

Carbon loss by water erosion in drylands: implications from a study of vegetation change in the southwest USA

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Review

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5 **Carbon loss by water erosion in drylands: implications from a study of vegetation**
6 **change in the southwest USA**
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8 **Brazier, R.E., Turnbull, L., Wainwright, J. and Bol, R.**
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10 **Keywords**
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15 **Abstract**
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17 Soil organic carbon (SOC) is an important component of the global carbon cycle, yet is rarely
18 quantified adequately in terms of its spatial variability resulting from losses of SOC due to
19 erosion by water. Furthermore, in drylands, little is known about the effect of widespread
20 vegetation change on changes in SOC stores and the potential for water erosion to redistribute
21 SOC around the landscape especially during high-magnitude runoff events (flash floods).
22 This study assesses the change in SOC stores across a shrub-encroachment gradient in the
23 Chihuahuan Desert of the south west USA. A robust estimate of SOC storage in surface soils
24 is presented, indicating that more SOC is stored beneath vegetation than in bare soil areas. In
25 addition, the change in SOC storage over a shrub-encroachment gradient is shown to be non-
26 linear and highly variable within each vegetation type. Over the gradient of vegetation
27 change, heterogeneity of SOC increases and newer carbon from C₃ plants becomes dominant.
28 This increase in heterogeneity of SOC is related to an increase in water erosion and SOC loss
29 from intershrub areas, which is self-reinforcing. Shrub-dominated drylands lose more than
30 three times as much SOC as their grass counterparts. The implications of this study are
31 twofold: 1. Quantifying the effects of vegetation change on carbon loss via water erosion and
32 the highly variable effects of land degradation on soil carbon stocks is critical. 2. If landscape
33 scale understanding of carbon loss by water erosion in drylands is required, studies must
34 characterise heterogeneity of ecosystem *structure* and its effects on ecosystem *function* across
35 ecotones subject to vegetation change.
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INTRODUCTION

Degradation in drylands and soil organic carbon losses

Terrestrial storage of soil organic carbon (SOC) far outweighs the storage of carbon in vegetation. While the carbon content of dryland soils is generally lower than other ecosystems (Figure 1), because of the significant area that drylands occupy, they contribute considerably to global terrestrial carbon storage. In addition, large regions of drylands are undergoing vegetation change, often in the form of woody species invading native grasslands, and more recently, exotic grasses invading native grasses (Biedenbender et al. 2004). Both of these types of vegetation change have been shown to cause increases in runoff and erosion (Turnbull et al., 2010a; Wainwright et al., 2000; Wilcox et al., 2012), and thus, potential increase in carbon loss. Changes in vegetation and resulting changes in SOC storage, redistribution and loss from drylands therefore has the potential to alter the carbon balance of these systems, which may potentially have global-scale biogeochemical feedbacks.

Figure 1a. Shows potential for global carbon storage in soils. Figure 1b. shows the carbon storage in soils for drylands only. Dryland soils are estimated to store up to 300 tonnes of carbon per hectare with larger potential carbon storage than vegetation, illustrating that carbon in drylands is stored predominantly in the soil. After: <http://www.wri.org/publication/content/8242>

The sources of error surrounding both estimates of carbon stores and the loss of carbon via erosion processes from dryland soils are numerous. First, carbon content of soils is highly variable in space (Conant et al. 1998; Schlesinger et al. 1996) even within the same soil series. This heterogeneity ensures that unless detailed, geospatial sampling strategies are employed; assessments of average soil-carbon stores and losses can be misleading. Secondly, coarse resolution, point measures of carbon storage are often used to infer what the wider, landscape-scale storage may be, via some form of upscaling or extrapolation (for example see Schuman et al. 2002). As there may be broad controls on SOC storage, such as changing vegetation type which is not assessed at the point scale, approaches to estimate large-scale carbon stores may also be flawed and inherently uncertain. Thirdly, losses of SOC are often extrapolated to larger scales using ‘delivery ratio’ approaches whereby the proportion of material lost is scaled by an empirical constant, or fixed percentage, Lal (2003) uses a sediment delivery ratio of 10% and an SOC content of eroded soil of 2-3%, for example. Such approaches assume that the delivery ratio is constant through time, or at least is fixed over the period of time that is observed, and therefore are not explicit about the spatio-temporal variability in carbon loss that will occur on an event basis. Intuitively, losses of carbon associated with eroded sediment should be highly variable with factors such as soil type and land use, as Quinton et al. (2006) suggest. However, typically, such variability and the error that it leads to in estimating carbon stores and losses is not quantified. Furthermore, Parsons et al. (2006) have questioned the very basis for extrapolation based on the delivery-ratio concept, in that it scales in a way that we know is incompatible with the scaling of soil-erosion rates themselves (Parsons et al, 2004, 2006; Wainwright et al., 2000, 2008).

This paper focuses on the semi-arid grasslands of the southwestern USA which have experienced the encroachment of woody species over large areas (Buffington and Herbel, 1965). Though it has been suggested that encroachment of woody species into semi-arid grasslands has a positive effect on SOC storage overall (Jackson et al. 2002), in view of the uncertain effects of vegetation change on soil erosion and thus SOC storage, the effects of vegetation change on SOC budgets are thus highly variable and poorly understood. The semi-arid grasslands of the southwestern USA experience high intensity, monsoon rainfall régimes,

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3 and high rates of overland flow and erosion that may lead to irreversible land degradation
4 (Turnbull et al. 2008b). In these areas, the spatial heterogeneity of resources increases as
5 woody shrubs encroach into grasslands forming islands of fertility around the woody shrubs
6 (Schlesinger et al. 1990; Kieft et al., 1998). Such resource islands may retain carbon, for
7 example, more effectively than inter-shrub areas (Tongway and Ludwig, 1990; Ludwig et al.,
8 2000), but little is known about the overall net effect of degradation and vegetation change on
9 net carbon storage (loss or gain). To understand how shrub encroachment affects the retention
10 and loss of SOC, we first need to quantify how shrub encroachment affects the spatial
11 distribution of SOC, and how shrub encroachment alters the magnitude and the spatial
12 distribution of eroding areas.
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15 Turnbull et al. (2010a) studied a continuum of vegetation change across a shrub-
16 encroachment gradient the northern Chihuahuan desert in New Mexico. Sites were
17 established to quantify rainfall/runoff and soil-erosion fluxes from a pristine black grama
18 (*Bouteloua eriopoda*) grassland site, through two intermediate sites with both black grama
19 grass and creosotebush shrub (*Larrea tridentata*), to a creosotebush-dominated shrubland
20 site. Results from this study showed that as grass cover decreases and shrub cover increases,
21 the relationship between rainfall and runoff becomes more directly coupled. Higher runoff
22 coefficients occur on shrubland than grassland, demonstrating that the loss of water from
23 semi-arid landscapes may be increased as the land becomes progressively more degraded.
24 Turnbull et al. (2010a) also reported that the total amount of erosion increased as runoff
25 coefficients increased, and that total erosion increased across the transition from grass to
26 shrub-covered plots. Grass-dominated plots yielded a significantly lower mass of sediment
27 per unit of runoff than the shrub-dominated plots. These results suggest that losses of soil
28 organic carbon may also increase as a result of shrub encroachment into grassland. Thus,
29 SOC loss from hillslopes may be highly variable across the landscape depending on the level
30 of shrub encroachment and land degradation, evidenced by increases in soil erosion.
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33 34 *Aim and objectives*

