

SOIL EROSION IN THREE GRAZED PLANT COMMUNITIES IN NORTHEASTERN PATAGONIA

ROMINA G. PALACIO^{1*}, ALEJANDRO J. BISIGATO^{1,2} AND PABLO J. BOUZA^{1,2}

¹Centro Nacional Patagónico, Puerto Madryn, Argentina

²UNPSJB, Puerto Madryn, Argentina

Received 3 December 2013; Revised 19 March 2014; Accepted 24 March 2014

ABSTRACT

Grazing has been identified as the main cause of land degradation in Patagonia. However, land degradation is highly variable among areas, even within the same paddock. This strongly suggests that different plant communities differ in their resistance to land degradation. In this study, we have evaluated soil erosion at both microsite and community scales in coexisting plant communities subject to sheep grazing in northeastern Patagonia. Three plant communities coexist in the area: two shrub steppes dominated by *Chuquiraga avellanedae* Lorentz and *Nassauvia ulicina* (Hook. f.) Macloskie, and a grass steppe dominated by *Nassella tenuis* (Phil.) Barkworth. At a community scale, our results indicate that shrub steppes generally experienced soil erosion, whereas the grass steppe commonly did not show signs of soil erosion/deposition. At a microsite scale, non-vegetated soil surface types and degraded mounds never accumulated sediments, regardless of plant community. In contrast, we found that in some sites, the intact mounds and grasses entrapped sediments, but in other sites, soil erosion prevailed. Our results highlight the fact that soil erosion measurements are scale dependent, because results at microsite and community scales often differ. When comparing among communities, grass steppe is more intensely grazed, but at the same time, it shows less evidence of past and present erosion. In contrast, the *N. ulicina* community showed a direct relationship between grazing and soil erosion. Finally, soil erosion was not related to grazing in the *C. avellanedae* community. Our results demonstrate that the grass steppe is more resistant to land degradation than shrub steppes. Copyright © 2014 John Wiley & Sons, Ltd.

KEYWORDS: soil texture; soil superficial types; scales; plant cover; soil erosion

INTRODUCTION

Arid and semiarid ecosystems around the world exhibit contrasting levels of land degradation and, in extreme cases, evidence of desertification (Dregne, 1996). Most authors agree that desertification processes result from a mixture of climatic variability and human actions such as grazing, cultivation, wood extraction and/or mining (Darokh, 1998; Geist & Lambin, 2004; Yan & Cai, 2013; Zhao *et al.*, 2013). In turn, desertification has local, regional and global effects such as changes in carbon and water dynamics (Maestre *et al.*, 2006), biodiversity losses, reduced productivity, species invasions and soil erosion (Ludwig *et al.*, 2007).

Desertification is commonly associated with shrub invasion, and there are many scientific research papers around the world indicating that some grasslands have become shrublands during the last century (Archer *et al.*, 1995; Schlesinger *et al.*, 1996; Turnbull *et al.*, 2010; Álvarez-Martínez *et al.*, 2013). Grass-dominated and shrub-dominated communities differ in their aptitude to retain resources. In general, grasslands exhibit lower soil erosion rates than shrublands

(Abrahams *et al.*, 1995), but severely degraded shrublands can also show low sediment losses when most of the available sediments have already been exhausted (Cerdà, 1998). As land degradation is spatially heterogeneous, different communities (grasslands and shrublands) can occur simultaneously in the same area (Bestelmeyer, 2006).

Plant cover in arid ecosystems is low and clumped leading to a spatial pattern formed by two phases: bare soil and vegetation patches (Cerdà, 1997). Both phases constitute a source-sink system where bare soil areas provide water, sediments and nutrients, which are trapped by vegetation patches. Because of this, there are different microsites – where sediments are removed and re-deposited – coexisting at the same time in a desert community. However, at community scale, most soil is retained, because removals and depositions at different microsites are roughly in equilibrium. When plant patches are disturbed, for example, by overgrazing, they partially lose their ability to retain resources, which can be lost from the system (Tongway *et al.*, 2003; Ludwig *et al.*, 2005).

