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Selective erosion of clay, organic carbon and total nitrogen in grazed semiarid rangelands of northeastern Patagonia, Argentina

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ABSTRACT

In arid and semiarid rangelands, soil erosion has been widely considered an important soil degradation process and one of the main factors responsible for declining soil fertility. In this study, we determined the sediment production and the enrichment ratios of clay, organic C, and total N by using rainfall simulations on runoff plots (0.60 × 1.67 m) in three plant communities of northeastern Patagonia: grass (GS), degraded grass with scattered shrubs (DGS), and degraded shrub steppes (DSS). Our results clearly indicate that spatial variability in soil loss rate and enrichment process exists as a result of the local differences in both plant composition and soil surface characteristics. Sediment production was significantly lower in the GS (14.2 g m^{-2}) compared with the DGS and DSS (38.2 and 51.5 g m^{-2}, respectively). In the GS, the enrichment ratio of clay was significantly greater (3.9) and enrichment ratio of organic C was lower (3.1) than in the DGS and the DSS, though differences in enrichment ratios of total N were not significant. The high rate of soil loss and nutrients through overland-flow may limit the opportunities that promote the pathway from DGS back to GS community, favoring the dominance of shrubs.

1. Introduction

In arid and semiarid rangelands, soil erosion has been widely recognized an important soil degradation process and is considered one of the main factors responsible for declining soil fertility potential and desertification (Bestelmeyer et al., 2006; Michaelides et al., 2009; Schlesinger et al., 1990). In fact, soil erosion not only promotes the loss of fine-size soil particles and organic matter, which play an essential role in aggregate stability (Mbagwu et al., 1994), but also removes plant nutrients in solution or as part of solid components (Girmay et al., 2009; Palis et al., 1997, 1990). Therefore, sediments transported from interrill areas are generally enriched with clay, organic matter and nutrient elements as compared with the soils from which they are derived (Alberts et al., 1983, 1980; Bertol et al., 2007). According to Di Stefano and Ferro (2002), after splashing of the soil aggregates by raindrop impacts, physical selectivity occurs within the erosion process both because the energy of the interrill flow is not sufficient to transport many of the soil particles, mainly sand, silt and soil aggregates, and because the larger particles and aggregates are preferentially deposited during transport. Litter lying on the soil surface is also easily transported by interrill flow.

The selectivity of the erosion processes is often expressed as an enrichment ratio, ER. Massey and Jackson (1952) calculated the ER as the ratio of the soil component content of sediments (eroded soil) to that of source soil. The importance of analysis of sediment transported by runoff to quantify the complex effect of erosion was already known in the last half century (Alberts et al., 1983, 1980; Barrow and Kilmer, 1963; Lal, 1976; Sharpley, 1985) and considered in detail in recent decades (e.g., Bertol et al., 2007; Di Stefano and Ferro, 2002; Girmay et al., 2009; Palis et al., 1997, 1990; Schiettecatte et al., 2008; Schlesinger et al., 2000).

Although during the last decades some efforts have been made to evaluate the soil erosion rates and its effect on the spatial pattern of vegetation (e.g., Chartier and Rostagno, 2006; Parizek et al., 2002; Pierson et al., 1994), many authors argued that we are yet far from a complete understanding of the implications of arid land degradation (e.g., Lal, 2001; Ludwig et al., 2005; Peters et al., 2006). In this sense, the loss of soil fertility in rangeland degradation processes has recently been identified as one of the main issues in rangeland ecology and management (Briske et al., 2005; SRM Task Group, 1995). The relative importance of erosion and soil fertility loss is hypothesized to play a central role in resource redistribution and vegetation patch structure (Bestelmeyer et al., 2006; Briske et al., 2008; Schlesinger et al., 1990; van de Koppel et al., 2002).

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Thus, the changes in various ecological processes (e.g., soil erosion, nutrient cycling, and productivity) may produce a complex pattern of alternative states, typically observed in intensively grazed rangelands.

In this context, state-and-transition models (Westoby et al., 1989) hold great potential to aid in understanding rangeland ecosystems' response to natural and/or management-induced disturbances by providing a framework for organizing current understanding of potential ecosystem dynamics. In essence, the measurements of soil erosion processes are consistent with the idea that soil erosion can cause and reinforce transitions between states. Such statements about soil processes are typically assumed in such models but the use of process-based experiments to define the attributes distinguishing states have seldom been attempted and infrequently directly measured (Bestelmeyer et al., 2003). Investigations of these pattern-process linkages could provide insights to improve monitoring and management that may not arise from simpler, current state-and-transition models (Bestelmeyer et al., 2011; Chartier et al., 2011; de Soyza et al., 2000).