35 The aim of this paper is to build upon the work of Turnbull et al., (2010a,b, 2011), to explore
36 the relationships between the spatial distribution of SOC, overland flow, erosion and
37 associated carbon loss across a shrub-encroachment gradient in a semi-arid region of the
38 southwestern USA. We quantify how the spatial variability of SOC pools changes over the
39 shrub-encroachment gradient, and how soil carbon loss also changes over the shrub-
40 encroachment gradient under the influence of high intensity monsoon rainfall events.
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43 The objectives of the paper are:

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45 1. To assess the storage of soil organic carbon across a transition from pristine grassland
46 to heavily degraded shrubland;
- 47 2. To analyse the changes in spatial variability and type of SOC across the vegetation
48 transition;
- 49 3. To quantify redistribution of SOC across the ecotone via the abiotic processes of
50 rainfall, runoff and erosion.
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52 53 **METHODS**

54 *Site description*

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56 The Sevilleta National Wildlife Refuge (SNWR) has been designated a wildlife refuge since
57 1973 and was established as a Long Term Ecological Research (LTER) site in 1988, (Gosz,
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1993). The site is located in central New Mexico, USA (34°19' N, 106°42' W), draining into the Rio Grande (Figure 1a). The region has a semi-arid climate, with long-term average annual precipitation of ca. 256 mm, the majority of which falls as intense rainfall in a summer monsoon period (June – September) (Dahm and Moore (1994). The SNWR is located at the northern margin of the Chihuahuan desert, and is a transition zone between four major biomes: the Great Plains grassland, the Great Basin cool shrub steppe, the Mogollon Coniferous woodland and the Chihuahuan warm-temperate semi-desert. Soil series are shallow and classified as Turney Loams overlying a well-developed calcium carbonate layer which occurs between 0.25 and 0.45 m below the soil surface (Turnbull *et al.*, 2008a).

Long-term ecological monitoring at the Sevilleta indicates that large areas of grassland dominated by the perennial bunchgrass black grama have changed to domination by woody shrubs – especially creosotebush – a transition that has been widely documented across the southwestern USA in the last 200 years (e.g. Buffington and Herbel, 1965). The vegetation change that has taken place at Sevilleta has been accompanied by a shift in ecosystem structure at the landscape level (Cross and Schlesinger, 1999). Aerial photographs of the creosotebush to grassland ecotone covering the period 1935 to 1984 indicate that creosotebush clumps have increased and extended their range into the grassland (Gosz, 1993), thus indicating that the grass-shrub ecotone is dynamic. The δ^{13} carbon signature of soil organic carbon measured at stages over this grass-shrub ecotone supports the interpretation that creosotebush (C_3 vegetation) occupy soils that were once dominated by C_4 grasses (Turnbull *et al.*, 2008a).

Experimental design

Four sites were set up over the shrub-encroachment gradient. These sites (Figure 2a.) were selected to represent different stages over the shrub-encroachment gradient, such that plot 1 was the grass end-member, plot 2 was a grass (dominant) - shrub mix, plot 3 was a shrub (dominant) and grass mix, and plot 4 was the shrub end-member. At each site, a 10 × 30 m plot was built to measure runoff, erosion and nutrient export, as described in Turnbull *et al.* (2010a). Characterization plots were also set up, comprising two 5 m × 30 m areas at either side of the monitoring plots. The plots were located within 1 km of each other, on Turney loam soils and with similar aspects and plan-planar slope form with slope gradients ranging between 1.8° and 3.8° (Table 1). Measurement of soil characteristics and runoff monitoring took place during two monsoon rainy seasons (2005 and 2006), during which 17 high intensity, rainfall/runoff events were measured across all four sites.

Table 1. Vegetation and soil characteristics of the four study sites over the shrub-encroachment gradient.

Assessing storage of soil organic carbon (SOC) across the transition

To address objective one, soil and vegetation characteristics were assessed via near-ground remote sensing and a nested geostatistical soil sampling strategy across each of the four sites. At each site a number of variables were assessed, including the spatial properties of vegetation cover, soil bulk density, total soil organic carbon, and the δ^{13} carbon ($\delta^{13}C$) content of the soil.

The spatial properties of vegetation, including the proportion of the plot classified as bare soil, grass-covered or shrub-covered were observed using a Canon 6-megapixel digital camera suspended using a mobile platform on a cable above the ground to capture images of

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3 the ground surface. Control points were marked on the ground, to ensure that each of the 75
4 images for each plot covered six fixed locations. Control points were surveyed using a Leica
5 1200 digital level, enabling the imagery to be geocorrected and combined using Erdas
6 Imagine™ 8.0 image processing software (Figure 2a.). The resulting images were then
7 processed in ENVI 4.0 to map the surface cover of each plot manually. Images were taken
8 immediately prior to the onset of the 2005 monsoon season.
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11 Soil samples were taken at 90 locations within each 300 m² characterization plot (Figure 2b)
12 using a nested-grid of sample points to characterize both the short and medium range
13 variations in soil-surface characteristics, in an effort to avoid scale-dependent
14 characterization (Thomas and Kunin, 1999). Soil bulk density, was assessed by driving a
15 cylindrical tin with a volume of 2.21×10^{-4} m³ (0.05-m depth, 0.075-m diameter) into the
16 soil. The soil was excavated from around the tin and a pointing trowel was used to slice the
17 tin out of the soil so that the soil surface was flush with the tin. The soil sample was then
18 dried at 105°C for 24 h and weighed. Bulk density was calculated as the mass of soil divided
19 by the volume of the sampling tin.
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22 The total organic carbon content of the soil was measured by taking a sub-sample of the bulk
23 density samples and sieving through a 2-mm screen. The sub-2 mm fraction was ground to a
24 fine powder. As the soils were rich in carbonate, inorganic carbon was removed from the soil
25 using 75 ml of 2M HCl added to approximately 5 g of ground soil, left for 1 week, and then
26 filtered through a glass-fibre filter paper and washed three times with 100 ml deionised water
27 to remove the HCl. Samples were then air-dried prior to analysis.
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30 The $\delta^{13}\text{C}$ values of each of the 360 soil samples were analysed at the Rothamstead Research -
31 North Wyke laboratories using a NA 1500 elemental analyzer (Carlo Erba, Milan, Italy) and
32 an automated continuous flow ANCA 20/20SL system (Europa, Crewe, UK). The natural
33 abundance values were expressed as δ values, which represents the ratios of $^{13}\text{C}/^{12}\text{C}$ relative
34 to the international VPDB and AIR standard, respectively. The $\delta^{13}\text{C}$ values and proportion of
35 carbon derived from C₄ sources was estimated as in Turnbull et al. (2008a).
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38 To determine statistically significant differences in soil characteristics between different
39 surface-cover types at each site, t tests were used. Independent sample t tests were used to test
40 for significant difference between two different surface cover types (plots one and four) and
41 Tamhane's t test for significant difference between three different surface-cover types (plots
42 two and three).
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45 To explore the differences in SOC storage between each plot and beneath different surface
46 cover within each plot, values of SOC from the soil taken for bulk density analysis were
47 calculated. The area of each plot under each cover type was then multiplied by the average of
48 these SOC values for each cover type. Total SOC storage was then calculated for each plot to
49 a depth of 0.05 m.
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51 *Analyzing the changes in spatial variability of SOC across the vegetation transition*

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54 To address objective two, geostatistical analysis was conducted on the SOC and $\delta^{13}\text{C}$
55 datasets, following Turnbull et al., (2010b). Geostatistical analysis was also performed on
56 the vegetation data which were binary-coded to distinguish between vegetated and non-
57 vegetated areas and allow comparison of variability between vegetation and SOC types. The
58 spatial variability within each site was assessed by determining the extent to which each
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3 variable was spatially autocorrelated (Olea, 1999). The scale at which surface characteristics
4 are scale dependent was determined by calculation of the semi-variogram (e.g. Rossi *et al.*,
5 1992) and summary statistics including the range of autocorrelation were compared between
6 plots.
7

