Runoff and erosional responses to a drought-induced shift in a desert grassland community composition

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[1]   This study investigates how drought-induced change in semiarid grassland community affected runoff and sediment yield in a small watershed in southeast Arizona, USA. Three distinct periods in ecosystem composition and associated runoff and sediment yield were identified according to dominant species: native bunchgrass (1974–2005), forbs (2006), and the invasive grass, *Eragrostis lehmanniana* (2007–2009). Precipitation, runoff, and sediment yield for each period were analyzed and compared at watershed and plot scales. Average watershed annual sediment yield was 0.16 t ha^{-1} yr^{-1}. Despite similarities in precipitation characteristics, decline in plant canopy cover during the transition period of 2006 caused watershed sediment yield to increase 23-fold to 1.64 t ha^{-1} yr^{-1} comparing with preceding period under native bunchgrasses (0.06 t ha^{-1} yr^{-1}) or succeeding period under *E. lehmanniana* (0.06 t ha^{-1} yr^{-1}). In contrast, measurements on small runoff plots on the hillslopes of the same watershed showed a significant increase in sediment discharge that continued after *E. lehmanniana* replaced native grasses. Together, these findings suggest alteration in plant community increased sediment yield but that hydrological responses to this event differ at watershed and plot scales, highlighting the geomorphological controls at the watershed scale that determine sediment transport efficiency and storage. Resolving these scalar issues will help identify critical landform features needed to preserve watershed integrity under changing climate conditions.

1.   Introduction

[2]   Invasive grasses may affect ecosystems at various levels: resource uptake (light, water, and nutrient), resource supply (alteration of soil processes), geomorphology, microclimate, and others [*D'Antonio and Vitousek*, 1992]. In the United States, alien grass invasion is most severe in arid and semiarid rangelands of the west, where shifts in plant community composition have resulted in dramatic changes in coupled ecological and hydrological processes. Woody plant encroachment into areas historically dominated by perennial grasses can alter runoff and deep soil recharge [*Seyfried et al.*, 2005; *Wilcox et al.*, 2007], in addition to altering erosion rates [*Nearing et al.*, 2005; *Nichols*, 2006; *Pierson et al.*, 1994; *Pierson et al.*, 2007; *Ritchie et al.*, 2005]. In contrast, the hydrological consequences of the replacement of native perennial herbaceous plants by morphologically similar invasive species are less well studied. As with woody plant expansion, invasive herbaceous plants create alternate ecological steady states, which require new management practices and extensive efforts to restore plant community and ecosystem function to native conditions [*DiTomaso*, 2000; *Dukes and Mooney*, 2004].

[3]   Ecological studies have shown that invasive grasses can better extract soil water, more effectively drying soils and outcompeting native grasses and shrubs [*Eissenstat and Caldwell*, 1988; *Kolb et al.*, 2002]. Fires induced by the invasive annual grass, cheatgrass (*Bromus tectorum*), elicit changes in soil infiltration properties, resulting in drier soils and increased sheet flow and sediment yield [*Blank et al.*, 1994; *Pierson et al.*, 2001; *Pierson et al.*, 2002]. In California coastal sage scrub, a diverse complex of invasive cool season annuals reduces percolation into vadoze-zone depriving shrubs of deeper regolith water [*Wood et al.*, 2006]. Increased dominance by star thistle (*Centaurea solstitialis*), which extracts soil water across a more extensive soil profile than native perennial or annual grasses, has been shown to promote soil drying and decreased permeability that limits deep soil recharge [*Enloe et al.*, 2004]. Despite the strong coupling between surface hydrological and sediment dynamics, few studies have directly assessed the impact of invasive species on these important rangeland processes. Most of these studies were conducted in cold

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deserts. *Lacey et al.* [1989] showed that hillslope plots dominated by the invasive forb, spotted knapweed (*Centaurea maculosa*), increased runoff by 56% and sediment yield by 192% compared to native bunchgrass plots. This disparity in magnitude between runoff and sediment yield suggests dominance by exotic herbaceous plants could extensively and negatively impact rangeland hydrological function.

[4] Since its introduction for rangeland reclamation in the 1930s, the South African C_4 bunchgrass, Lehmann lovegrass (*Eragrostis lehmanninana* Nees.), has spread across the semiarid grasslands of southwestern United States dramatically reducing floral diversity [*Anable et al.*, 1992; *Cox et al.*, 1990; *Geiger and McPherson*, 2005; *McClaran and Anable*, 1992]. Lehmann lovegrass seems less sensitive to grazing and is capable of higher annual productivity than native grasses [*Anable et al.*, 1992; *Cox et al.*, 1990], which may accelerate fire frequency [*Anable et al.*, 1992; *McGlone and Huenneke*, 2004]. Lehmann lovegrass maintains green leaves year-round, facilitating rapid recovery from drought, allowing greater exploitation of winter and early monsoon season rain [*Cox et al.*, 1990; *Frasier and Cox*, 1994; *Geiger and McPherson*, 2005; *Hamerlynck et al.*, 2010] and increased overall ET [*Frasier and Cox*, 1994]. Lehmann lovegrass also tends to have sparser canopies and lower litter accumulation, which sometimes result in drier soils, possibly from increased soil evaporation [*English et al.*, 2005; *Hamerlynck et al.*, 2010; *Moran et al.*, 2009; *Yepez et al.*, 2005].

