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

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Abstract:
Soil organic carbon (SOC) is an important component of the global carbon cycle yet is rarely quantified adequately in terms of its spatial variability resulting from losses of SOC due to erosion by water. Furthermore, in drylands, little is known about the effect of widespread vegetation change on changes in SOC stores and the potential for water erosion to redistribute SOC around the landscape especially during high-magnitude run-off events (flash floods). This study assesses the change in SOC stores across a shrub-encroachment gradient in the Chihuahuan Desert of the south-west USA. A robust estimate of SOC storage in surface soils is presented, indicating that more SOC is stored beneath vegetation than in bare soil areas. In addition, the change in SOC storage over a shrub-encroachment gradient is shown to be nonlinear and highly variable within each vegetation type. Over the gradient of vegetation change, the heterogeneity of SOC increases, and newer carbon from C3 plants becomes dominant. This increase in the heterogeneity of SOC is related to an increase in water erosion and SOC loss from inter-shrub areas, which is self-reinforcing. Shrub-dominated drylands lose more than three times as much SOC as their grass counterparts. The implications of this study are twofold: (1) quantifying the effects of vegetation change on carbon loss via water erosion and the highly variable effects of land degradation on soil carbon stocks is critical. (2) If landscape-scale understanding of carbon loss by water erosion in drylands is required, studies must characterize the heterogeneity of ecosystem structure and its effects on ecosystem function across ecotones subject to vegetation change. Copyright © 2013 John Wiley & Sons, Ltd.

KEY WORDS carbon; soil carbon; soil erosion; run-off; vegetation change; ecotone

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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. Although 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 run-off 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 have the potential to alter the carbon balance of these systems, which may potentially have global-scale biogeochemical feedbacks.

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 upsampling or extrapolation (see, e.g. Schuman et al. 2002). As there may be broad controls on SOC storage, such as changing vegetation type that 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 a 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 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 are not quantified. Furthermore, Parsons et al. (2006a, b) 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 south-western USA that have experienced the encroachment of woody species over large areas (Buffington and Herbel, 1965). Although 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 south-western USA experience high-intensity monsoon rainfall régimes and high rates of overland flow and erosion that may lead to irreversible land degradation (Turnbull et al., 2008b). In these areas, the spatial heterogeneity of resources increases as woody shrubs encroach into grasslands forming islands of fertility around the woody shrubs (Schlesinger et al., 1990; Kieft et al., 1998). Such resource islands may retain carbon, for example, more effectively than inter-shrub areas (Tongway and Ludwig, 1990; Ludwig et al., 2000), but little is known about the overall net effect of degradation and vegetation change on net carbon storage (loss or gain). To understand how shrub encroachment affects the retention and loss of SOC, we first need to quantify how shrub encroachment affects the spatial distribution of SOC and how shrub encroachment alters the magnitude and the spatial distribution of eroding areas.

Turnbull et al. (2010a) studied a continuum of vegetation change across a shrub-encroachment gradient in the northern Chihuahuan desert in New Mexico. Sites were established to quantify rainfall/run-off and soil erosion fluxes from a pristine black grama (Bouteloua eriopoda) grassland site, through two intermediate sites with both black grama grass and creosotebush shrub (Larrea tridentata), to a creosotebush-dominated shrubland site. Results from this study showed that as grass cover decreases and shrub cover increases, the relationship between rainfall and run-off becomes more directly coupled. Higher run-off coefficients occur on shrubland than grassland, demonstrating that the loss of water from semi-arid landscapes may be increased as the land becomes progressively more degraded. Turnbull et al. (2010a) also reported that the total amount of erosion increased as run-off coefficients increased and that the total erosion increased across the transition from grass to shrub-covered plots. Grass-dominated plots yielded a significantly lower mass of sediment per unit of run-off than the shrub-dominated plots. These results suggest that losses of SOC may also increase as a result of shrub encroachment into grassland. Thus, SOC loss from hillslopes may be highly variable across the landscape, depending on the level of shrub encroachment and land degradation, evidenced by increases in soil erosion.

**Aim and objectives**

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

The objectives of the paper are as follows:

1. To assess the storage of SOC across a transition from pristine grassland to heavily degraded shrubland;
2. To analyse the changes in spatial variability and type of SOC across the vegetation transition;
3. To quantify the redistribution of SOC across the ecotone via the abiotic processes of rainfall, run-off and erosion.