Understanding the potential of soil fertility loss and its hazards associated with rangeland degradation is essential to effectively address landscape change resulting from climate and land use change (Ludwig et al., 2005; Nearing et al., 2011; Newman et al., 2006). The objectives of this research were: (i) to determine the sediment production and the enrichment ratios of clay, organic C, and total N in transported sediment under simulated rainfall in three plant communities spatially related along a degradation gradient; and (ii) to investigate possible factors contributing to change in enrichment ratios of these soil components.

2. Material and methods

2.1. Study area

The study area is located in northeastern Patagonia, Argentina (43°00′S and 64°36′W). In this region the climate is arid and temperate. Mean annual temperature is 12.5 °C (Barros, 1983) and the average precipitation is 258 mm (1995–2004) (Chartier and Rostagno, 2006).

In the area, we selected two contiguous ecological sites: a pediment-like plateau and a flank pediment. The ecological site term is referred to areas that differed in landform, although soil type, vegetation composition and grazing management were similar. In this context, Beeskow et al. (1987) described a pediment-like plateau as an erosional surface of low relief covered by alluvium, whereas flank pediments (as described by Fidalgo and Riggi, 1970) are short slope transport surfaces, generally developed between a plateau and a lower zone, with a base level controlled by a playa lake. The dominant soil is a Xeric Calciargid with Xeric Haplocalcid as subdominant (Soil Survey Staff, 1999). The Xeric Calciargid is shallow with loamy sand A horizon 10–20 cm thick, a sandy loam to sandy clay loam Bt horizon 10–15 cm thick, and a calcic horizon Bk 20–30 cm thick. The gravel content in the A horizon varies between 10 and 15%.

Vegetation cover varies from 40 to 60% and presents a patchy structure where three discrete plant associations or community units, as described by Whittaker (1975), are clearly recognizable: i) a grass steppe with scattered shrubs (GS), dominated by *Nassella tenuis* (Phil.) Barkworth and *Piptochaetium napostaense* (Speg.) Hack, ii) a degraded grass steppe with scattered shrubs (DGS), and iii) a degraded shrub steppe (DSS), dominated by *Chuquiraga avellanedae* Lorentz. These communities were identified along a gradient of grazing intensity (Beeskow et al., 1995) and correspond to three states or stages of range degradation (Chartier and Rostagno, 2006; Chartier et al., 2011). Grass dominated steppe represents the most desirable state in terms of livestock production, while shrub steppe represents the most degraded and least productive state. Sheep grazing for wool production is the main use of these rangelands where continuous grazing is practiced extensively at moderate to heavy intensity (0.3 sheep ha^{-1}).

2.2. Experimental procedure

For the field experiments, during the years 2003 and 2004, we randomly selected homogeneous vegetation patches at both ecological sites: eight in GS, fourteen in DGS, and eight in DSS. The combination of three different plant communities and two ecological sites (pediment-like plateau versus flank pediment) resulted in a total of 60 runoff experimental plots. In the DGS the number of patches was increased due to the greatest surface occupied by this community respect to the others. Inside each selected patch, experimental plots measuring 0.60×1.67 m (1 m^{-2}) were randomly located in the shrub interspaces of the different plant communities, where the erosion risk is maximum. The slope of the plots was homogeneous with an average of 4%.

Simulated rainfall was applied with a full cone, single nozzle rainfall simulator (Rostagno and Garayzar, 1995) at an intensity of 110 mm h^{-1} during 30 min. Runoff from each plot was collected and recorded by volume. The sediment production was obtained by decantation (72 h) and determined by drying (at 60 °C for 48 h). The dispersed particle-size analysis was obtained using the sieve pipette method (Gee and Bauder, 1986). The sediment samples were analyzed for organic C by the Walkley–Black method (Nelson and Sommers, 1982), and total N following the Kjeldahl procedure (Bremner and Mulvaney, 1982).