8 ***Quantifying the redistribution of organic carbon across the ecotone***

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10 To address objective three, for each rainfall event at each plot, all overland flow was routed
11 to a large stock-tank, which captured the total eroded sediment leaving each plot. Seventeen
12 runoff events were monitored; between four and six from each plot. During these events, all
13 overland flow, sediment and carbon was captured at the plot outlet. Total mass of sediment
14 was recorded for each event. Sediment was sieved to less than 2 mm, riffled and then sub-
15 sampled to analyze for total organic carbon content of eroded sediment. The total organic
16 carbon content of the eroded sediment was analysed as for the soils described above. The
17 experimental design of the flow-monitoring sites is described in more detail in Turnbull *et al.*
18 (2010a).
19

20 **RESULTS**

21 ***Storage of soil organic carbon across the vegetation transition***

22
23 The mean SOC values for each plot weighted by vegetation cover do not vary significantly (p
24 < 0.05), ranging from 2.54 mg cm⁻³ to 3.28 mg cm⁻³ (Table 2). The grass/shrub plot contains
25 the highest cover weighted average SOC, whereas the shrub/grass plot contains the lowest
26 average SOC value. The organic carbon content of the bare soil (Table 2) is similar across all
27 plots, varying from 2.34 (± 0.20) mg cm⁻³ on the grass plot, to 2.29 (± 0.19) mg cm⁻³ on the
28 shrub plot and showing no significant difference at $p < 0.05$. However, the organic carbon
29 content of the soil underlying the shrubs (on plot 2 = 6.72 \pm 0.59 mg cm⁻³, plot 3 = 6.09 \pm
30 0.64 mg cm⁻³ and plot 4 = 5.81 \pm 0.41 mg cm⁻³) was significantly greater than the organic
31 carbon content of the bare soil and the soil underlying the grass cover ($p < 0.05$). These
32 results suggest that there is more organic carbon stored beneath shrubs than in inter-shrub
33 areas, whether the intershrub areas are covered in grass or are devoid of vegetation.
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36 The isotopic signature of the soils, which underlie shrubs, was always lower than the
37 signature for soils beneath grass cover. $\delta^{13}\text{C}$ values for soils beneath shrubs ranged
38 from -20.94 \pm 0.28 ‰ on the shrub-grass plot to -22.76 \pm 0.18 ‰ on the shrub plot, whereas
39 $\delta^{13}\text{C}$ values for soils beneath grasses ranged from -18.58 \pm 0.30 ‰ on the shrub-grass plot
40 to -19.43 \pm 0.24 ‰ on the grass-shrub plot. These differences, between shrub soils and grass
41 soils were significant at $p < 0.05$.
42

43 The bare soils showed similar isotopic signatures on all plots, though the $\delta^{13}\text{C}$ values were
44 lowest, -18.58 \pm 0.13 ‰ on the grass plot and highest, -21.51 \pm 0.19 ‰ on the shrub plot. The
45 intermediate plots showed a mixed signal, particularly in the bare soils, with values of 19.65
46 \pm 0.16 ‰ for bare soils on the grass/shrub plot and 18.46 \pm 0.27 ‰ on the shrub/grass plot
47 illustrating the potential for C₃ carbon to be mixed with C₄ carbon even in the bare soils
48 between vegetation patches.
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3 **Table 2. Data describing the soil characteristics for SOC, $\delta^{13}\text{C}$ and the geostatistical**
4 **properties of vegetation. Data used in these analyses had outliers removed (see Turnbull**
5 **et al. 2011 for approach used)**
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9 **Table 3. Total estimated SOC storage within each plot for soils beneath each type of**
10 **surface cover to a depth of 0.05 m. SOC storage (kg) reflects the amount of SOC in each**
11 **plot under each surface cover type (means and standard deviations calculated from all**
12 **sample points, $n = 90$ for each plot). SOC storage (g m^{-2}) for each surface cover**
13 **illustrates variability of carbon stores across the ecotone, and total values for each plot**
14 **underline the importance of using such an area-weighted approach both to describe**
15 **SOC that is available for erosion and to support extrapolation of SOC storage data to**
16 **larger scales.**
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20 Data presented in Table 3 describe the total storage of SOC in each plot and the area-
21 weighted storage according to vegetation type. These data suggest that total SOC within the
22 surface soils is greatest on the grass/shrub plot and least on the shrub/grass plot, whilst SOC
23 storage is higher on the shrub plot than the grass plot. There is no significant difference
24 between these SOC totals at $p < 0.05$ and therefore there is no significant trend in SOC
25 storage observed across the transition from grass to shrub.
26

27
28 The data indicate a decline in the total SOC content under grass between the grass,
29 grass/shrub and shrub/grass plots, from 76.21 via 71.8 to 28.81 g m^{-2} respectively, but an
30 increase in the SOC under grass per unit area from 167.50 via 186.00 to 201.50 g m^{-2} .
31 Conversely, as the shrubs become more established, covering larger areas of each plot, the
32 SOC content of the soil beneath shrubs decreases from 336.00 g m^{-2} on the grass/shrub plot,
33 via 304.50 g m^{-2} on the shrub/grass plot to 290.50 g m^{-2} on the shrub plot (significant at $p <$
34 0.05).
35

36 *Spatial variability of SOC across the vegetation transition*

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38 Results of the spatial analysis of SOC across the ecotone illustrate significant differences in
39 the degree of autocorrelation between variables in the dataset. The range at which soil
40 organic carbon is autocorrelated (0.8, 2.2, 1.2, 1.4 m from grass to shrub sites respectively) is
41 greater than the range at which vegetation is autocorrelated (0.7, 0.7, 1.1, 0.9 m) over all
42 plots, although only marginally so over plots 1 and 3 (Table 2). On plot 2, SOC is
43 autocorrelated at a range more than three times the range at which vegetation is
44 autocorrelated (2.2 versus 0.7 m). The strength of autocorrelation, is slightly lower over plot
45 1 than plots 2, 3 and 4 which all have similar ranges, despite there being very obvious
46 changes in both the types and spatial structure of vegetation across the plots.
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50 Results of the geostatistical analysis of $\delta^{13}\text{C}$ data (Table 2), showed that the range at which
51 $\delta^{13}\text{C}$ is autocorrelated over plot 1 (7.1 m) is an order of magnitude greater than the range at
52 which vegetation is spatially autocorrelated (0.7 m). In addition, the range at which $\delta^{13}\text{C}$
53 is autocorrelated over plots 2 to 4 is also greater than the range at which vegetation is spatially
54 autocorrelated (ranges of 2.3, 1.1 and 1.8 m for grass/shrub, shrub/grass and shrub plots
55 respectively, compared to 1 m or less for vegetation ranges). Soil $\delta^{13}\text{C}$ is strongly
56 autocorrelated over plot 1, with a nugget variance of 0.1. Plots 2, 3 and 4 show a moderate
57 strength of spatial autocorrelation, with nugget variances of 0.4, 0.55 and 0.4 respectively.
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4 The high range of 7.1 from the grass plot $\delta^{13}\text{C}$ values highlights the fact that the maximum
5 scale of spatial variation in the grassland is significantly higher than on any of the mixed
6 vegetation plots. Such a result suggests that the SOC values in the grassland are more
7 homogeneous than where mixed vegetation species are evident in the other plots. Clustering
8 of similar $\delta^{13}\text{C}$ values in the shrub-dominated plots is more evident, suggesting a more
9 heterogeneous distribution of SOC where shrubs are present.
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11 *Quantifying the redistribution of organic carbon across the ecotone*