**METHODS**

**Site description**

The Sevilleta National Wildlife Refuge (SNWR) has been designated a wildlife refuge since 1973 and was established as a Long-Term Ecological Research site in 1988 (Gosz, 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 approximately 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 that 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 south-western USA in the last 200 years (e.g. Bunting 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 $\delta^{13}$ carbon signature of SOC measured at stages over this grass–shrub ecotone supports the interpretation that creosotebush (C3 vegetation) occupies soils that were once dominated by C4 grasses (Turnbull et al., 2008a).

Experimental design

Four sites were set up over the shrub-encroachment gradient. These sites (Figure 2) 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 $10 \times 30$ m plot was built to measure run-off, erosion and nutrient export, as described in Turnbull et al. (2010a). Characterization plots were also set up, comprising two $5 \times 30$ 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 I). Measurement of soil characteristics and run-off monitoring took place during two monsoon rainy seasons (2005 and 2006), during which 17 high-intensity rainfall/run-off events were measured across all four sites.

Assessing storage of soil organic carbon across the transition

To address the first objective, 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 SOC and the $\delta^{13}$ 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...
different surface cover types (plots 1 and 4) and were used to test for significant differences in soil characteristics between different surface cover types at each site. Independent sample t-tests were used. Independent sample t-tests were used to test for significant differences between two different surface cover types (plots 1 and 4) and Tamhane’s t-test for significant differences between three different surface cover types (plots 2 and 3).

To explore the differences in SOC storage between each plot and beneath different surface cover within each plot, the values of SOC from the soil taken for bulk density analysis were calculated. The area of each plot under each cover type was then multiplied by the average of these SOC values for each cover type. Total SOC storage was then calculated for each plot to a depth of 0.05 m.

### Analysing the changes in the spatial variability of soil organic carbon across the vegetation transition

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

### Quantifying the redistribution of organic carbon across the ecotone

To address the third objective, for each rainfall event at each plot, all overland flows were routed to a large stock tank, which captured the total eroded sediment leaving each plot. Seventeen run-off events were monitored, between four and six from each plot. During these events, all overland flow, sediment and carbon were captured at the plot outlet. The total mass of sediment was recorded for each event. Sediment was sieved to less than 2 mm, riffled and then sub-sampled to analyse for the total organic carbon content of eroded sediment. The total organic carbon content of the eroded sediment was analysed as for the soils described earlier. The experimental design of the flow-monitoring sites is described in detail in Turnbull et al. (2010a).

### RESULTS

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

The mean SOC values for each plot weighted by vegetation cover do not vary significantly ($p < 0.05$),
ranging from 2.54 to 3.28 mg cm\(^{-2}\) (Table II). The grass/shrub plot contains the highest cover-weighted average SOC, whereas the shrub/grass plot contains the lowest average SOC value. The organic carbon content of the bare soil (Table II) is similar across all plots, varying from 2.34 (±0.20) mg cm\(^{-2}\) on the grass plot to 2.29 (±0.19) mg cm\(^{-2}\) on the shrub plot and showing no significant difference at \(p < 0.05\). However, the organic carbon content of the soil underlying the shrubs (on plot 2 = 6.72 ± 0.59 mg cm\(^{-2}\), plot 3 = 6.09 ± 0.64 mg cm\(^{-2}\) and plot 4 = 5.81 ± 0.41 mg cm\(^{-2}\)) was significantly greater than the organic carbon content of the bare soil and the soil underlying the grass cover (\(p < 0.05\). These results suggest that there is more organic carbon stored beneath shrubs than in inter-shrub areas, whether the inter-shrub areas are covered in grass or are devoid of vegetation.

The isotopic signature of the soils, which underlie shrubs, was always lower than the signature for soils beneath grass cover. \(\delta^{13}C\) values for soils beneath shrubs ranged from −20.94 ± 0.28% on the shrub-grass plot to −22.76 ± 0.18% on the shrub plot, whereas \(\delta^{13}C\) values for soils beneath grasses ranged from −18.58 ± 0.30% on the grass-grass plot to −19.43 ± 0.24% on the grass-shrub plot. These differences, between shrub and grass soils, were significant at \(p < 0.05\).