2.3. Field sampling

Prior to simulated rainfall application, runoff plots were sampled along three 1.67 m equidistant, parallel transects. Distances between consecutive intercepted plants of perennial grasses were recorded along each transect. Ground (perennial grass, litter, and gravel) and bare soil cover were determined by the point quadrat method using 33 points per transect (Mueller-Dombois and Ellenberg, 1974). The diameter of the largest bare soil patch in each plot was also recorded. The A horizon thickness was determined by the depth to the Bt horizon in a pit opened adjacent to each plot. Adjacent to each runoff plot, topsoil samples (0–5 cm) were collected for texture, organic C and total N content following the same procedures described above to determine the enrichment ratio of the sediment. The ER of a soil constituent was calculated by dividing the content of the constituent in the transported sediment by its content in the original soil material (Avnimelech and McHenry, 1984). When the sediments are enriched with a given component, as compared to the contributing soils, the enrichment ratio is greater than unity.

2.4. Statistical analysis

The statistical difference in the runoff (L), the sediment production (g m^{-2}), and the enrichment ratios of clay, organic C, and total N among plant communities and between ecological sites were tested by a one-way analysis of variance. We used Fisher's LSD (Sokal and Rohlf, 1981) mean separation test with a 0.05 significance level. Pearson's correlation coefficients were used to assess the linear associations between the above variables.

Furthermore, the enrichment ratios for clay, organic C, and total N as related to the appropriate concentrations in the contributing soil were examined using non-linear regression techniques.

Enrichment ratios were used as dependent variables. Significance levels were determined at $P \leq 0.05$.

3. Results

3.1. Vegetation and soil surface characteristics

The dominant plant communities, which represent the widest range of heterogeneity in the study area, are summarized in Table 1. Soil properties as well as relative proportions and spatial pattern of plant communities were similar in the two ecological sites. However, the three plant communities presented contrasting soil surface characteristics within each site. The GS had greater perennial grass and litter cover than the DGS and DSS communities. In contrast, the values of gravel and bare soil cover increased from GS to DSS. Along the degradation gradient from GS to DSS communities (see Chartier and Rostagno, 2006) a finer soil texture was recorded. A horizon thickness was higher in the GS as compared with the shrub interspaces of the DSS. Soil organic C and total N content were positively correlated ($r = 0.68$, $P < 0.001$), recording the highest mean values in the GS with respect to the DGS and DSS communities.

3.2. Soil erosion and enrichment of soil particles

Runoff and sediment production were not significantly different between ecological sites ($P = 0.99$ and 0.63, respectively), nor were enrichment ratios for clay, organic C, and total N ($P = 0.48$, 0.82, and 0.97, respectively). Accordingly, we combined data from the two ecological sites in reporting results for the study area. However, runoff and sediment production were significantly different among plant communities ($P \leq 0.0001$) showing a higher value in the DSS than in the GS and DGS (Table 2). Sediment production was negatively correlated with perennial grass and litter cover ($r = -0.66$ and -0.64, respectively) and positively correlated with gravel cover ($r = 0.64$). Enrichment ratios for clay and organic C were significantly different among plant communities ($P \leq 0.0001$ in both variables), though differences in enrichment ratios of total N were not significant ($P = 0.48$).

The enrichment ratios for clay and organic C were inversely related to the corresponding concentration in the soil ($R^2 = 0.61$ and 0.43, respectively) (Fig. 1A and B). Similar trends were observed for the total N enrichment ratios ($R^2 = 0.22$, $P < 0.001$) (Fig. 1C). Moreover, the enrichment ratios of organic C and total N were closely and positively correlated ($r = 0.55$, $P < 0.001$) whereas the correlation coefficients between these components and clay enrichment were low ($r = -0.30$, $P < 0.02$; and $r = 0.28$, $P < 0.03$, respectively).

Table 2
Average (±1 SE, $n = 60$) runoff, sediment production and enrichment ratios of clay, organic carbon and total nitrogen in the transported sediments for the grass (GS), degraded grass (DGS), and degraded shrub steppe (DSS). Different lowercase letters indicate significant ($P < 0.05$) differences among plant communities.