12 *Runoff and carbon loss*

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16 Figure 3. illustrates the relationship between total runoff leaving each plot (l) and total loss of
17 organic carbon (g) for the 17 events monitored during the 2005 and 2006 monsoon seasons.
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20 **Figure 3. Total event runoff (l) and total mass of carbon lost per event (g) from each**
21 **plot. Solid lines represent best linear fits to data from each plot, to illustrate general**
22 **trend of increasing carbon loss with increasing discharge, but at decreasing rates across**
23 **the transition from shrub- to grass-dominated landscape.**
24

25 Analysis of the relationship between total event runoff and total carbon loss for each plot
26 shows an increasing loss of TOC with increasing flow from each plot. These linear
27 relationships are significant on all plots (grass: $r^2 = 0.84$, $p = 0.029$; grass-shrub: $r^2 = 0.8$, $p =$
28 0.246 ; shrub-grass: $r^2 = 0.99$, $p = 0.005$; shrub: $r^2 = 0.77$, $p = 0.079$). The slope of these
29 relationships increases with increasing level of shrub encroachment, grass = 0.7%, grass-
30 shrub = 1.6%, shrub-grass = 1.8%, shrub = 7.4%.
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33 Results show that the total losses of carbon are greater on the shrub-dominated plots, but also
34 that the mass of carbon per unit volume of runoff is higher when shrub vegetation dominates
35 over grass. Loads from all events recorded on the shrub and shrub grass plots range from
36 0.013 – 0.094 g C l⁻¹, whereas loads from the grass and grass-shrub plots range from 0.004 –
37 0.022 g C l⁻¹ and are significantly different ($t = 2.51$, $p < 0.05$) using a heteroscedastic t test,
38 assuming unequal variance. In addition, the total yield of carbon from the shrub plots may be
39 as much as six times higher (based on the largest flow events recorded) than the yield from
40 the grass plots. This results in a total yield of carbon (over all events; 10 from the grass plots,
41 7 from the shrub plots) from the shrub plots of 527 g (17.6 kg ha⁻¹) compared to 147 g (4.9 kg
42 ha⁻¹) from the grass plots.
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45 It is also instructive to look at the influence of the runoff coefficient (RC), which describes
46 the proportion of rainfall that left each plot as overland flow, on the loss of carbon from the
47 landscape. To address objective three, we explore whether increasing runoff coefficients will
48 lead to greater losses of carbon. As the same rainfall event falling on plots with different
49 vegetation cover would be expected to generate different runoff coefficients, (Turnbull et al.
50 2010a) the runoff coefficient may describe the effect that the changing structure of vegetation
51 (higher shrub canopies and denser foliage can intercept more rainfall than grasses, for
52 example) can have on the mobilisation of carbon. Figure 4. illustrates the relationship
53 between RC and total organic carbon loss (g) during each rainfall/runoff event from all plots.
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Figure 4. Total event runoff coefficient (RC) and total mass of carbon lost per event (g) from each plot. Solid arrow represents general trend of both increasing runoff and carbon loss across the transition from grass to shrub dominated landscape.

A power relationship fitted to the data ($r^2 = 0.78$, $p < 0.05$), describes a positive increase in carbon loss with an increased proportion of rainfall leaving the plot as overland flow. Critically, the increase in carbon loss is greater than the increase in runoff, across the plots, indicated by the exponent of 1.53. Thus, the process of carbon loss becomes more concentrated for a given unit of runoff, across the transition from grass to shrub-dominated landscape. In addition, high RCs lead to a wide range of potential carbon loss as is shown by the high level of variance in the data, particularly associated with the heavily degraded shrub plot, where the highest carbon losses (134 and 180 g per event) were observed. The general trends across the grass to shrub transition are therefore: (1) runoff coefficients increase, (2) total carbon loss increases at a faster rate than increases in runoff coefficient and (3) as runoff coefficients increase, so does the variability in carbon loss.

Erosion and carbon loss

Extending the analysis to consider the relationship between eroded sediment and carbon loss also shows a positive power relationship ($r^2 = 0.89$, $p < 0.05$) across all plots. As total erosion under each vegetation type increases, across the transition from grass to shrub vegetation, so does the amount of organic carbon lost from the system.

Considering erosion and carbon response from each plot separately, positive relationships between eroded sediment and carbon loss are evident over all the plots (Figure 5). These relationships are significant over all but the shrub plot (grass: $r^2 = 0.74$, $p = 0.039$; grass-shrub: $r^2 = 0.94$, $p = 0.004$; shrub-grass: $r^2 = 0.99$, $p = 0.011$; shrub: $r^2 = 0.31$, $p = 0.267$). The response of the shrub-dominated plots is similar, as is that of the grass-dominated plots, though the two pairs exhibit different behaviour when compared with each other. Total mass of organic carbon per unit of sediment lost from each plot varies from 0.011– 0.032 g C g⁻¹ with a mean of 0.019 g C g⁻¹ on the shrub plots, compared to 0.003 – 0.016 g C g⁻¹ with a mean of 0.008 g C g⁻¹ on the grass plots. A two-sample t test, assuming unequal variance demonstrates no significant difference at $p < 0.05$ between the amount of carbon lost from the two shrub plots or the carbon lost from the two grass plots. However, a significant difference is found ($t = 3.38$, $p < 0.05$) between the shrub-dominated plots and the grass-dominated plots. On average, shrub-dominated plots yield ca. 3 times more carbon per unit of eroded sediment, than their grass-dominated counterparts.

Figure 5. Total eroded sediment (g) and total organic carbon loss (g) from grass, grass-shrub, shrub-grass and shrub plots for all events monitored for carbon loss in 2005 and 2006. Shrub-dominated plots lose significantly more carbon than grass-dominated plots.

DISCUSSION

Storage and spatial variability of soil organic carbon across the vegetation transition

Total SOC

The greater organic carbon content of the soil underlying shrubs, when compared to that under grasses or bare soil suggests that concentration of carbon resources is taking place, as with nitrogen and phosphorus (Brazier, et al., 2007; Parsons et al., 2003; Schlesinger et al., 1996; 1999; 2000; Turnbull et al., 2010b) and sediment (Turnbull et al. 2010a) within this and comparable landscapes. Over the shrub-encroachment gradient, the heterogeneity of

resources including carbon increases, supporting the islands of fertility concept (Schlesinger et al., 1990). Such increasing heterogeneity is an important part of land degradation, inter-shrub areas become depleted of resources, whilst shrub-covered areas retain resources. The net effect of woody species encroachment may therefore be to increase SOC storage in the system, which is in agreement with most other research (summarized in Jackson et al. 2002), but here we also show that spatial distribution of SOC is fundamentally altered with woody species encroachment.

Results of the geostatistical analysis demonstrate a variable distribution of SOC across all sites, with the greatest range of spatial autocorrelation over the grass-shrub site (SOC range = 2.2 m, vegetation range = 0.7 m), which indicates that processes other than plant-soil feedbacks influence the distribution of SOC. Such findings are consistent with those of Turnbull et al. (2010b), who found that although there were biotic-abiotic feedbacks operating at the scale of individual plants, the spatial distribution of vegetation alone did not explain the distribution of nutrients in the soil. One explanation may lie in the variability of soil mineralogy across the vegetation transition, which can exert a strong control on SOC turnover rates independently of vegetation structure (Torn et al. 1997). Other workers have shown that SOC can turnover decadal (Townsend et al. 1996) well within the 200 years that is postulated for the woody shrub encroachment of the semi-arid grasslands studied here (Buffington and Herbel, 1965). Even if SOC turnover rates are relatively slow in semi-arid areas (Montaña et al. 1988) the input of high intensity, monsoon rainfall year on year might be enough to accentuate the heterogeneity of SOC levels across the transition.