The bare soils showed similar isotopic signatures on all plots, although the \(\delta^{13}C\) values were lowest −18.58 ± 0.13% on the grass plot and highest −21.51 ± 0.19% on the shrub plot. The intermediate plots showed a mixed signal, particularly in the bare soils, with values of 19.65 ± 0.16% for bare soils on the grass/shrub plot and 18.46 ± 0.27% on the shrub/grass plot, illustrating the potential for \(C_3\) carbon to be mixed with \(C_4\) carbon even in the bare soils between vegetation patches.

Data presented in Table III describe the total storage of SOC in each plot and the area-weighted storage according to vegetation type. These data suggest that the total SOC within the surface soils is greatest on the grass/shrub plot and least on the grass/grass plot, whereas SOC storage is higher on the shrub plot than the grass plot. There is no significant difference between these SOC totals at \(p < 0.05\), and therefore, there is no significant trend in SOC storage observed across the transition from grass to shrub.

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

**Spatial variability of soil organic carbon across the vegetation transition**

Results of the spatial analysis of SOC across the ecotone illustrate significant differences in the degree of

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

Data used in these analyses had outliers removed (see Turnbull et al. (2011) for details). SOC, soil organic carbon; SE, standard error.

The high range of 7.1 from the grass plot $\delta^{13}$C values highlights the fact that the maximum scale of spatial variation in the grassland is significantly higher than on any of the mixed vegetation plots. Such a result suggests that the SOC values in the grassland are more homogeneous than where mixed vegetation species are evident in the other plots. Clustering of similar $\delta^{13}$C values in the shrub-dominated plots is more evident, suggesting a more heterogeneous distribution of SOC where shrubs are present.

Quantifying the redistribution of organic carbon across the ecotone

Run-off and carbon loss. Figure 3 illustrates the relationship between the total run-off leaving each plot (in litres) and the total loss of organic carbon (in grams) for the 17 events monitored during the 2005 and 2006 monsoon seasons.

Analysis of the relationship between the total event run-off and the total carbon loss for each plot shows an increasing loss of TOC with increasing flow from each plot. These linear relationships are significant on all plots (grass: $r^2 = 0.84$, $p = 0.029$; grass-shrub: $r^2 = 0.8$, $p = 0.246$; shrub-grass: $r^2 = 0.99$, $p = 0.005$; shrub: $r^2 = 0.77$, $p = 0.079$). The slope of these relationships increases with increasing level of shrub encroachment: grass = 0.7%, grass-shrub = 1.6%, shrub-grass = 1.8% and shrub = 7.4%.

Results show that the total losses of carbon are greater on the shrub-dominated plots and also that the mass of carbon per unit volume of run-off is higher when shrub vegetation dominates over grass. Loads from all events recorded on the shrub and shrub grass plots range from 0.013 to 0.094 g C l$^{-1}$, whereas loads from the grass and grass-shrub plots range from 0.004 to 0.022 g C l$^{-1}$ and are significantly different (t = 2.51, $p < 0.05$) using a heteroscedastic t-test, assuming unequal variance. In addition, the total yield of carbon from the shrub plots may be as much as six times higher (based on the largest flow events recorded) than the yield from the grass plots. This results in a total yield of carbon (over all events; 10 from the grass plots and 7 from the shrub plots) from the shrub plots of 527 g (17.6 kg ha$^{-1}$) compared with 147 g (4.9 kg ha$^{-1}$) from the grass plots.

It is also instructive to look at the influence of the run-off coefficient, which describes the proportion of rainfall