	Rangeland plant communities		
	GS	DGS	DSS
Runoff generation (L)	19.6 ± 1.4a	28.8 ± 1.2b	38.7 ± 0.9c
Sediment production (g m^{-2})	14.2 ± 3.6a	38.2 ± 3.5b	51.5 ± 4.4c
ER Clay	3.9 ± 0.4a	2.4 ± 0.2b	1.5 ± 0.3b
ER Organic C	3.1 ± 0.6a	6.7 ± 0.5b	7.1 ± 0.6b
ER Total N	3.6 ± 0.5a	4.6 ± 0.3a	4.5 ± 0.4a

4. Discussion

4.1. Vegetation and soil surface characteristics

In the Punta Ninfas rangelands, a mosaic of patches of three plant communities were found co-occurring in the study area: grass with scattered shrubs (GS), degraded grass with scattered shrubs (DGS), and degraded shrub steppes (DSS). Pediment-like plateau and flank pediment sites presented similar proportions and spatial pattern of plant communities, showing that the stages of degradation apply equally to both ecological sites, with contrasting differences in soil surface characteristics in the three plant communities (GS, DGS and DSS) within each site (Table 1). Thus, while in the shrub interspaces of the GS and DGS communities the ground cover is dominated by perennial grasses and litter, in the DSS community these soil protection factors are in part replaced by gravels. Indeed, in severely degraded areas, gravels become the dominant cover, commonly developing a desert pavement (Rostagno and Degorgue, 2011) with pedestalled grass plants and gravels, vesicular crust, a high percentage of bare ground and exposed roots of woody plants. Alongside decreases in grass cover and a greater distance between perennial grass plants, the cover of bare ground and the size of the bare soil patches showed a marked increase in DGS as compared with GS, but DGS did not differ from DSS in these measurements. In this community, the greater gravel cover reduced the size of the bare soil patches. According to the current interpretation (Chartier and Rostagno, 2006; Parizek et al., 2002; Rostagno, 1989), differences in surface soil properties among

Table 1
Surface characteristics and soil physical and chemical properties (mean ± SE values, $n = 60$) for the different plant communities, grass (GS), degraded grass (DGS) and degraded shrub steppe (DSS), in the two ecological sites.

	Pediment-like plateau			Flank pediment		
	GS	DGS	DSS	GS	DGS	DSS
Perennial grass cover (%)	36.7 ± 5.1	17.8 ± 2.5	9.0 ± 3.2	52.9 ± 5.7	18.3 ± 1.8	5.6 ± 1.2
Litter cover (%)	53.2 ± 5.1	24.6 ± 2.4	10.5 ± 3.6	29.4 ± 4.9	25.0 ± 4.6	3.6 ± 0.9
Gravel cover (%)	0.6 ± 0.08	3.7 ± 1.5	48.8 ± 8.9	0.5 ± 0.2	7.6 ± 2.7	44.7 ± 3.0
Bare soil cover (%)	9.4 ± 1.9	53.9 ± 4.2	31.7 ± 4.3	17.2 ± 3.0	49.2 ± 4.2	46.1 ± 2.7
Distance between perennial grass plants (cm)	9.2 ± 0.7	19.1 ± 1.8	71.9 ± 9.8	9.6 ± 0.7	20.7 ± 1.7	88.5 ± 10.0
Diameter of the largest bare soil patch (cm)	6.0 ± 0.6	27.2 ± 3.5	28.3 ± 7.6	7.9 ± 1.9	21.1 ± 3.7	6.6 ± 1.6
A horizon thickness (cm)	35.1 ± 3.0	24.8 ± 3.1	5.6 ± 1.0	27.2 ± 2.0	14.4 ± 2.6	2.6 ± 0.5
Sand (g kg^{-1})	743.4 ± 7.0	715.2 ± 9.2	656.0 ± 32.7	661.9 ± 37.6	746.6 ± 14.4	615.9 ± 45.3
Silt (g kg^{-1})	183.3 ± 8.7	194.4 ± 5.2	191.7 ± 12.2	249.5 ± 24.5	164.0 ± 11.1	192.3 ± 17.8
Clay (g kg^{-1})	73.3 ± 8.5	90.4 ± 8.0	152.3 ± 29.8	93.6 ± 13.4	89.1 ± 8.7	191.8 ± 34.6
Organic C (g kg^{-1})	11.0 ± 1.7	7.8 ± 0.5	5.9 ± 0.4	12.0 ± 1.1	6.4 ± 0.5	6.5 ± 0.6
Total N (g kg^{-1})	1.2 ± 0.2	0.8 ± 0.05	0.6 ± 0.04	1.3 ± 0.1	0.6 ± 0.03	0.5 ± 0.03