C₃ versus C₄ SOC

To address Objective 2, the isotopic values of SOC were quantified, in order to estimate spatial variability of different types of carbon. The $\delta^{13}\text{C}$ for the soils underlying the shrubs were closer to those reported in Turnbull *et al.* (2008a) for the *Larrea tridentata* shrub end-member (ca. -26 ‰) than for the *Bouteloua eriopoda* end-member, which has an isotopic signature of -14.0 ‰. The soil carbon in the region of the shrubs is dominated by C₃ carbon derived from the woody vegetation and retained beneath these shrubs. Conversely, the $\delta^{13}\text{C}$ values beneath grass and in bare areas were nearer to the *Bouteloua eriopoda* grass end-member values, than the *Larrea tridentata* shrub end-member. This finding demonstrates the mixture of soil organic carbon that is found, particularly in the bare soils, which may previously have been covered by C₄ grasses, but which may now lie beneath concentrated flow paths between C₃ woody shrubs. Liao *et al.* (2007) report complementary findings from the Rio Grande Plains of southern Texas, USA, where C₃ trees and shrubs have encroached C₄ grasslands since western migration of European settlers. The authors demonstrate that carbon stocks increase in close proximity to woody vegetation due to the retention of older C₄ carbon and the capture of new, C₃ carbon beneath the plant canopy. Similarly, Kieft *et al.* (1998) found that SOC beneath *Larrea tridentata* was consistently higher than the SOC in grassland soils and also that soils beneath *Larrea tridentata* canopies consistently had the highest SOC values. Finally, Turnbull *et al.* (2011) showed that plant-soil interactions at the local scale modify soil properties, nitrogen and organic matter, which were found to be depleted in bare soils relative to grass and shrub-covered areas. The implications of these results are twofold: (1) The older, C₄ SOC in the soils remains buried under shrubs, as also implied by the work of Parsons *et al.* (1992), in A horizons which, since shrub encroachment, have accumulated younger C₃ carbon both due to deposition of leaf litter but also due to transport of carbon rich material beneath the shrub canopy by rainsplash and overland flow; (2) not all the SOC is flushed out of bare parts of the landscape immediately, so SOC change lags vegetation change, possibly by some decades. Such a time lag is important both for

analyzing carbon budgets (i.e. it is not possible simply to quantify land use and extrapolate SOC properties from it) and for mitigation purposes, as more SOC remains in the landscape as a legacy of past vegetation.

Controls on spatial distribution and variability of SOC

The fact that different ranges of autocorrelation were found between SOC values and vegetation cover indicates that processes other than simply vegetation-soil feedbacks control the spatial distribution of carbon across the shrub-encroachment gradient. Müller et al. (2007) working on an analogous semi-arid site in southern New Mexico revealed that autocorrelation lengths for soil moisture and infiltration (for example) were also not just controlled by vegetation (shrub size), but were related to the pattern of concentrated flowpaths between shrubs. We suggest that the abiotic (hydrological) controls of runoff and erosion – processes that lead to the redistribution of soil organic carbon in the landscape – are perhaps more important than the biotic controls related to vegetation structure or size. This conclusion is reinforced by the strength or degree of autocorrelation, which is consistent over the grass, grass-shrub and shrub-grass sites and is slightly greater over the shrub plot, despite there being very obvious changes to both the types and spatial structure of vegetation across the sites.

It has also been recognised that soil moisture can exert a strong control over ecosystem structure (Noy-Meir, 1973). Thus, it is possible that the spatio-temporal dynamics of soil moisture may be an important factor in controlling the spatial patterns of soil organic carbon observed here. Soil moisture data collected at these sites (Turnbull *et al.* 2010a), illustrated the varying spatial structure of soil moisture contents across the ecotone during the monsoon seasons of 2005 and 2006. These data showed that in general, soil-moisture contents are higher in bare surface soil than in soil under vegetation and that soil moisture under grasses was typically higher than soil moisture under shrubs. Therefore, where soil moisture is at its highest, soil organic carbon levels are at their lowest. We interpret these findings in two ways: (1) Higher antecedent soil moisture at the onset of a storm, particularly in bare soils, will lead to higher rates of overland flow, due to locally occurring saturation excess runoff generation, which in turn reinforces the spatial distribution of soil organic carbon in a similar pattern to the runoff and erosion. i.e. depletion of soil organic carbon from the bare soils, retention under vegetation. (2) Shrubs appear to be more efficient at removing water from the soil than grasses (Wainwright *et al.*, 2000) and subsequently returning carbon to the soil in the form of leaf litter (see Wainwright 2009 for discussion), which leads to elevated levels of C₃ soil organic carbon in the areas beneath shrubs, in excess of the areas beneath grass and areas of bare soil.

Quantifying the redistribution of organic carbon across the ecotone

The results suggest that as overland flow increases from each plot, so does carbon loss. In addition, the rate of carbon loss from the system increases across the transition from grass to shrub land, illustrating that the more degraded a semi-arid landscape becomes the more carbon it will lose. Previous researchers have studied the effects of vegetation change on ecosystem function from various perspectives. Abrahams *et al.* (1995) studied the effect of vegetation change on interrill erosion in semi-arid areas, concluding that once desertification commences a 'self-perpetuating' increase in the spatial heterogeneity of soil resources will continue. This work builds on the findings of Schlesinger *et al.* (1990) who concluded that increased spatial heterogeneity will lead to the formation of 'islands of fertility', due to the

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3 positive feedback of resource distribution, where resources will be concentrated, leading to
4 depletion of resources in inter-shrub areas.
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7 Turnbull *et al.* (2011) observe similar dynamics between overland flow and the loss of total
8 nitrogen (N) and total phosphorus (P), across the same grass to shrub transition as reported
9 here, suggesting that the progressive degradation of grasslands in semi-arid areas is likely to
10 result in significant alterations to macronutrient cycles that are primary controls on vegetation
11 growth. Working in a similar, semi-arid environment, Barger *et al.* (2006) found that the vast
12 majority of both total carbon and total nitrogen yields were correlated with soil erosion
13 losses. It is likely therefore that all of these nutrients are either bound to sediment particles or
14 behave very similarly, in the form of plant litter that is mobilised and transported by overland
15 flow. Thus, soil erosion, without intervention will continue to deplete carbon resources in
16 semi-arid soils.
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19 The proportion of rainfall that leaves each plot as overland flow also increases across the
20 transition from grass to shrubland. Parsons *et al.* (1996) studied two end-member (grass and
21 shrub) plots in southeast Arizona and demonstrated that the interrill areas of shrub-dominated
22 hillslopes yield both higher runoff rates and higher runoff velocities than their grass-
23 dominated counterparts. From our data, concurrent with this increase in overland flow, there
24 appears to be an increase in the carbon leaving the system, with shrub plots yielding up to
25 three times as much carbon as grass plots. Clearly, more energy is available via concentrated
26 hydrological flowpaths in shrub-dominated landscapes, as geomorphic features, such as
27 gullies or rills may develop (Cerdà 1997; Reid *et al.* 1999; Parsons *et al.*, 1996; Parsons and
28 Wainwright 2006), so it is likely that more soil organic carbon is entrained as these flowpaths
29 become more connected in a shrub-dominated landscape than in a grass dominated landscape.
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33 Data presented here demonstrate a strong relationship between soil erosion and soil organic
34 carbon loss, as the former increases, so does the latter at all stages over the shrub-
35 encroachment gradient, but to a greater extent when shrubs are the dominant vegetation type.
36 This finding shows that even during the intermediate stage of transition, where there is still
37 grass present in inter-shrub areas (here ca. 14% of the soil in the shrub-grass plot is covered
38 in grass), there is still a similar yield of carbon per unit of soil erosion as on the shrub plot,
39 where grass vegetation is no longer present. Other workers have shown variability in rates of
40 above ground carbon stores under single vegetation types such as *Prosopis glandulosa*
41 (honey mesquite) (Asner *et al.* 2003); in savanna grassland systems subject to vegetation
42 change, through long-term monitoring (Buffington and Herbel, 1965); or through soil organic
43 carbon isotope analysis of woody shrub invasion of grasslands (Boutton *et al.* 1998).
44 However, these workers did not observe the changes to carbon storage that were brought
45 about by erosion processes in interaction with vegetation change. What is shown here is that
46 *both* the stores and spatial distributions of carbon change across the shrub-encroachment
47 gradient *and* the rates of carbon loss also change, a finding, which underlines the significance
48 of water erosion on the loss of carbon from dryland environments.
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51 CONCLUSIONS