[5] Lehmann lovegrass was first documented at U.S. Department of Agriculture-Agricultural Research Service (USDA-ARS) Walnut Gulch Experimental Watershed (WGEW), near Tombstone, AZ, in 1994, when it was recorded at 7 out of 53 permanent transects, and again at 13 transects in 1999 [*King et al.*, 2008]. In 2005 Lehmann lovegrass cover was less than 1% at all but one location, where it accounted for 8% of cover and 24% of total vegetation [*King et al.*, 2008]. In 2006, following several years of extremely low rainfall, widespread mortality of native bunchgrasses and shrubs, and proliferation of broadleaf forbs, the species Lehmann lovegrass was noticed across the Kendall grassland site at WGEW, with a subsequent drastic increase in its abundance [*Moran et al.*, 2009]. Detailed observations of vegetation dynamics, such as these, combined with long-term precipitation, runoff, and sediment records are extremely rare and present a unique opportunity. This study seeks to establish and quantify the impact of the altered desert grassland community composition and structure on plot and watershed-level sediment response.

2. Methods

2.1. Experimental Site

[6] The Kendall grassland site is located within the 150 km² USDA-ARS WGEW, which is part of the upper San Pedro River basin in southeastern Arizona, USA. The climate of the area is semiarid with annual precipitation of 345 mm and a highly spatially and temporally varying precipitation pattern dominated by the North American Monsoon. Monsoon storms are typically characterized as short-duration, high-intensity, localized rainfall events. Mean annual temperature is 17.7°C.

[7] The study was conducted on Watershed 63.112 (109°56′28″W; 31°44′10″N) located in the Kendall grass-

land site in WGEW, 12 km NE from Tombstone, AZ, at an elevation of 1526 m. The site is predominantly covered by grass and forbs with some shrubs and succulents with a combined canopy cover of approximately 35%. Ground cover during the rainy season has been measured at 28% rock, 42% litter, and 14% plant basal cover [*Nearing et al.*, 2007]. Historically, the dominant desert grassland bunchgrasses at the site have been black grama (*Bouteloua eriopoda*), side-oats grama (*B. curtipendula*), three-awn (*Aristida* sp.), and cane beardgrass (*Bothriochloa barbinodis*) [*King et al.*, 2008], and more recently, Lehmann lovegrass (*Eragrostis lehmanniana*) [*Moran et al.*, 2009].

[8] Soils at the Kendall site are a complex of Stronghold (coarse-loamy, mixed, thermic Ustollic Calciorthids), Elgin (fine, mixed, thermic, Ustollic Paleargids), and McAllister (fine-loamy, mixed, thermic, Ustollic Haplargids), with Stronghold the dominant soil [*USDA*, 2003]. Stronghold soils have a surface A horizon (0–3 cm) containing 67% sand, 16% silt, and 17% clay, with 79% coarse fragments (>2 mm). The organic carbon content of the soil surface (0–2.5 cm) is 1.1%. Watershed 63.112 has an area of 1.8 ha and average gradient of 12.5% and is drained by concentrated flow paths that terminate in a swale above the outlet. The swale historically is a site of water and sediment storage within the watershed [*Nearing et al.*, 2005].

2.2. Instrumentation and Sampling

[9] Precipitation was measured using a weighing-type rain gauge with 0.25 mm, 1 min resolution located on the watershed. In cases when precipitation records were missing the data from a nearest gage at WGEW was used. Runoff rates from the watershed were measured using a 1:3 V-notch weir with stage recorder. Sediment was collected using a depth-integrating pump sampler with floating intake that rises in response to the flow depth [*Simanton et al.*, 1993]. The size of the sampled sediment was limited by the 6.4 mm diameter perforations on the sampler arm. Samples were taken during flow events at increasing intervals (3–10 min) depending on the flow duration. Runoff samples were used to determine sediment concentration and total sediment yield.

