover in erosion processes on rangelands and agrees well with prior observations (Goff et al., 1993; Mergen et al., 2001). For example, in a study of semiarid grassland community (Polyakov et al., 2010b) a drought induced temporary decline in canopy cover caused a 23-fold sediment yield increase compared with the preceding period, despite similarities in precipitation characteristics.

4. Discussion

The HCPC sites (ER2, ER5, and Willow) overall had lower ^{137}Cs estimated erosion rates than Mesquite/Natives sites (ER3, ER4G, ER4S). The primary erosion mechanisms on well-vegetated HCPC sites are splash and sheet flow, hence the erosion rates are relatively low. Mesquite/Natives sites with shrub encroachment and greater bare soil areas (Table 2) undergo a shift towards concentrated flow as the dominant runoff and erosion mechanism resulting in increased erosion. There is a structural threshold between homogeneously vegetated HCPC sites and fragmented Mesquite/Natives sites characterized by larger vegetative patches surrounded by interconnected bare soil susceptible to channelization.

Large contrast was observed between estimated erosion rates on two adjacent sites ER4G and ER4S characterized by almost identical abiotic factors but different vegetation (grass versus shrub respectively). Grass rangelands subject to shrub encroachment usually demonstrate an increase in soil erosion due to reduced density of understory species and surface litter (Pierson et al., 2007). Similarly Quine et al. (1994) using ^{137}Cs method on uncultivated arid site in Spain found greater erosion rates under trees than on brush-grass covered areas. The opposite was observed in this study. A 34% difference in erosion on grass versus shrub site (11.0 and 7.4 t ha^{-1} y^{-1}, respectively) might be attributed to mesquite removal operation that involved severe soil surface disturbance. It has been shown that in arid rangeland conditions a lack of canopy cover can result in 20 fold increase in annual sediment yield compared to the preceding period under native grasses (Polyakov et al., 2010b). This suggests that major soil losses occur during transition from one state within STM to another, when the soil surface remains unprotected for a relatively short time, rather than during more prolonged stable period under one state or another. An example of this effect was documented in detail by Polyakov et al. (2010b) in the case of transition in southeastern Arizona of a native bunch grass ecosystem to one invaded over a short period of time to a site dominated by invasive Lehmann lovegrass (*Eragrostis lehmanniana*).

Erosion rates on rainfall plots located within the Mesquite/Native state were overall greater than erosion rates on plots within HCPC state. The ranges of regression slopes β_1 for these two states were 0.012 to 0.127 and 0.006 to 0.051 respectively (Eq. (3), Table 4). This trend was very similar to the results from ^{137}Cs analysis (Table 3), namely HCPC state was characterized by lower erosion rates and lesser variation of those rates from year to year. In addition the ranking of sites by erosion rate from least to greatest (Willow, ER5, ER2, ER4S, ER3, and ER4G) as determined by ^{137}Cs method (Table 3) was almost the same as the ranking (ER5, Willow, ER2, ER4S, ER3, and ER4G) based on rainfall

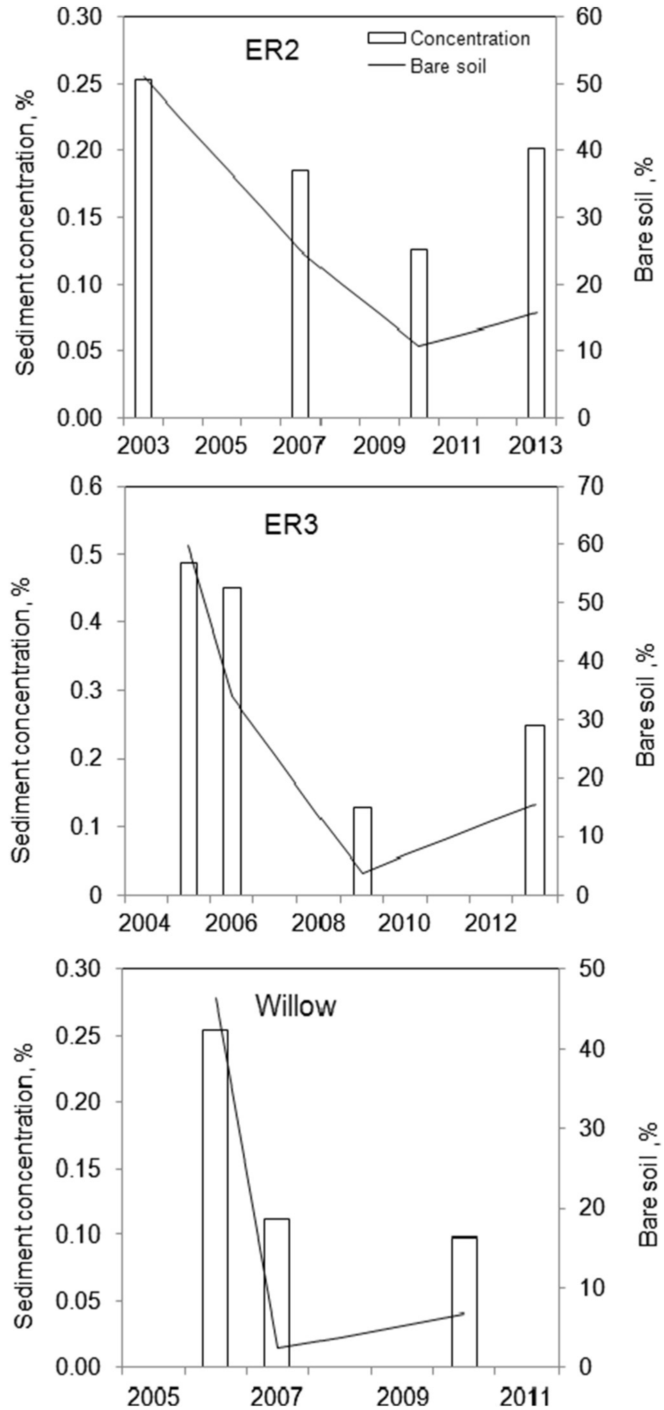

Fig. 3. Ground cover and sediment concentration in runoff and its temporal change on three sites after wildfire. Wildfire occurred in 2005 on ER3 and Willow, and in 2000 on ER2.