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54 A robust estimation of the changing carbon storage in semi-arid soils subject to vegetation
55 encroachment is presented. Objective one showed that more SOC is stored in soils beneath
56 shrubs than grasses and that soils beneath both vegetation types store more SOC than bare
57 soils in inter-plant areas. Results also suggested that while there are differences in the area-
58 weighted SOC storage across the grass to shrub transition, these differences are not consistent
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3 over the shrub-encroachment gradient. The total amount of SOC stored beneath the grasses
4 varied significantly across the transition, increasing in concentration, as the grass cover
5 became more sparse. These data illustrate that the surviving grasses in the landscape, perhaps
6 being better established and thus resilient to the processes of erosion are able to retain more
7 SOC, than their counterparts in the 'pristine' grass plot. Similar results are seen for shrubs,
8 where progressively more SOC is found in storage beneath the shrub canopies, as the shrubs
9 become more established in the landscape. Furthermore, because of the elevated topography
10 of the shrub mounds, runoff typically diverges into inter-shrub areas increasing erosion while
11 reducing erosion from beneath the shrubs. The consequent capture of resources beneath
12 shrubs and its accentuation in parallel with increasing erosion in inter-shrub areas exemplifies
13 the role that abiotic processes play in redistributing SOC around the landscape and enhancing
14 the heterogeneity of carbon storage within the soil.
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18 Results of this monitoring work also described the type of SOC that was present across the
19 transition, in terms of its isotopic signature, in order to determine whether the source of the
20 carbon stored in the soil (C_3 or C_4 plant derived carbon) differed across the vegetation
21 transition. Soil beneath vegetation tends to be dominated by carbon which has derived from
22 that vegetation type, C_3 carbon beneath shrubs and C_4 carbon beneath grasses. The bare areas
23 of soil are characterised by an intermediate carbon signal, reflecting the mixture of older,
24 grass-derived carbon and younger inputs of shrub-derived carbon from surrounding
25 vegetation.
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28 Objective two quantified the spatial variability of SOC across the vegetation transition.
29 Results showed that heterogeneity of SOC increases from a relatively consistent distribution
30 of SOC under grasslands, to a highly variable or patchy distribution under shrubs. It is clear
31 that shrub species afford good local protection to the soil against intense rainfall (via more
32 dense canopy cover) so they may conserve SOC by reducing the energy that is available to
33 mobilise sediment and carbon via erosion. Shrub species may also be more efficient at
34 supplying carbon to the soil. Shrubs are better able to capture soil moisture (soil moisture
35 levels are consistently lower beneath shrubs than elsewhere in the ecotone) and convert this
36 to C_3 -carbon via the annual deposition of leaf litter. Whilst die-back of grasses will contribute
37 some C_4 -carbon to the SOC pool, it appears to be significantly less than the C_3 -carbon
38 contribution.
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41 Objective three quantified redistribution of SOC across the ecotone due to water erosion
42 processes. Results suggested that greater losses of soil and carbon will occur as the ecosystem
43 degrades and shrubs become better established than grasses, with a concomitant increase in
44 the amount of bare soil that is present. Landscapes that are covered by shrub species such as
45 *Larrea tridentata* are therefore not only more susceptible to erosion, but will also lose more
46 carbon per unit of soil loss than their grassland counterparts. In part, this difference is driven
47 by the positive feedback that is established as a landscape degrades, whereby bare areas of
48 soil become progressively more eroded and depleted of nutrients, which makes them less
49 viable locations for vegetation to re-establish. In addition, the difference is controlled by the
50 increased spatial heterogeneity that is associated with woody shrub encroachment, when
51 compared to the more homogeneous distribution of resources that are found under grass
52 species such as *Bouteloua eriopoda*.
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56 The implications of land degradation on both contemporary and future rates of carbon loss
57 from drylands are shown to be important, especially when woody shrubs encroach pristine
58 grasslands. Across a grass-shrub ecotone, we show that changes in ecosystem *structure*, or
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3 pattern of carbon storage and its *function*, particularly the redistribution of carbon by
4 overland flow will lead to significant increases in the loss of soil organic carbon from the
5 system as woody shrubs become more dominant.
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For Peer Review

<i>Plot</i>	<i>Vegetation characteristics</i>			<i>Soil characteristics</i>			
	% vegetation cover	% grass cover	% shrub cover	% pebbles	% sand	% silt	% clay
Grass	45.5	45.5	0	27.8	50.8	18.8	2.6
Grass-shrub	43.0	38.6	4.4	20.8	56.2	20.2	2.8
Shrub-grass	26.2	14.3	11.9	48.8	32.6	17.1	1.5
Shrub	23.3	1.0	22.3	34.0	43.8	20.0	2.2

Table 1. Vegetation and soil characteristics of the four study sites across the Grass to Shrub transition.

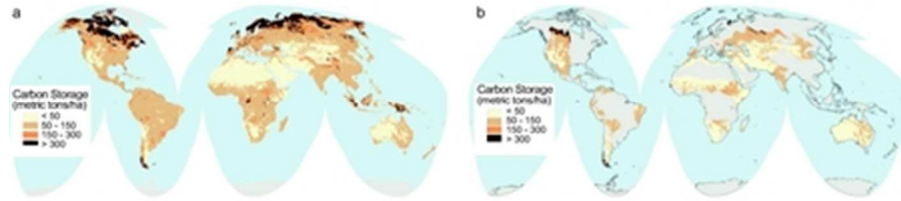
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Plot	Surface cover (<i>N</i> sampling points)	Surface cover (%)	Statistical/ geostatistical analyses	SOC (mg cm ⁻³)	$\delta^{13}\text{C}$	Vegetation
1	Bare (44)	54.5	Mean \pm S.E.	2.34 \pm 0.20 a	-18.58 \pm 0.13 a	
	Grass (46)	45.5		3.35 \pm 0.21 b	-19.09 \pm 0.18 b	
			Cover weighted mean	2.80 \pm 0.20	19.41 \pm 0.15	
			Range	0.8	7.1	0.70
			Nugget	0.65	0.1	0.65
			Sill	0.35	0.9	0.35
2	Bare (34)	57.0	Mean \pm S.E.	2.72 \pm 0.27 a	-19.65 \pm 0.16 a	
	Grass (31)	38.6		3.72 \pm 0.33 a	-19.43 \pm 0.24 a	
	Shrub (25)	4.4		6.72 \pm 0.59 b	-21.96 \pm 0.21 b	
			Cover weighted mean	3.28 \pm 0.31	-19.67 \pm 0.19	
			Range	2.2	2.3	0.70
			Nugget	0.5	0.4	0.65
			Sill	0.5	0.6	0.35
3	Bare (32)	73.8	Mean \pm S.E.	1.68 \pm 0.14 a	-18.46 \pm 0.27 a	
	Grass (35)	14.3		4.03 \pm 0.39 b	-18.58 \pm 0.30 a	
	Shrub (23)	11.9		6.09 \pm 0.64 c	-20.94 \pm 0.28 b	
			Cover weighted mean	2.54 \pm 0.24	18.77 \pm 0.28	
			Range	1.2	1.1	1.10
			Nugget	0.5	0.55	0.00
			Sill	0.5	0.45	1.00
4	Bare (47)	76.7	Mean \pm S.E.	2.29 \pm 0.19 a	-21.51 \pm 0.19 a	
	Shrub (43)	23.3		5.81 \pm 0.41 b	-22.76 \pm 0.18 b	
			Cover weighted mean	3.11 \pm 0.24	21.80 \pm 0.50	
			Range	1.4	1.8	0.90
			Nugget	0.5	0.4	0.50
			Sill	0.5	0.6	0.50