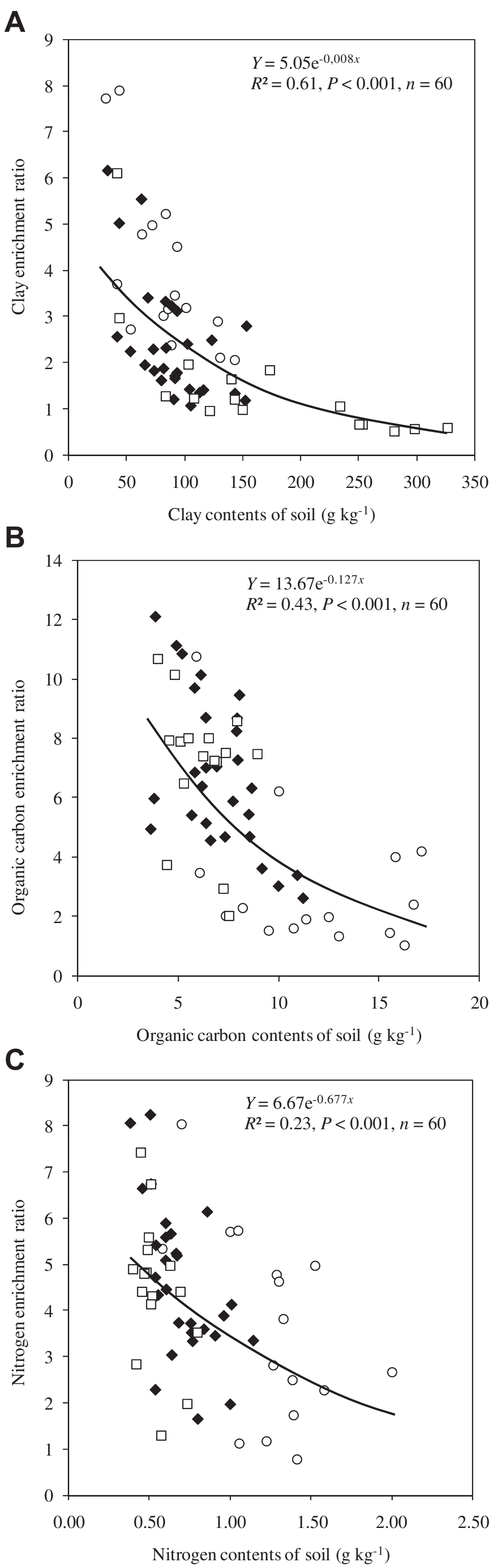

Fig. 1. Enrichment ratios for (A) clay, (B) organic carbon, and (C) total nitrogen as related to the appropriate concentration in the contributing soil. Values above unity indicate that the sediments are enriched with a given component, as compared to the original soil materials. Solid line is the exponential fit for runoff plot samples. Symbols indicate different plant communities: grass (○), degraded grass (◆), and degraded shrub steppe (□).

plant communities are mainly due to accelerated soil erosion processes.

4.2. Soil and nutrient losses by erosion

Results showed that variation in plant composition and soil surface characteristics are closely related and jointly control soil erosion loss in semiarid lands of northeastern Patagonia. Thus, the reduction in perennial grasses and litter cover in both DGS and DSS communities was followed by an increase in sediment production and runoff, as compared with the GS (Table 2). Increases in the amount of sediment removed from most degraded areas may be explained in part by the high splash detachment by raindrop impact, associated with bare soil cover (Table 1). In concordance, many studies of rangeland runoff and erosion processes suggested that the increment of sediment production would be related to an inadequate type and amount of vegetative cover to prevent accelerated soil erosion (e.g., Chartier and Rostagno, 2006; Ludwig et al., 2005; Nearing et al., 2011; SRM Task Group, 1995).

The fact that in northeastern Patagonian rangelands, soils enriched in clay content are associated with limited soil infiltration rates can contribute to explain the high runoff and erosion rates as found in degraded communities (e.g., DGS and DSS, in Table 2). In fact, clay-rich soils, mainly smectitic (Bouza et al., 2007), were locally found in the *C. avellanedae* shrub interspaces of the DSS (Table 1), where the A horizon has been largely stripped by erosion and the underlying restrictive argillic horizon exposed (Chartier and Rostagno, 2006; Rostagno, 1989). When the argillic horizon is close to the soil surface, it is generally associated with desert pavements in which vesicular layers, sedimentary crusts, and surface clay seals are prominent (Rostagno and Degorgue, 2011). The low hydraulic conductivity of argillic horizons as well as clay seals and vesicular layers are also responsible of greater runoff and sediment delivery (Chartier et al., 2011).