<table>
<thead>
<tr>
<th>Plot</th>
<th>Surface cover</th>
<th>SOC storage (kg)</th>
<th>SOC storage (g m$^{-2}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Grass</td>
<td>Bare</td>
<td>19.13 ± 1.65</td>
<td>117.00 ± 10.00</td>
</tr>
<tr>
<td>Grass</td>
<td>Grass</td>
<td>22.85 ± 1.43</td>
<td>167.50 ± 10.50</td>
</tr>
<tr>
<td>Grass</td>
<td>Total</td>
<td>41.99 ± 1.54</td>
<td>139.98 ± 10.09</td>
</tr>
<tr>
<td>Shrub</td>
<td>Bare</td>
<td>23.26 ± 2.31</td>
<td>136.00 ± 13.50</td>
</tr>
<tr>
<td>Shrub</td>
<td>Grass</td>
<td>21.54 ± 1.91</td>
<td>186.00 ± 16.50</td>
</tr>
<tr>
<td>Shrub</td>
<td>Shrub</td>
<td>4.44 ± 0.39</td>
<td>336.00 ± 29.50</td>
</tr>
<tr>
<td>Shrub</td>
<td>Total</td>
<td>49.23 ± 1.15</td>
<td>164.10 ± 14.84</td>
</tr>
<tr>
<td>Shrub</td>
<td>Bare</td>
<td>18.60 ± 1.55</td>
<td>84.00 ± 7.00</td>
</tr>
<tr>
<td>Shrub</td>
<td>Grass</td>
<td>8.64 ± 0.84</td>
<td>201.50 ± 19.50</td>
</tr>
<tr>
<td>Shrub</td>
<td>Shrub</td>
<td>10.87 ± 1.14</td>
<td>304.50 ± 32.00</td>
</tr>
<tr>
<td>Shrub</td>
<td>Total</td>
<td>38.11 ± 1.18</td>
<td>127.04 ± 12.60</td>
</tr>
<tr>
<td>Shrub</td>
<td>Bare</td>
<td>26.35 ± 2.19</td>
<td>114.50 ± 9.50</td>
</tr>
<tr>
<td>Shrub</td>
<td>Shrub</td>
<td>20.31 ± 1.43</td>
<td>290.50 ± 20.50</td>
</tr>
<tr>
<td>Shrub</td>
<td>Total</td>
<td>46.65 ± 1.81</td>
<td>155.51 ± 11.52</td>
</tr>
</tbody>
</table>

Table III. Total estimated SOC storage within each plot for soils beneath each type of surface cover to a depth of 5 cm

Soil organic carbon (SOC) storage (in kilograms) 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 (in grams per squared meter) for each surface cover illustrates the 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.
that left each plot as overland flow, on the loss of carbon from the landscape. To address the third objective, we explore whether increasing run-off coefficients will lead to greater losses of carbon. As the same rainfall event falling on plots with different vegetation cover would be expected to generate different run-off coefficients (Turnbull et al. 2010a), the run-off coefficient may describe the effect that the changing structure of vegetation (e.g. higher shrub canopies and denser foliage can intercept more rainfall than grasses) can have on the mobilization of carbon. Figure 4 illustrates the relationship between the run-off coefficient and the total organic carbon loss (in grams) during each rainfall/run-off event from all plots.

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 run-off, across the plots, indicated by the exponent of 1.53. Thus, the process of carbon loss becomes more concentrated for a given unit of run-off, across the transition from grass to shrub-dominated landscape. In addition, high run-off coefficients 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 as follows: (1) run-off coefficients increase, (2) total carbon loss increases at a faster rate than increases in run-off coefficient and (3) as run-off 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 except 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, although the two pairs exhibit different behaviour when compared with each other. The total mass of organic carbon per unit of sediment lost from each plot varies from 0.011 to 0.032 g C g\(^{-1}\) with a mean of 0.019 g C g\(^{-1}\) on the shrub plots, compared with 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 approximately three times more carbon per unit of eroded sediment than their grass-dominated counterparts.

**DISCUSSION**

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

**Total soil organic carbon.** The greater organic carbon content of the soil underlying shrubs, when compared with that under grasses or bare soil, suggests that the 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).

![Figure 4](image-url)  
Figure 4. Total event run-off coefficient and total mass of carbon lost per event (in grams) from each plot. Solid arrow represents the general trend of both increasing run-off and carbon loss across the transition from grass-dominated to shrub-dominated landscape

![Figure 5](image-url)  
Figure 5. Total eroded sediment (in grams) and total organic carbon loss (in grams) 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.
Such increasing heterogeneity is an important part of land degradation, where inter-shrub areas become depleted of resources, whereas 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 the 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 researchers have shown that SOC can have decadal turnover (Townsend et al. 1996) well within the 200 years that is postulated for the woody shrub encroachment of the semi-arid grasslands studied here (Bufflington 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.