simulation (Fig. 2, Table 4). On some ecological sites particularly large temporal differences in steady state S_y, indicated by variation of the slope parameter β_1, were associated with known natural disturbances or management practices.

Wildfires occurred on ER2 in 2000 and on Willow and ER3 in 2005. All these sites displayed increased sediment concentration in runoff shortly after the event (0.25%, 0.49%, and 0.25% respectively) (Fig. 3). This increase in erosion was caused by decline in litter, basal, and canopy cover due to fire, and associated increase in bare soil to over 50% of the area. Plot photographs obtained during simulations were consistent with soil cover measurements. In the following years as vegetation recovered from fire, a new layer of organic litter accumulated and the percentage of bare soil decreased to 4% on ER3 and Willow, and 11% on ER2. During the vegetation and surface condition recovery, sediment concentration in runoff decreased 2–3 fold (Fig. 3). Post-fire soil microtopography and associated ground cover have been previously identified as important determinants of the potential for increased inter-rill erosion in arid environments (Soto and Diaz-Fierros, 1998; Pierson et al., 2002; Badia and Marti, 2008).

Ecological sites ER4S and ER4G share similar abiotic characteristics and are located in close proximity of each other. Rainfall simulation conducted in 2006 preceded mesquite removal on ER4G by two months. By 2010, the mesquite had re-sprouted throughout the ER4G site; however mesquite plants were not found inside of the rainfall plots. At the same time canopy cover on ER4G decreased from 38% in 2006 to 27% in 2010 and then increased to 45% in 2013. This was accompanied by decline of forbs, which were completely replaced by grasses by 2013. ANCOVA showed that there was significant difference between the regression exponent β_1 for 2006, 2010, and 2013 (Table 4) that span the vegetation transition on the plots. The resulting anomalously large sediment yield in 2010 might have been related to mechanical disturbance associated with mesquite removal. However, cover data for the period between 2006 and 2010 is lacking, making it difficult to interpret its cause.

The ER5 site was excluded from grazing since the mid-1980s, but was grazed prior to that time. Its low erosion rate under the rainfall simulator was consistent with its highest litter (91%) and canopy cover (76%) among all locations. The ER2 site had wildfire in 2000 and was heavily grazed until the mid-2000s. These factors might explain the relatively low litter cover (19%) in 2003, which steadily increased to 67% in 2010. The temporal vegetation changes are consistent with the sediment yield trends (Fig. 3). The β_1 of 2003 regression was significantly greater than β_1 of any other year, while β_1 of 2007 and 2013 regressions was similar.

Climatic forecasts suggest an increase in frequency and severity of droughts as well as intensity of extreme precipitation events (Easterling et al., 2000a, 2000b; Christensen and Hewitson, 2007; Seager et al., 2007). These conditions are likely to alter runoff and soil recharge (Seyfried et al., 2005), initiate changes in plant community composition towards shrub encroachment into areas historically dominated by grasses (Huxman et al., 2005; Browning et al., 2008), and transitions across structural thresholds into different states of the STM (Briske et al., 2005). Considering this trend, erosion rates on arid rangelands are likely to increase in the future.

5. Conclusion

Long term (50 years) soil erosion rates on loamy upland rangeland estimated using the ^{137}Cs method showed that the HCPC state of the STM had lower erosion rates than Mesquite/Natives state. There was no significant relationship between abiotic factors such as rock fraction or slope gradient and erosion rate. The level of ^{137}Cs concentration CV allows differentiating erosion rates between two locations if the difference exceeds ~3 t h^{-1} y^{-1}. Greater number of samples or larger size of individual sample may improve the accuracy of the method.

Site classification within the STM was based on their current (recent ~10 years) ecological characteristics. Prior conditions of the ecological sites are not well known. Erosion estimate using ^{137}Cs method integrates the effects of management practices, vegetative cover or disturbances of a site over the past 50 years. Hence, proper interpretation of these erosion results requires knowledge of past states as well as the magnitude and frequency of transitions that have occurred on the location during this time.

Erosion response to rainfall on runoff plots varied greatly depending on the ecological site and the time of rainfall simulation. Temporal variability of steady state sediment yield, S_y, within a site was attributed to natural or management driven changes of plant community and surface conditions. Namely, sediment yield was the highest following wildfire, which caused rapid decrease of surface cover. In the following years, as vegetation recovered and surface litter accumulated, sediment concentration in runoff decreased 2–3 fold. Sediment yield under artificial rainfall is an indicator of the erosion potential at the time of simulation. Whether this potential results in actual soil loss from the site depends on the timing and magnitude of natural rainfall events in relation to changing surface conditions. The results of this study suggest that within state variation of erosion potential can be much greater than long term differences between states or ecological sites. Rainfall simulation results must be placed in the context of the range of possible vegetation and soil surface conditions within a given ecological site for meaningful interpretation.

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