[10] Vegetation cover has been measured at the Kendall grassland along paired 33 m line transects established in 1967 [*King et al.*, 2008]. Line intercepts of canopy cover were recorded and identified to species. Plant basal area was identified as *E. lehmanniana*, forbs, native bunchgrass, shrub, dead native grass, and dead shrub. Vegetation surveys were conducted from August to October. In addition, plant density (number m^{-2}) and individual plant basal area (cm^2) was measured from October to November 2007 for 848 and 252 Lehmann lovegrass and native bunch grass individuals, respectively, in 50 1 square meter plots located across the 63.112 watershed.

[11] A plot study under simulated rainfall was conducted using the Walnut Gulch Rainfall Simulator (WGRS) [*Paige et al.*, 2004]. The WGRS is an oscillating boom simulator equipped with four VeeJet 80100 nozzles and can produce rainfall intensities ranging from 12 to 178 mm h^{-1}. Four 2 × 6 m plots were subjected to a sequence of three rainfall simulations (dry, wet, and very wet). The "dry" simulation was applied under existing soil moisture conditions at constant rate of 60 mm h^{-1} for 45 min. After cessation of runoff from the dry run, a wet run was applied at increasing rates

from 25 to 178 mm h^{-1}. During the wet run, rainfall rates were changed after runoff reached steady state rate for at least 5 min. Runoff rate was measured with an electronic staff gage and a precalibrated flume. Sediment concentrations were measured with timed grab samples, which were dried and weighed to compute sediment concentrations.

[12] Canopy and ground cover on the rainfall simulation plots were measured at points on a 15 × 20 cm grid (400 points per plot). Canopy cover was recorded as shrub, grass, and forbs, identified as annual or perennial and species, if possible. Ground cover was recorded as rock (>2 mm), gravel, litter (herbaceous, woody, and embedded), vegetative base, and bare soil and was measured both inside and outside canopy cover.

2.3. Data and Analysis

[13] Watershed data gathered for the study included hyetographs of rainfall events and hydrographs (1963–2009), sediment concentration in watershed runoff (1974–1977, 1995–2009), and vegetation and ground cover from permanent transects (1994, 1999, annually 2005–2009).

[14] Runoff plot data included four replications of steady state runoff and sediment discharge rates at five levels of rainfall intensity (62, 104, 126, 159, and 179 mm h^{-1}) collected for 2 years under native bunchgrass dominance (2002 and 2004) and for 2 years under Lehmann lovegrass dominance (2007 and 2008). Canopy and ground cover was measured at the time of simulation.

[15] Runoff samples were weighed, air dried, and weighed again to determine sediment concentration. Watershed sediment yield was calculated by integrating the product of sediment concentration and flow rate. Total sediment yield was calculated for the events where three or more sediment samples were obtained. Runoff events with fewer than three sediment samples were considered to be inadequately sampled [*Nearing et al.*, 2007]. Runoff events with missing rainfall data were assigned precipitation from the nearest rain gauge.

[16] Standardized Precipitation Index (SPI), a measure of precipitation deviation from normal [*McKee et al.*, 1993; *McKee et al.*, 1995], was calculated using SPI SL 6.exe software (http://www.drought.unl.edu/monitor/spi). SPI of 0 indicates normal precipitation, −1 to −1.99 severely dry and <−2.0 extreme dry conditions. Hydrological year SPI (i.e., 12-month October SPI) were calculated from continuous monthly observations (1958–2007) from the WGEW Field Office rain gauge of 63.81. SPI provides a rigorous cross-site precipitation context that allows our results to be compared to observations in locations with distinctly different annual rainfall regimes [*McKee et al.*, 1995].

[17] Variables used in the analysis were precipitation amount P_t (mm), intensity P_i (mm h^{-1}), peak intensity P_p (mm h^{-1}), maximum precipitation in a 30 min interval I30 (mm), energy of precipitation event E (MJ ha^{-1}), runoff amount Q_t (mm), runoff peak Q_p (mm h^{-1}), runoff rate q_r (mm h^{-1}), sediment discharge rate q_s (g s^{-1}) sediment yield S_y (t ha^{-1}), slope S_0 (%), plant basal area A_b (cm^2), and canopy cover C_c by species or type (%).

[18] The data were examined for normality of variables and residuals distribution (Shapiro-Wilks normality test) and linearity of model relationship. Linear regression models were used to describe relationship between Q_t, S_y, q_s, and independent variables. When relating sediment yield to runoff, linear regression was calculated on logarithmically transformed data. The method involves transformation of power relationship $Y = \beta_0 X \beta^1$ into a linear function $\log(Y) = \beta'_0 + \beta_1 \log(X)$, then running a simple linear regression. The resulting coefficient β_1 becomes the optimal exponent, and β_0, back transformed from β'_0, becomes the multiplier for the power function. The stepwise method [*SAS*, 2008], which combined forward-selection and backward-elimination steps, was used to select the most significant predictors from a pool of available variables. Analysis of covariance (ANCOVA) and a *t* test was used to test the hypotheses that slope and intercept of the regression equations differed.