Several studies have found that soil erosion rates differ among microsites. Most of these studies compared vegetated and non-vegetated microsites (Ludwig *et al.*, 2007). Vegetated microsites exhibit lower erosion rates than exposed

Correspondence to: R. G. Palacio, Centro Nacional Patagónico, Blvd Brown 2915 U9120ACD Puerto Madryn, Argentina.
E-mail: palacio@cenpat.edu.ar

microsites, showing in some cases net deposition of sediments (Rostagno, 1989; Puigdefábregas *et al.*, 1999; Kakembo *et al.*, 2012). These differences between vegetated and non-vegetated microsites have been attributed to several mechanisms such as soil entrapment (Rostagno & Degorgue, 2011), presence of biological soil crusts (Belnap & Gillette, 1998), mechanical protection by plant canopy and plant litter (Bochet *et al.*, 1999; Kröpfl *et al.*, 2013; Gabarrón-Galeote *et al.*, 2012), changes in top soil structure and/or organic matter (OM) content (Bochet *et al.*, 1999), mechanical protection by mycorrhizal fungi (Burri *et al.*, 2013) and progressive salinization (Reid *et al.*, 1993).

Beyond these differences among vegetated and non-vegetated microsites, several authors have reported that different vegetated microsites exhibit contrasting soil erosion rates. For example, Bochet *et al.* (1999) found that deciduous shrubs protect less efficiently the soil beneath them than evergreen shrubs, leading to higher erosion rates below deciduous plants. Likewise, Cerdà (2001) measured soil erosion rates in non-vegetated plots differing in gravel cover. He, as Poesen *et al.* (1994), found that as gravel cover increases, soil erosion rate generally decreases. However, depending on the type of fine porosity, the surface slope, and the position and size of rock fragments, gravels can also enhance sediment yield (Poesen *et al.*, 1994).

Arid Patagonia has been grazed by sheep since the beginning of the XX century (Ares *et al.*, 1990). Many areas were overgrazed triggering erosive processes (Soriano & Movia, 1986) and reducing forage availability which frequently causes ranch abandonment. According to del Valle *et al.* (1998), more than 80% of Patagonia exhibits signs of moderate to severe desertification. These signs include total plant cover reduction, changes in floristic composition and the spatial pattern of vegetation (Bisigato & Bertiller, 1997; Ares *et al.*, 2003; Bisigato *et al.*, 2005). In this study, we evaluated soil erosion at microsite and community scales in three coexisting plant communities subject to sheep grazing in northeastern (NE) Patagonia. Our objectives were (i) to evaluate, at microsite and community scales, the magnitude and occurrence of soil erosion in three plant communities in NE Patagonia and (ii) to evaluate the effect of grazing intensity on soil erosion.

MATERIAL AND METHODS

Study Area

The study area is located in the north-east of the Chubut province in Patagonia (42°58'S, 64°33'W). It is an undulating plateau dissected by coastal valleys and interrupted by numerous depressions with playa lakes. According to Beeskow *et al.* (1987), the pediment-like plateau is an erosional surface of low relief. The geological substratum from which soils have developed is a gravelly sand to sandy clay

alluvial stratum of which the thickness ranges between 50 and 100 cm. This deposit of Holocene age rests on the Plio-Pleistocene formation locally known as 'Patagonian Gravel' (Fidalgo & Riggi, 1970; Haller *et al.*, 2005). The climate is arid and windy with a mean annual precipitation of 258 mm (1995–2004) (Chartier & Rostagno, 2006) and a mean annual temperature of 12.5 °C. The highest mean wind velocity (6 m s⁻¹) occurs during summer when southwest winds are dominant.

In the study area, no fire has been recorded for the last 50 years (Chartier *et al.*, 2009). Wind and water erosion are important geomorphic processes structuring the patchy soil and vegetation (Chartier *et al.*, 2009). Sheep grazing for wool production is the main use of these rangelands where continuous grazing is practiced extensively from moderate to heavy intensity (0.3 sheep ha⁻¹) in paddocks commonly exceeding 2,500 ha in size (Beeskow *et al.*, 1995).

Three plant communities were identified in the area: two shrub steppes (dominated by *Chuquiraga avellanadae* Lorentz and *Nassauvia ulicina* (Hook.f.) Macloskie, respectively) and a grass steppe (dominated by *Nassella tenuis* (Phil.) Barkworth) (Beeskow *et al.*, 1995). Plant cover varies from 15% to 50% in a patchy pattern in the case of shrub steppes. At a landscape scale, vegetation is characterized by a mosaic of the aforementioned communities, formed by stands hardly ever exceeding a few hectares in area. Soils are Calciargids/Natrargids (*C. avellanadae* community), Natrargids (*N. ulicina* community) and Haplocalcids (*N. tenuis* community).