$C_3$ versus $C_4$ soil organic carbon. To address the second objective, the isotopic values of SOC were quantified, in order to estimate the spatial variability of different types of carbon. The $\delta^{13}C$ for the soils underlying the shrubs were closer to those reported in Turnbull et al. (2008a) for the *L. tridentata* shrub end-member (approximately $-26\%$) than for the *B. eriopoda* end-member, which has an isotopic signature of $-14.0\%$. The soil carbon in the region of the shrubs is dominated by $C_3$ carbon derived from the woody vegetation and retained beneath these shrubs. Conversely, the $\delta^{13}C$ values beneath grass and in bare areas were nearer to the *B. eriopoda* grass end-member values than the *L. tridentata* shrub end-member. This finding demonstrates the mixture of SOC that is found, particularly in the bare soils, which may previously have been covered by $C_4$ grasses but which may now lie beneath concentrated flow paths between $C_3$ woody shrubs. Liao et al. (2007) reported complementary findings from the Rio Grande Plains of southern Texas, USA, where $C_3$ trees and shrubs have encroached $C_4$ grasslands since the western migration of European settlers. The authors demonstrate that carbon stocks increase in close proximity to woody vegetation due to the retention of older $C_4$ carbon and the capture of new $C_3$ carbon beneath the plant canopy. Similarly, Kieft et al. (1998) found that SOC beneath *L. tridentata* was consistently higher than the SOC in grassland soils and also that soils beneath *L. 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 $C_4$ 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 $C_3$ carbon both due to the deposition of leaf litter and also due to the transport of carbon-rich material beneath the shrub canopy by rain splash 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 analysing 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 the spatial distribution and variability of soil organic carbon

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. (2008) 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 flow paths between shrubs. We suggest that the abiotic (hydrological) controls of run-off and erosion – processes that lead to the redistribution of SOC 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 recognized 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 SOC 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, SOC 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 run-off generation, which in turn reinforces the spatial distribution of SOC in a similar pattern to the run-off and erosion, i.e. depletion of SOC from the bare soils, and 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 the elevated levels of $C_3$ SOC in the areas beneath shrubs, in excess of the areas beneath grass and in 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 shrubland, 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 positive feedback of resource distribution, where resources will be concentrated, leading to the depletion of resources in inter-shrub areas.

Turnbull et al. (2011) observed similar dynamics between overland flow and the loss of total nitrogen (N) and total phosphorus (P), across the same grass to shrub transition as reported here, suggesting that the progressive degradation of grasslands in semi-arid areas is likely to result in significant alterations to macronutrient cycles that are primary controls on vegetation growth. Working in a similar, semi-arid environment, Barger et al. (2006) found that the vast majority of both total carbon and total nitrogen yields were correlated with soil erosion losses. It is likely therefore that all of these nutrients are either bound to sediment particles or behave very similarly, in the form of plant litter that is mobilized and transported by overland flow. Thus, soil erosion, without intervention, will continue to deplete carbon resources in semi-arid soils.

The proportion of rainfall that leaves each plot as overland flow also increases across the transition from grass to shrubland. Parsons et al. (1996) studied two end-member (grass and shrub) plots in south-east Arizona and demonstrated that the interrill areas of shrub-dominated hillslopes yield both higher run-off rates and higher run-off velocities than their grass-dominated counterparts. From our data, concurrent with this increase in overland flow, there appears to be an increase in the carbon leaving the system, with shrub plots yielding up to three times as much carbon as grass plots. Clearly, more energy is available via concentrated hydrological flow paths in shrub-dominated landscapes, as geomorphic features, such as gullies or rills, may develop (Cerdà 1997; Reid et al. 1999; Parsons et al., 1996; Parsons and Wainwright 2006), so it is likely that more SOC is entrained as these flow paths become more connected in a shrub-dominated landscape than in a grass-dominated landscape.

Data presented here demonstrate a strong relationship between soil erosion and SOC loss, as the former increases, so does the latter at all stages over the shrub-encroachment gradient, but to a greater extent when shrubs are the dominant vegetation type. This finding shows that even during the intermediate stage of transition, where there is still grass present in inter-shrub areas (here, approximately 14% of the soil in the shrub-grass plot is covered in grass), there is still a similar yield of carbon per unit of soil erosion as on the shrub plot, where grass vegetation is no longer present. Other researchers have shown variability in rates of above-ground carbon stores under single vegetation types such as Prosopis glandulosa (honey mesquite) (Asner et al. 2003); in savanna grassland systems subject to vegetation change, through long-term monitoring (Buffington and Herbel, 1965); or through SOC isotope analysis of woody shrub invasion of grasslands (Boutton et al. 1998). However, these researchers did not observe the changes to carbon storage that were brought about by erosion processes in interaction with vegetation change. What is shown here is that both the stores and spatial distributions of carbon change across the shrub-encroachment gradient, and the rates of carbon loss also change, a finding, which underlines the significance of water erosion on the loss of carbon from dryland environments.