Moreover, in severely degraded areas, perennial grass cover decreases while increasing the distance between perennial grasses and size of the bare soil patches (Table 1). This surface condition is followed by greater interconnection among bare soil patches, which in appearance may greatly affect the natural function and structure of landscape. Recent conceptual advances in community and landscape ecology highlight the importance of spatial heterogeneity and connectivity of bare soil patches within the hillslope in determining runoff and soil erosion (Chartier and Rostagno, 2006; Davenport et al., 1998; Newman et al., 2010). Thus, runoff is concentrated downslope in the intershrub areas, increasing the velocity of the flow and exceeding the critical threshold to carry out sediments (Michaelides et al., 2009). The higher energy of the runoff will accelerate erosion in the bare soil areas. In this context, these findings support the emerging notion that, in some arid and semiarid rangeland ecosystems, the size and connectivity of the bare soil patches may be even a better indicator of an area's ability to lose soil and water resources than the average amount of vegetation cover (Bestelmeyer et al., 2006; Chartier et al., 2011; de Soyza et al., 2000; Ludwig et al., 2007).

Results from this study indicated that increase in runoff and soil erosion rates as well as size and connectivity of the bare soil patches were associated with lower clay enrichment ratio, as clearly evidenced along the soil degradation gradient from GS to DSS communities (Table 1). As has been documented in earlier studies, the soil erosion process tends to selectively remove finer particles of soil that are more vulnerable to loss than the coarser soil particles (Farenhorst and Bryan, 1995; Schiettecatte et al., 2008; Sharpley, 1980). In concordance, we found a maximum enrichment ratio of clay sized particles in the eroded materials of 3.9 for the GS community, in contrast to those of 2.4 and 1.5 for DGS and DSS communities, respectively (Table 2). This decline in enrichment of clay particles may be attributed to the much greater aggregate stability associated with clay mineral content in DSS (17.2%) as compared with the GS (8.3%, Table 1). In this community, aggregated particles may be easily broken down into smaller aggregates or clay-sized particles, and therefore transported by runoff. Mbagwu et al. (1994) carried out an experimental study on the water-stability of aggregates of surface soils and found that some of the soil properties which influence aggregate stability are the organic C contents and the concentration and nature of the constituent aggregate-stabilizing agents (such as amount of clay, polyvalent cations, oxides of iron and aluminum). Moreover, the physical selectivity of these processes can be variable depending on the interrill or rill erosion process (Di Stefano and Ferro, 2002). Interrill flow does not have sufficient energy to transport coarse particles or soil aggregates and then they are preferentially deposited during transport. On the contrary, coarser enrichment sediments are usually associated with concentrated-flow erosion, because it is less disruptive (Alberts et al., 1983).

Regarding organic C losses, the enrichment ratios were higher than one, recording the highest values in the DGS and DSS compared with the GS (Table 2). Field observations revealed that in the shrub dominated plots, litter and seeds were the first soil components removed by overland flow because of its low specific weight and its concentration on the soil surface. However, though a high proportion of organic C in the sediments was plant residues, enrichment of organic C was inversely proportional to grass and litter cover (Table 1). Thus, while in the GS community the percentage of grass and litter reached 86.1% of total ground cover, in the DSS the ratio decreased to 14.3%. The high grass plant density found in the GS community, as indicated by the short distance between perennial grasses (Table 1), usually confers an interwoven made by crisscrossing strips of leaves and litter creating barriers or obstacles that hold back the transport of suspended litter materials in runoff flow (Chartier and Rostagno, 2006). In contrast, the highest sediment removal was recorded in the DSS where litter is easily removed by surface runoff, increasing the organic C concentration in the eroded sediments.

Our results showed that N is also being disproportionally lost from each plant community, however, mean differences in enrichment ratio of total N were not significant (Table 2). According to Gallardo and Schlesinger (1995), in the degraded areas of southern New Mexico, where semiarid grasslands have been invaded by shrubs, the proportional loss of organic C exceeds that for soil N, so that soil C/N ratios decreases and C becomes limiting for microbial biomass as desertification proceeds. As precedent studies confirm, small decreases in soil C and N, besides other soil physical and chemical properties, indicate the impact of soil degradation on soil quality (Alberts et al., 1980; Barrow and Kilmer, 1963; Girmay et al., 2009). However, in arid and semiarid environments, an accurate measurement of the rate of total N enrichment from runoff field plots may be difficult to obtain because the complex interactions of different enrichment processes and the larger within-field variability of N content in soil surface.