Sampling

We selected five study sites in each plant community subjected to different grazing pressure, estimated through sheep faeces density (Table I). Faeces density is frequently used as a surrogate of grazing intensity in arid and semiarid environments around the world (Lange, 1969; Abensperg-Traun *et al.*, 1996; Adler *et al.*, 2005), and it was also extensively used in NE Patagonia (Bisigato & Bertiller, 1997; Ares *et al.*, 2003; Larreguy *et al.*, 2011). Sheep faeces may persist for years upon the surface of arid soils (Lange, 1969). The density of the sheep faeces was measured in 80 quadrats (0.25 m²) placed every 2.5 m on four transects (50 m) at each site. Afterwards, sites will be named with the initials of the dominant species in that community (CA, NU and NT) and a roman number from I to V, where V indicates the site with the highest grazing pressure. Plant cover by species was estimated along two 50 m transects (N-S and E-W) by the intercept method (Canfield, 1941).

At every site, a seventh linear 50 m transect was randomly located. For every metre along the transect, we drove two iron stakes into the ground with a separation distance between them of 25 cm. Soil level was measured in the middle

Table I. Faeces density (mean \pm SE), plant cover, number and diameter (mean \pm SE) of bare soil areas, and maximum slope at every study site

Community	<i>Nassauvia ulicina</i>					<i>Chuquiraga avellanae</i>		
	NU-I	NU-II	NU-III	NU-IV	NU-V	CA-I	CA-II	CA-III
Site								
Faeces density (number/m ²)	64.8 \pm 7.5	65.0 \pm 7.5	78.0 \pm 11.6	81.6 \pm 12.4	93.9 \pm 10.7	44.8 \pm 7.4	59.8 \pm 8.2	72.1 \pm 13.1
Shrubs cover (%)	47.83	34.1	36.83	42.62	33.28	30.75	23.44	36.17
<i>Chuquiraga aurea</i>	0.15	0.33	0	0.41	1.28	1.07	0	0
<i>Chuquiraga avellanae</i>	7.31	13.95	9.69	14.72	11.58	26.93	23.03	31.46
<i>Ephedra ochreatea</i>	0.36	0	0.17	0.05	0.66	0	0	0
<i>Lycium chilense</i>	0	0.10	0	0	0	0	0.35	0.05
<i>Margyricarpus pinnatus</i>	0	0	0	0	0	0	0	0
<i>Menodora robusta</i>	0	0	0	1.51	0	0	0	3.65
<i>Mulinum spinosum</i>	0	0	0	0	0.95	0.74	0	0
<i>Nassauvia ulicina</i>	39.08	19.72	26.76	25.04	18.77	0	0	0.58
<i>Paroniquia</i> sp.	0.01	0	0	0	0	0	0.06	0
<i>Perezia recurvata</i>	0	0	0	0.02	0	0	0	0
<i>Prosopidastrum globosum</i>	0.16	0	0.21	0.12	0.04	1.21	0	0.06
<i>Prosopis denudans</i>	0.76	0	0	0.75	0	0.80	0	0.37
<i>Schinus johnstonii</i>	0	0	0	0	0	0	0	0
Grasses cover (%)	0.08	0.88	0.49	0.37	0.55	5.61	7.39	0.86
<i>Jarava neaei</i>	0	0	0	0	0.05	0.10	0	0
<i>Nassella tenuis</i>	0.02	0.23	0.14	0.02	0.15	3.91	6.23	0.28
<i>Pappostipa humilis</i>	0.03	0.09	0.03	0.03	0	0	0	0
<i>Pappostipa speciosa</i>	0	0.40	0.27	0.29	0.35	1.33	1.02	0.42
<i>Piptochaetium napostaense</i>	0	0	0	0	0	0.22	0	0
<i>Poa ligularis</i>	0.03	0.16	0.05	0.03	0	0.05	0.14	0.16
Bare soil areas								
Number/50 m	95	75	99	111	97	120	139	43
Intercepted diameter (cm)	25.2 \pm 2.5	36.0 \pm 5.1	27.4 \pm 2.0	28.2 \pm 2.7	27.4 \pm 2.8	25.2 \pm 2.9	23.2 \pm 2.9	76.4 \pm 17.4
Maximum slope (%)	0.43	0.12	0.96	0.52	0.44	0.60	0.83	0.16