[19] The analysis was performed using SAS 9.1 statistical package [*SAS*, 2008]. In all statistical tests, $P = 0.05$ was used, unless otherwise indicated.

3. Results

3.1. Plant Cover

[20] Before 2006, vegetative cover on the watershed was dominated by native grasses such as black grama (*Bouteloua eriopoda*), sideoats grama (*Bouteloua curtipendula*), curly mesquite (*Hilaria bilangeri*), and three-awn sp. (*Aristida* spp.), with shrub species such as fairy duster (*Calliandra eriophylla*) and longleaf jointfir (*Ephedra trifurca*), and herbaceous forbs such as desert zinnia (*Zinnia pumila*), hairyseed bahia (*Bahia absinthifolia*), and leatherweed (*Croton corymbulosus*). A sharp decline in vegetative cover, particularly in grama grass species and shrubs, occurred in 2006, when forb relative cover increased (Figure 1). In the same year, *E. lehmanniana* was first documented on permanent transects at the Kendall site, accounting for 1.3% of total C_c and 7.7% of relative C_c. These values increased to 31% and 45.2%, respectively, by 2008. Although 2007 and 2008 showed some recovery of native bunchgrasses, *E. lehmanniana* became a dominant species over this 2 year period (Figure 1).

[21] In 2007 the basal area of individual *E. lehmanniana* plants was relatively small (3 cm^2 per plant) compared to native bunchgrasses (11.7 cm^2 per plant); however, *E. lehmanniana* was more numerous (12 plants per square meter) than native bunchgrasses (1 plant per square meter). At the same time, *E. lehmanniana* accounted for 38.2% of total C_c, compared to 26% for native bunchgrasses.

[22] On the runoff plots, vegetation dynamics were similar to those from the permanent transects; shrub C_c contributions declined dramatically from 21% in 2004 to 0.5% in 2007 and further to 0.1% in 2008. Perennial grass C_c increased slightly over the same time from 29% to 31% and to 34%. However, the biggest change occurred within grass canopy composition. *E. lehmanniana* was present on the plots as early as 2002 (5%), but its presence became substantial only in 2007 and 2008 when its contribution to the total C_c increased to 52% and 84%, respectively, replacing native bunchgrasses almost entirely.

[23] Considering the changes in vegetation described above, three distinctive stages could be identified: a period dominated by *B. eriopoda* (1974–2005), a transition period dominated by forbs (2006), and a posttransition period dominated by *E. lehmanniana* (2007–present) [*Moran et al.*, 2009].
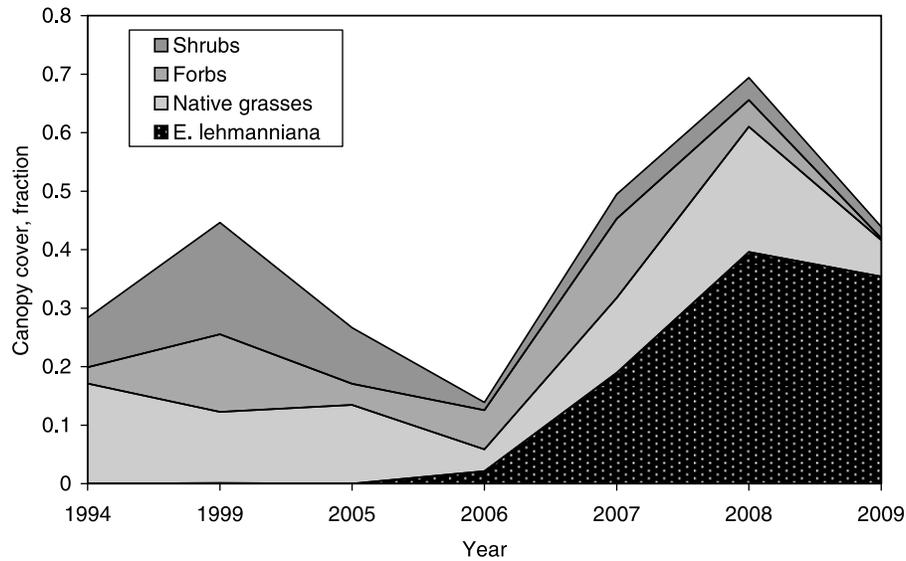
Figure 1. Fraction of total canopy cover of different plant types on permanent transects at Kendall grassland site between 1994 and 2009.

3.2. Precipitation and Runoff

[24] Annual P_t at the watershed varied between 153 and 570 mm during the record period and was 345 mm on average. The monthly rainfall varied significantly, with 60% of the total rainfall occurring in July through September. Events with P_p less than 25 mm h^{-1} accounted for 80% of total P_t and 97% of the total number of events. A period between 2002 and 2008 was characterized by prolonged drought. SPI indicated dry or severely dry (2003, 2005, 2006) conditions (Figure 2) for 8 consecutive years.