CONCLUSIONS

A robust estimation of the changing carbon storage in semi-arid soils subject to vegetation encroachment is presented. The first objective showed that more SOC is stored in soils beneath shrubs than grasses and that soils beneath both vegetation types store more SOC than bare soils in inter-plant areas. Results also suggested that although there are differences in the area-weighted SOC storage across the grass to shrub transition, these differences are not consistent over the shrub-encroachment gradient. The total amount of SOC stored beneath the grasses varied significantly across the transition, increasing in concentration as the grass cover became more sparse. These data illustrate that the surviving grasses in the landscape, perhaps being better established and thus resilient to the processes of erosion, are able to retain more SOC than their counterparts in the pristine grass plot. Similar results are seen for shrubs, where progressively more SOC is found in storage beneath the shrub canopies, as the shrubs become more established in the landscape. Furthermore, because of the elevated topography of the shrub mounds, run-off typically diverges into inter-shrub
areas increasing erosion while reducing erosion from beneath the shrubs. The consequent capture of resources beneath shrubs and its accentuation in parallel with increasing erosion in inter-shrub areas exemplifies the role that abiotic processes play in redistributing SOC around the landscape and enhancing the heterogeneity of carbon storage within the soil.

Results of this monitoring work also described the type of SOC that was present across the transition, in terms of its isotopic signature, in order to determine whether the source of the carbon stored in the soil (C₃ or C₄ plant derived carbon) differed across the vegetation transition. Soil beneath vegetation tends to be dominated by the carbon that has been derived from that vegetation type: C₃ carbon beneath shrubs and C₄ carbon beneath grasses. The bare areas of soil are characterized by an intermediate carbon signal, reflecting the mixture of older grass-derived carbon and younger inputs of shrub-derived carbon from surrounding vegetation.

The second objective quantified the spatial variability of SOC across the vegetation transition. Results showed that the heterogeneity of SOC increases from a relatively consistent distribution of SOC under grasslands to a highly variable or patchy distribution under shrubs. It is clear that shrub species afford good local protection to the soil against intense rainfall (via more dense canopy cover) so they may conserve SOC by reducing the energy that is available to mobilize sediment and carbon via erosion. Shrub species may also be more efficient at supplying carbon to the soil. Shrubs are better able to capture soil moisture (soil moisture levels are consistently lower beneath shrubs than elsewhere in the ecotone) and convert this to C₃ carbon via the annual deposition of leaf litter. Although dieback of grasses will contribute some C₄ carbon to the SOC pool, it appears to be significantly less than the C₃ carbon contribution.

The third objective quantified the redistribution of SOC across the ecotone due to water erosion processes. Results suggested that greater losses of soil and carbon will occur as the ecosystem degrades and shrubs become better established than grasses, with a concomitant increase in the amount of bare soil that is present. Landscapes that are covered by shrub species such as _L. tridentata_ are therefore not only more susceptible to erosion but will also lose more carbon per unit of soil loss than their grassland counterparts. In part, this difference is driven by the positive feedback that is established as a landscape degrades, whereby bare areas of soil become progressively more eroded and depleted of nutrients, which makes them less viable locations for vegetation to re-establish. In addition, the difference is controlled by the increased spatial heterogeneity that is associated with woody shrub encroachment, when compared with the more homogeneous distribution of resources that are found under grass species such as _B. eriopoda_.

The implications of land degradation on both contemporary and future rates of carbon loss from drylands are shown to be important, especially when woody shrubs encroach pristine grasslands. Across a grass–shrub ecotone, we show that changes in the ecosystem structure, or pattern of carbon storage and its function, particularly the redistribution of carbon by overland flow will lead to significant increases in the loss of SOC from the system as woody shrubs become more dominant.

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**REFERENCES**