Site acronyms indicate the plant community (NU, *Nassauvia ulicina* community; CA, *Chuquiraga avellanae* community; and NT, *Nassella tenuis* community) and grazing pressure (increasing from I to V). Bold indicates life-form totals.

between each pair of stakes with a total station Pentax V-227. This electro-optical instrument measures distances with a precision of ± 3 mm within a range of 1,300 m. We also drove three 50 cm iron stakes forming a triangle containing the total station and the last transect (SM 1). These stakes were long enough to reach subsuperficial soil horizons exhibiting hard consistency (argilic and calcic horizons), what prevent their movement during the sampling period. They defined a reference plane, and the soil level measurements were always referred to it. This protocol avoids the inclusion of errors due to slight differences in the total station position in different sampling dates. Measurements were made seasonally over 2 years (eight sampling dates). However, in some sites, a sampling date was discarded because of missed or corrupted data.

Likewise, the soil surface type (modified from Bouza & del Valle, 1998, Table II and SM 2) was registered at each point where soil level was measured. We also recorded the intercepted length of each bare soil patch along the last transect.

Finally, for each site and each soil surface type, a superficial (0–2 cm) soil sample was randomly taken then combined accordingly to the proportion of each soil surface type in the site in a unique pooled sample. The soil texture

of the pooled samples was determined by the pipette method after removal of OM with H₂O₂ 30% (Kilmer & Alexander,

Table II. Soil surface types

Type	Name	Description
Non-vegetated types		
1	Bare Soil	Plane surface without gravel nor vegetation
2	Bare soil with gravel	Surface with some loose gravel
3	Desert pavement	Surface covered mostly by gravel
Vegetated types		
4	Grass	Presence of clumps of grass
5	Intact mound	Elevated area of soil on which stands a clump of shrub intact
6	Degraded mound	Elevated area of soil showing signs of erosion by water and/or wind with sparse vegetation

See supplementary material for pictures.

Table I. Continued

<i>Chuquiraga avellanadae</i>		<i>Nassella tenuis</i>				
CA-IV 79.8 ± 14.3	CA-V 92.6 ± 16.9	NT-I 54.6 ± 10.0	NT-II 76.5 ± 7.2	NT-III 105.0 ± 8.2	NT-IV 158.9 ± 18.7	NT-V 175.3 ± 11.0
29.6	26.06	6.84	3.58	1.65	1.46	0
0.02	0	0	0	0	0	0
29.58	26.06	5.49	0	1.61	0.43	0
0	0	0	0	0	0	0
0	0	0	0	0	0.21	0
0	0	0.1	0	0.04	0.09	0
0	0	0	0	0	0	0
0	0	0	3.01	0	0	0
0	0	0	0	0	0	0
0	0	0	0	0	0	0
0	0	1.25	0.57	0	0	0
0	0	0	0	0	0	0
0	0	0	0	0	0.73	0
1.07	3.92	8.25	12.93	14.46	16.21	16.12
0.02	0	0	0	0	0	0
0.46	2.91	7.40	9.65	14.46	15.79	15.99
0	0	0.03	0	0	0	0
0.50	0.63	0.61	2.88	0	0	0
0	0	0.16	0.36	0	0.42	0.13
0.09	0.38	0.05	0.04	0	0	0
64	78	174	208	227	233	280
55.3 ± 7.0	41.8 ± 4.1	18.5 ± 1.2	14.6 ± 0.8	14.0 ± 0.7	13.1 ± 0.8	11.3 ± 8.9
0.53	0.30	1.63	0.46	0.64	0.27	0.48

1949). OM content was analysed with the Walkley & Black (1934) method, and the soil erodibility (K) was estimated using the Universal Soil Loss Equation nomograph (Wischmeier & Smith, 1978). At every site, we also calculated the maximum slope of a planar trend surface adjusted to 441 soil level measurements taken with the total station in a 21 × 21 grid.

Data Analysis

The significance of the differences in plant cover, number of bare soil areas per transect, soil texture, OM content, soil erodibility and maximum slope among plant communities were evaluated by one-way analysis of variance using package 'stats' (Chambers & Hastie, 1992) of the R-Project (<http://www.r-project.org>). Before the analysis, percentages were angular transformed. Differences among communities in intercepted length of bare soil areas were evaluated by Kruskal–Wallis test, because data did not fulfil analysis of variance assumptions. Within each community, the relationship between variables and grazing pressure (as estimated by faeces density) was evaluated by regression analysis using package 'stats' (Chambers & Hastie, 1992) of the R-Project.