[25] Four periods of precipitation data that correspond to available watershed sediment yield data and *E. lehmanniana* invasion timeline were compared: 1974–1977, 1995–2005,

2006, and 2007–2009. Precipitation frequency distributions of four periods were not significantly different from each other (Kolmogorov-Smirnov nonparametric test). However, there was a slight increase in the occurrence of large events over time. Rainfalls greater than 30 mm contributed 15% of the total P_t in 1974–1993, whereas in 1994–2009, the contribution of same size rainfalls increased to 18% of the total.

[26] The average event P_t among four periods ranged between 3.8 mm (2007–2009) and 5.2 mm (1974–1977), and these values were not significantly different from each other. Similarly, the average maximum precipitation in a 30 min interval, I30 (mm), during 2007–2009 (5.3 mm) was
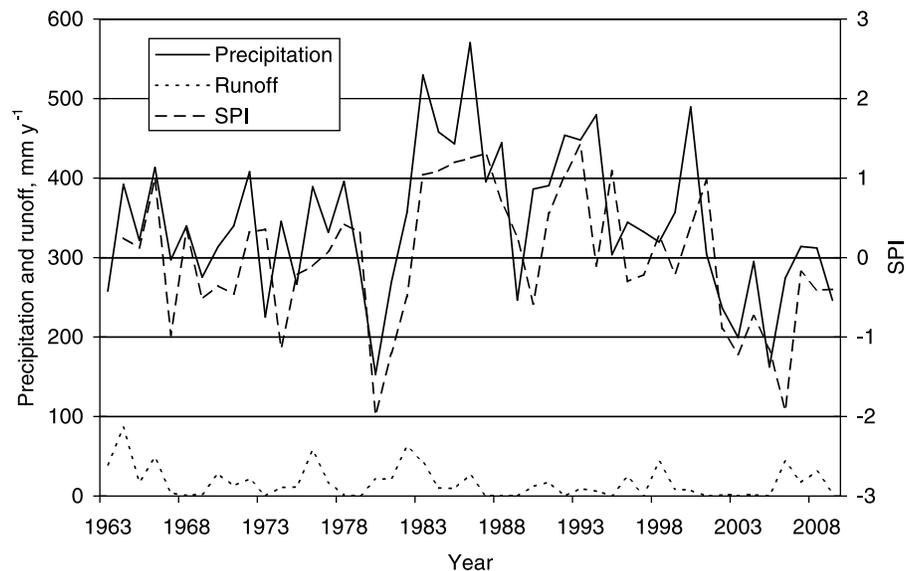

Figure 2. Watershed 112 annual precipitation and runoff with hydrological year standardized precipitation index (October 12-month SPI).

Table 1. Precipitation, Runoff, and Sediment Yield on Watershed 112 at WGEW[a]

Period	Precipitation			Runoff			Sediment			Concentration $g \, L^{-1}$
	n	mm	mm yr^{-1}	n	mm	mm yr^{-1}	n	t ha^{-1}	t ha^{-1} yr^{-1}	
1974–1977	258	1333	333	32	97.5	32.5	19	0.37	0.10 a	0.38
1995–2005	790	3344	304	27	88.6	8.1	19	0.69	0.06 a	0.78
2006	64	274	274	15	43.4	43.4	12	1.62	1.62 b	3.74
2007–2009	228	872	291	12	52.7	17.6	8	0.17	0.06 a	0.32
Total	1340	5823		86	282.2		58	2.85		

[a]Values with the same letter do not significantly differ (Duncan multiple range test).

the smallest among all periods but not significantly different from the I30 in 2006 (6.5 mm) and 1995–2005 (8.7 mm).

[27] Number of rainfall events that produced runoff varied overtime with noticeable increase during the transition period: 3.4 % in 1995–2005, 23.4% in 2006, and 5.2% in 2007–2009. Average annual Q_t followed similar pattern of change with values of 8.1, 43.4, 17.6 mm yr^{-1} during periods dominated by native bunchgrass, forbs, and *E. lehmanniana*, respectively. Long-term average annual Q_t was 16.9 mm yr^{-1} or 4.8% of P_t (Table 1). The best predictor for runoff volume Q_t was the I30, which explained between 18% and 84% of Q_t variability (Table 2). During the transition and post-transition periods (2006–2009) runoff from the watershed increased, as indicated by statistically significant threefold increase in regression slopes comparing to the previous periods (Table 2). The values of I30 that caused runoff initiation during the periods of native grass domination, transition, and *E. lehmanniana* domination were 19.5, 8.8, and 20.4 mm, respectively.