This study revealed that total N and organic C enrichment ratios were highly correlated, as has been reported in rainfall simulation experiments for other soils elsewhere (e.g., Avnimelech and McHenry, 1984; Palis et al., 1997). Indeed, the data reported here showed a strong, negative relationship between the organic C and total N enrichment ratios and the corresponding nutrient concentration in the 0–5 cm soil layer (Fig. 1B and C). Thus, the variation in total N and organic C losses across rangeland plant communities is also consistent with previous studies in other arid ecosystems showing a decrease in soil quality as a result of land degradation due to heavy grazing disturbance (Bisigato et al., 2008 and cites therein), and with studies demonstrating loss of nutrients as a consequence of soil erosion (Girmay et al., 2009; Palis et al., 1997, 1990; Schlesinger et al., 2000).

4.3. Implications for rangeland management

Our results provide evidence that vegetation exerts a first-order control on selective erosion of soil fertility constituents through direct interactions with soil quality and surface characteristics. Perennial grass and litter cover was found to be strongly related to selective transport of finer sediment particles, organic C and N (Table 2). In the context of state-and-transition models, these results are consistent with the idea that measurements of soil erosion rate and soil fertility loss can be used to distinguish the alternative states. According to our previous studies (Chartier and Rostagno, 2006), a reversible transition or pathway occurs between the GS and DGS, associated to a decrease in the perennial grass cover. However, given a long time period, accelerated erosion will result in enough soil loss to prompt the transition to a new alternative stable state. The second transition involves changes in the soil and occurs when soil physical and chemical properties are altered irreversibly. This transition from DGS to DSS could be reached when, as consequence of the historical soil erosion, the A horizon decreases below a given thickness or, in the extreme case, a clay-rich horizon is exhumed and affect irreversibly hydrological and ecological processes (Chartier et al., 2011).

At the landscape scale, the extensive replacement of these grasslands by shrublands results in desertification through increased spatial heterogeneity of soil resources, which was also reported for other arid and semiarid ecosystems (e.g., Bestelmeyer et al., 2011; de Soyza et al., 2000; Schlesinger et al., 1990). In fact, changes in plant composition may be directly produced by intensive grazing or indirectly following a positive feedback mechanism through the decrease in perennial grass cover (Michaelides et al., 2009; Newman et al., 2006; van de Koppel et al., 2002). This biological feedback mechanism maintains or reinforces the degraded plant communities and limits the potential return to the previous conserved plant community (Bestelmeyer et al., 2006; Briske et al., 2008; Chartier and Rostagno, 2006). As can be ascertained from the above discussion, keeping the perennial grass dominated steppes may require a conservative management strategy and identify the opportunities that favor the pathway from DGS back to GS community.

Findings from this study suggest that pattern–process relationships can be integrated with state-and-transition models to provide a quantitative rationale and indicators for distinguishing transitional from degraded states of these rangelands. For example, measurements might focus on the perennial grass cover, distance between perennial grass plants or size and connectivity of the bare soil patches, based on these changes could be an early warning indicator of soil fertility degradation and loss of productivity. Additionally, monitoring strategies could focus on detecting changes over time in the relative proportion of plant communities, indicating whether transition communities between the reference and degraded communities (e.g., DGS) tend to expand.

5. Conclusions

In the northeastern rangelands of Patagonia, spatial variation imposed by plant composition and soil surface characteristics can explain the local differences in both sediment production and enrichment ratios of clay, organic C, and total N. The different processes involved in the soil loss by water erosion are the causative effects of the selective transport of clay, organic C and total N. In fact, while the GS represents a resource-conserving community, with lower sediment production and enrichment ratios of organic C and total N, the DSS represents an undesirable community with higher sediment yield and soil fertility losses. Alternatively, the DGS is a transitional community that, without management intervention to prevent soil erosion, will likely change to the more severely degraded DSS community.

Results from this study indicate that changes over time in the relative proportion of plant communities as well as in the distance between perennial grasses or size and connectivity of the bare soil patches may be early warning indicators of the potential for soil fertility degradation and loss of productivity. These results represent important tools for guiding land management activities, to design and implement monitoring schemes toward more conservative practices.

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