Table 2. Data describing the soil characteristics for SOC, $\delta^{13}\text{C}$ and the geostatistical properties of vegetation. Data used in these analyses had outliers removed (see Turnbull et al. 2011 for details)

Plot	Surface cover	SOC storage (kg)	SOC storage (g m^{-2})
Grass	Bare	19.13 \pm 1.65	117.00 \pm 10.00
	Grass	22.86 \pm 1.43	167.50 \pm 10.50
	Total	41.99 \pm 1.54	139.98 \pm 10.09
Grass/shrub	Bare	23.26 \pm 2.31	136.00 \pm 13.50
	Grass	21.54 \pm 1.91	186.00 \pm 16.50
	Shrub	4.44 \pm 0.39	336.00 \pm 29.50
	Total	49.23 \pm 1.15	164.10 \pm 14.84
Shrub/grass	Bare	18.60 \pm 1.55	84.00 \pm 7.00
	Grass	8.64 \pm 0.84	201.50 \pm 19.50
	Shrub	10.87 \pm 1.14	304.50 \pm 32.00
	Total	38.11 \pm 1.18	127.04 \pm 12.60
Shrub	Bare	26.35 \pm 2.19	114.50 \pm 9.50
	Shrub	20.31 \pm 1.43	290.50 \pm 20.50
	Total	46.65 \pm 1.81	155.51 \pm 11.52

Table 3. Total estimated SOC storage within each plot for soils beneath each type of surface cover to a depth of 5 cm. SOC storage (kg) reflects the amount of SOC in each plot under each surface cover type (means and standard deviations calculated from all sample points, n = 90 for each plot). SOC storage (g m^{-2}) for each surface cover illustrates variability of carbon stores across the ecotone, and total values for each plot underline the importance of using such an area-weighted approach to describe SOC that is available for erosion.