3.3. Sediment

3.3.1. Watershed

[28] During 19 years of observation, there were 58 successfully sampled sediment events (67% of all recorded runoff events). The total S_y for the period of record was 2.86 t ha^{-1}, 57% of which occurred in 2006. Annual S_y for different periods varied from 1.64 t ha^{-1} yr^{-1} (2006) to 0.01 t ha^{-1} yr^{-1} (1974–1977) (Table 1). There was a significant difference between average annual S_y during the transition and other periods. Event S_y also varied greatly: the three largest sediment events all occurred in 2006 and accounted for 42% (1.20 t ha^{-1}) of the total S_y during the period of record.

[29] Stepwise regression analysis showed that the Q_t was the best overall predictor of S_y. To determine whether there was a difference between vegetation periods in runoff-adjusted sediment yield, a log-transformed power function was used on data from each period separately. Regression results are presented in Table 2.

[30] In 2006 sediment yield increased sharply compared with the previous period (1995–2005) as indicated by greater exponent parameter ($\beta_1 = 1.547$ and 1.143, respectively) (Table 2 and Figure 3). In the period immediately following vegetation transition (2007–2009), sediment yield to runoff relationship returned to its pretransition state. This means, for example, that a 10 mm runoff event in 2006 produced sediment yield that was 7 and 10 times greater than in 1995–2005 and 2007–2009, respectively. ANCOVA showed that there was no significant difference between β_1 of the regression models of 1995–2005 and 2007–2009, whereas the increase of the same parameter in 2006 was

statistically significant comparing to adjacent periods. The 2006 regression equation also had a higher R^2 (0.95) than other periods.

[31] The β_1 of 1974–1977 regression was not significantly different from β_1 of any other period. The equation parameters were affected by an observation with anomalously large sediment yield, which might have been related to decreased vegetative cover early in the monsoon season (3 July 1977). Canopy cover data for 1974–1977 is also lacking, making it difficult to interpret and compare runoff and sediment yield relationship for this period.

3.3.2. Runoff Plots

[32] Differences in erosion rates among years on runoff plots were tested by developing regression equations with the steady state q_s (g s^{-1}) as the dependent variable and q_r (mm h^{-1}), S_0 (%), and P_i (mm h^{-1}) as the independent variables. Slope variable was initially included because slope varied slightly between plots (approximately 1% difference between replicated plots) and the 2002 plots were on a slope approximately 3% steeper than the 2005–2008 plots, which were on an 11% slope. Stepwise regression analysis showed that q_r was the best predictor for steady state q_s, explaining between 38% (2008) and 74% (2005) of its variability. It was also the only variable that met the 0.05 significance level criteria for entry into the final model.

[33] Similar to the watershed analysis, a regression on log-transformed power function was utilized. ANCOVA showed that there was no significant difference between the exponent (β_1) for 2005, 2007, and 2008 (Table 3) that span the vegetation transition on the plots. However, a significant difference was found between the multipliers (β'_0) of the regression before (2005) and after (2007–2008) the spread

Table 2. Regression Equation Coefficients for Runoff ($Q = \beta_0 + \beta_1 I30$) and Sediment Yield ($S_y = \beta'_0 Q_t^{\beta_1}$) From the Watershed[a]

Period	β_0	β_1	n	R^2
	$Q_1 = \beta_0 + \beta_1 I30$			
1974–1977	−0.3059* ab	0.0918 a	32	0.18
1995–2005	−1.2724* c	0.0652 a	27	0.75
2006	−2.2340 a	0.2525 b	15	0.75
2007–2009	−5.8372 b	0.2857 b	12	0.84
	$S_y = \beta'_0 Q_t^{\beta_1}$			
1974–1977	0.0021 a	1.267 ab	19	0.64
1995–2005	0.0049 b	1.143 a	18	0.83
2006	0.0134 c	1.547 b	12	0.95
2007–2009	0.0061 bd	0.876 a	8	0.89