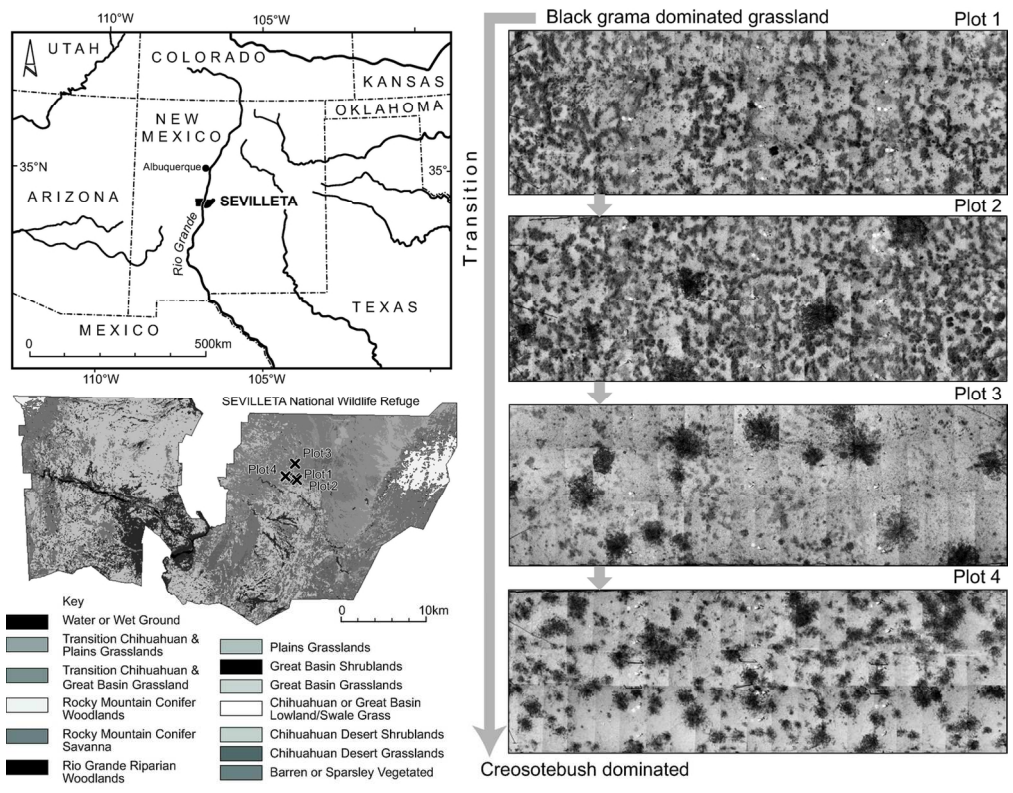


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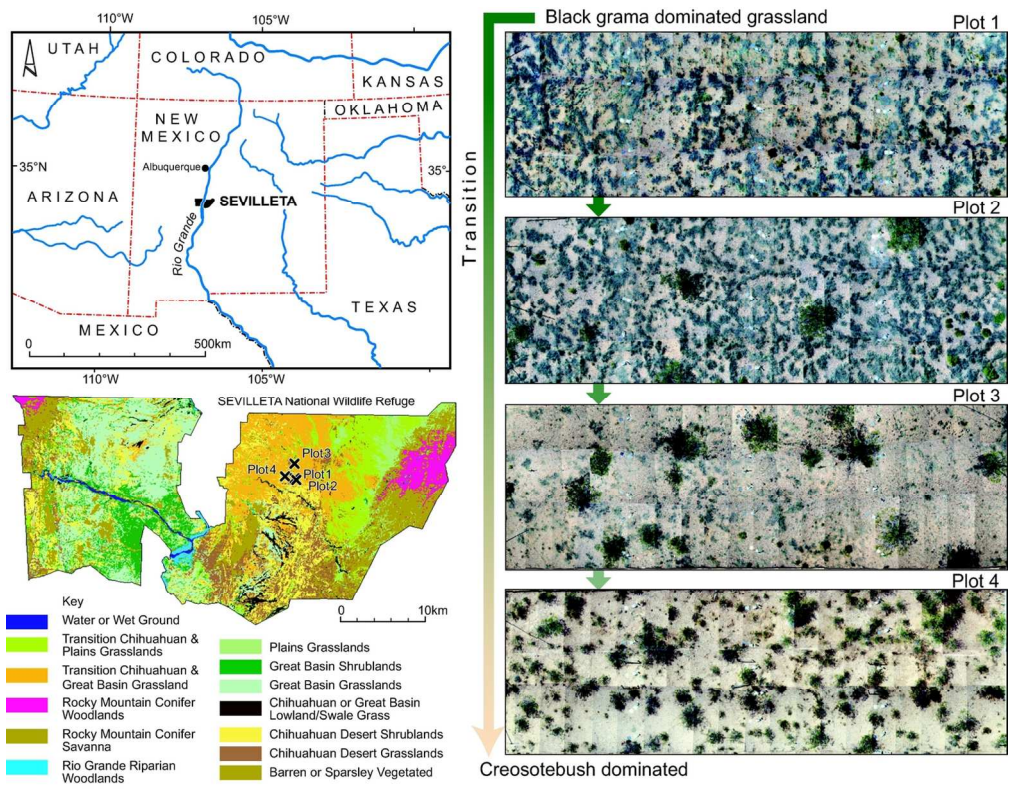
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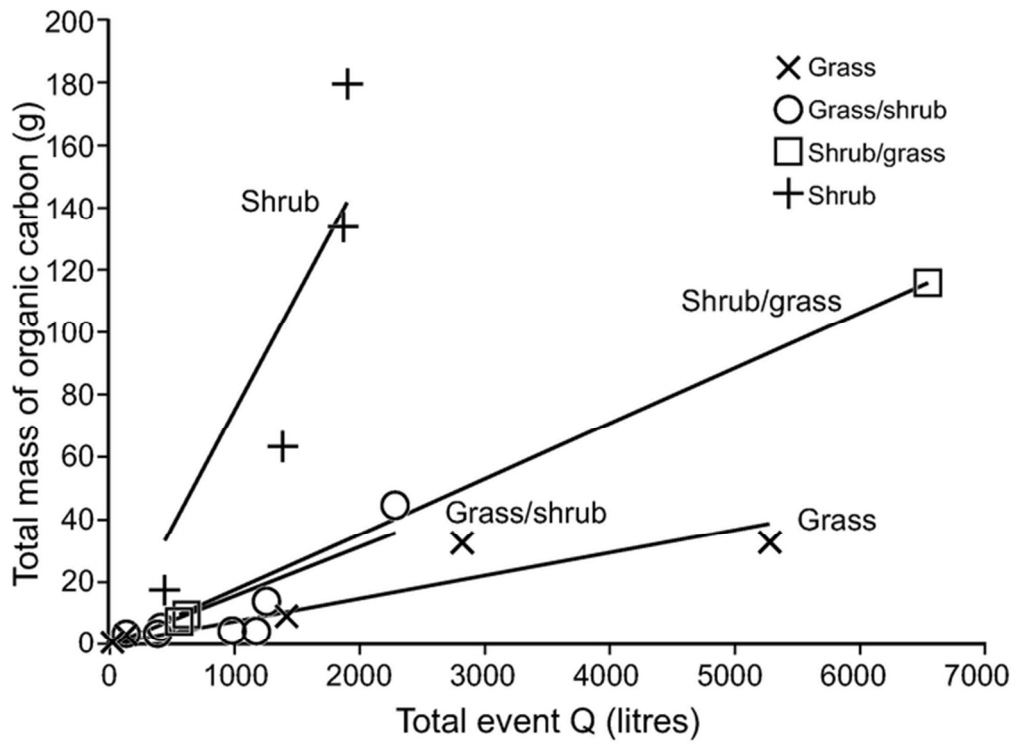
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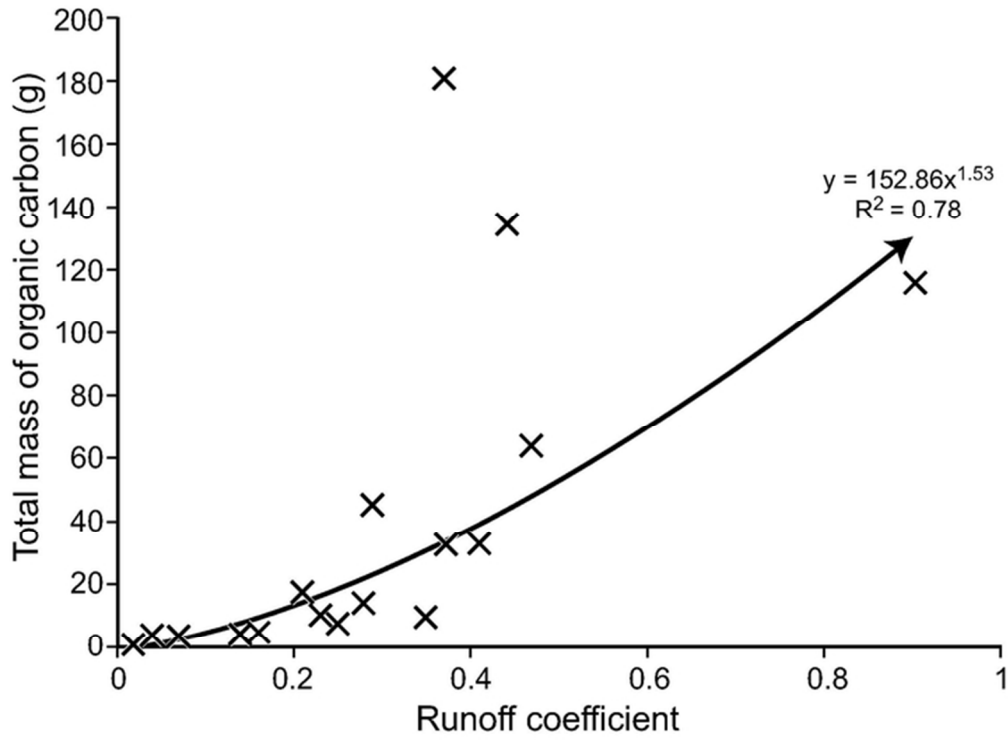
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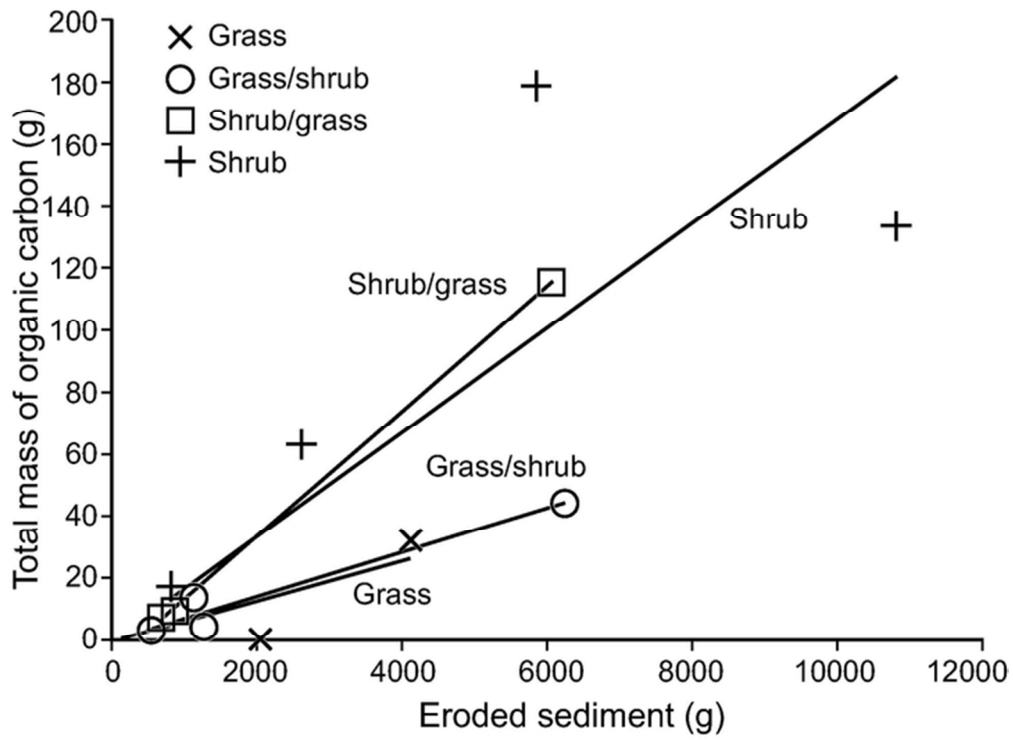


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