[a]All parameters are significant at $P = 0.05$, except where indicated: Asterisk indicates not significant. Values with the same letter do not significantly differ (ANCOVA).
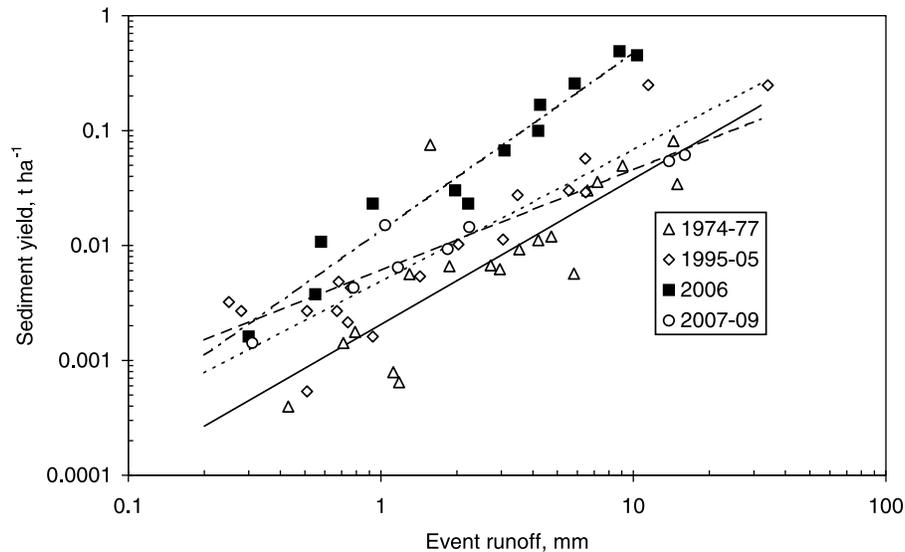
Figure 3. Event sediment yield as a function of runoff volume during different vegetation periods: native bunchgrasses domination (1974–1977, 1995–2005), forb-dominated transition (2006), and under *E. lehmanniana* dominance (2007–2009).

of *E. lehmanniana*. In 2007–2008 sediment discharge rate q_s increased comparing to 2002–2005 throughout the range of tested rainfall intensities q_r (Figure 4). This difference in q_s was more pronounced at lower q_r (80% increase at 60 mm h^{-1}) and less pronounced at greater q_r (10% increase at 160 mm h^{-1}).

4. Discussion and Conclusions

[34] One of the strongest droughts on record triggered a rapid transition in plant community on the Kendall grassland site (Figure 2). Hydrologic behavior of the watershed during the period of observation was defined by two interrelated factors: changes in plant total cover and shift in species composition. The results of the study indicate that change in plant community structure (spread of *E. lehmanniana*) increased sediment yield from the slopes (Figure 4). However, the geomorphological features of the site resulted in a dramatic recovery to predrought sediment yields at the watershed scale (Figure 3).

[35] Watershed hydrological response to precipitation during the transition period in 2006 was very different from the hydrological responses during any other period. Year 2006 was similar to the median year (57th percentile) in annual precipitation but was among years with some of the largest
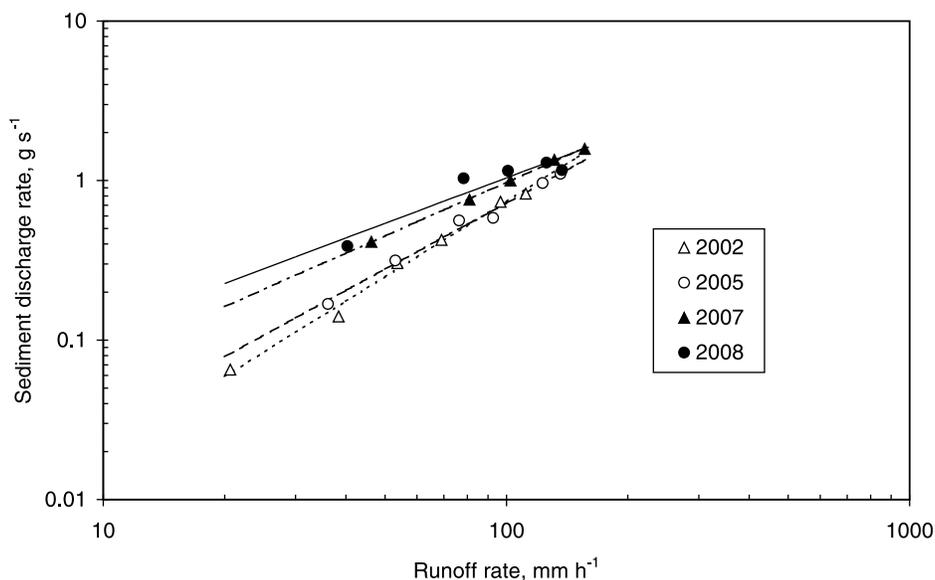

Figure 4. Steady state sediment discharge as a function of runoff rate on hillslope runoff plots dominated by native bunchgrasses (2002, 2005) and *E. lehmanniana* (2007, 2008).

Table 3. Regression Equation Coefficients for Sediment Yield From Runoff Plots ($q_s = \beta'_0 q_r^{\beta 1}$)[a]

Year	β'_0	β_1	n	R^2
2002	0.0005 ab	1.5696 a	6	0.99
2005	0.0013 b	1.3785 b	6	0.98
2007	0.0058 a	1.1142 b	5	0.99
2008	0.0132 a	0.9484 b	5	0.85

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

annual runoff on record (91th percentile). Increased runoff was attributed to a sharp decline in plant canopy cover (Figure 1), primarily due to die out of shrubs and native bunchgrasses. The presence of *E. lehmanniana* at that time was relatively small (1.3% cover) to have a significant effect on hydrological processes. Loss of plant cover in 2006 resulted in a lowered threshold for runoff generation. This increased the fraction of events that produce runoff to 23.4%, more than triple the long-term average (7.2%). The runoff to precipitation ratio during the 2006 events was double of that during other periods. In addition, average sediment concentration (3.4 g L^{-1}) was 3–30 times higher than in any other year (0.1–1.2 g L^{-1} in 2007 and 1996, respectively). All these resulted in the function of sediment yield versus runoff for 2006 having a much steeper slope than for other periods (Figure 3). However, after the recovery of perennial grass cover in 2007 and beyond (Figure 1), the slope of the regression decreased to the pretransition level (Figure 3).

[36] Experiments on runoff plots were not conducted in 2006, when according to permanent transects vegetation cover was at its minimum, complicating a direct comparison between watershed and plot data. However, Figure 4 clearly shows the difference between the intercepts of the regression lines before the vegetation transition (2002, 2005) and after the spread of *E. lehmanniana* (2007, 2008). This difference is statistically significant and indicates that after the recovery of vegetation in 2007, hillslope erosion rates have not returned to their pretransition levels.

[37] Examination of the native bunchgrass and *E. lehmanniana* growth patterns might provide an insight into the difference between plot and watershed hydrologic response to the vegetation change. Basal area measurements made on the watershed in 2007 indicate that *E. lehmanniana* was more numerous than native bunchgrasses with 12 times more individual plants per unit area but with an average individual plant basal area of only 3 cm^2, compared to 11.7 cm^2 for native bunchgrasses. At the same time, canopy cover of the native and invasive species was similar (13% and 19%, respectively). Native bunchgrasses tend to develop large clumps of stems, which create effective obstructions for water flow and induce formation of debris dams. The dams back up water thus allowing for greater infiltration [*Nearing et al.*, 2007]. Before *E. lehmanniana* invasion, this resulted in a microtopography characteristic for the Kendall site where small terraces formed upslope of large clumps of vegetation. With die out of native grasses and greater spread of *E. lehmanniana*, there were fewer obstructions, which allowed water to move down the slope more rapidly, likely increasing runoff and sediment yield (Figure 4).

[38] This mechanism, however, did not lead to statistically significant increases in sediment yield on the watershed

scale after the recovery of vegetation and spread of *E. lehmanniana*. This might have happened due to topographical features of watershed 112, which has a swale at the base of the slope above the flume that serves as a depositional zone. *Nearing et al.* [2005] found that the long-term (1963–2004) average annual sediment delivery ratio on the same watershed was essentially zero with most of the sediment eroded from the hillslopes deposited in the swale. This zone might have trapped some of the sediment generated on the hillslopes, thus alleviating some of the effect of vegetation change on sediment generation. It is plausible that due to the loss of vegetative cover in 2006, the swale area temporarily changed from a net deposition zone to a zone of transport or net loss. Plant canopy recovery in 2007 (Figure 1) allowed the swale to resume sediment yield attenuation. Aerial photographs and on-site observation show that swale soils support more grass growth than surrounding slopes, which also might have helped to trap sediment. Because of buffering function of the swale, it might take more time for the sediment effects of vegetation change to become apparent at the watershed scale.

[39] Long-term monitoring was found to be essential for accurate characterization of watershed processes [*Moran et al.*, 2008]. Without long-term monitoring, the spike in sediment yield would not have been identified, and neither would the rapid recovery to predrought sediment yield values (Figure 3). Swales such as found on watershed 112 are common on small watersheds at higher elevations in WGEW; lower-elevation watersheds, however, tend to be more incised and have a more efficient sediment transport mechanism [*Nearing et al.*, 2007] possibly due to recent down-cutting and lowering of the base level within WGEW in general [*Osterkamp*, 2008]. Such landform features are important for watershed hydrological function, however evaluating health, and integrity must be done at both hillslope and watershed scale. Climate predictions indicate an increase in frequency of extreme climate conditions, both in terms of severity of drought, and the size and intensity of precipitation events [*Christensen and Hewitson*, 2007; *Easterling et al.*, 2000a; *Easterling et al.*, 2000b; *Seager et al.*, 2007]. Such dramatically fluctuating conditions are likely to have a strong impact on arid and semiarid plant community structure and ecosystem processes [*Fay et al.*, 2003; *Fay et al.*, 2008; *Heisler-White et al.*, 2009; *Knapp et al.*, 2008]. Hence, conserving key morphological features and plant community composition in semiarid and arid land watersheds will be critical in maintaining the integrity of rangeland health under changing climate conditions.

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