Temporal changes in soil level at every point in each transect were inspected by linear regression. We used correlation coefficients as effect sizes in a formal meta-analysis using the random effect DerSimonian–Laird approach

(Schulze, 2004). First, correlation coefficients were transformed using Fisher's z -transformation.

$$z_r = \frac{1}{2} \ln \left(\frac{1+r}{1-r} \right)$$

where r is the Pearson correlation coefficient.

This analysis was performed both, at a community scale (including all sampling points in each site) and at a microsite scale (for each soil surface type at every site). Meta-analysis was performed using the package 'metacor' (Laliberté *et al.*, 2010) of the R-Project (<http://www.r-project.org>).

The maximum slope of each site was analysed with the function 'surf.ls' from the R-package 'spatial' (Venables & Ripley, 2002), which fits a trend surface by least squares.

RESULTS

Sites Description

Plant communities differ in plant cover and number and size of bare soil patches (Table I). Shrub steppes (*N. ulicina* and *C. avellanadae* communities) exhibit higher total plant cover than the grass steppe (*N. tenuis* community) ($F_{2,12} = 54.5$, $p < 0.001$, $n = 15$) (Table I). Specific cover and shrub and grass covers were highly variable along grazing gradients in the shrub steppes. In contrast, grass cover increases

($n=5$, $F_{1,3}=11.6$, $p=0.042$, $r^2=0.726$) and shrub cover decreases ($n=5$, $F_{1,3}=12.1$, $p=0.040$, $r^2=0.735$) with grazing (i.e. faeces density) in the grass steppe. Shrub steppes exhibited less and bigger bare soil patches than grass steppe sites ($F_{2,12}=27.0$, $p<0.001$, $n=15$; $H_{(2)}=9.62$, $p=0.008$, $n=15$, respectively) (Table I). Along the grazing gradient, bare soil patches were smaller ($n=5$, $F_{1,3}=12.2$, $p=0.040$, $r^2=0.738$) and more numerous ($n=5$, $F_{1,3}=11.5$, $p=0.042$, $r^2=0.726$) in the grass steppe. Concerning soil surface types, *N. ulicina* community exhibits an increase in non-vegetated soil surface types and a decrease in vegetated soil surface types along the grazing gradient (Figure 1). This change was a consequence of the rise in the frequency of desert pavements and a reduction in the frequency of intact mounds. In contrast to the clear pattern found in the *N. ulicina* community, changes in the relative frequency of soil surface types in the *C. avellanadae* community were not related to

grazing pressure (Figure 1). However, it should be mentioned here that this community was the only one among the three considered in this study where the frequency of non-vegetated soil surface types exceeded that of vegetated ones. Finally, in the *N. tenuis* community, the frequency of vegetated soil surface types was the highest at intermediate grazing pressures (Figure 1). Both degraded and intact mounds are restricted to low grazing pressures, whereas soil with gravel is found at the highest grazing pressures. Plant communities did not differ in maximum slope (Table I), and no relationship between grazing pressure (i.e. sheep faeces density) and maximum slope was found within communities.

Soil Texture, Soil Erodibility and Organic Matter Content

Soil texture was sandy loam in most sites. On average, *N. tenuis* soils have more silt ($F_{2,12}=4.48$, $p=0.035$, $n=15$) and less sand ($F_{2,12}=3.9$, $p=0.05$, $n=15$) than those belonging to shrub steppes (Table III). Within communities, a clear pattern in soil texture along the grazing gradient was only evident in the case of the *N. tenuis* community, where sand content decreases ($n=5$, $F_{1,3}=10.2$, $p=0.05$, $r^2=0.697$) and silt content marginally increases ($n=5$, $F_{1,3}=9.3$, $p=0.056$, $r^2=0.674$) with grazing pressure (Table III).

Plant communities did not differ in soil erodibility (K) ($F_{2,12}=0.58$, $p=0.57$, $n=15$) but they showed significant differences in OM content ($F_{2,12}=12.10$, $p=0.001$, $n=15$) (Table III). *N. tenuis* community exhibited higher OM content than shrub steppes. Soil erodibility (K) and OM content were not related to grazing pressure within communities.

Soil Erosion at a Community Scale

We found evidence of soil losses at a community scale in most of the shrub steppes sites. All NU sites, irrespective of

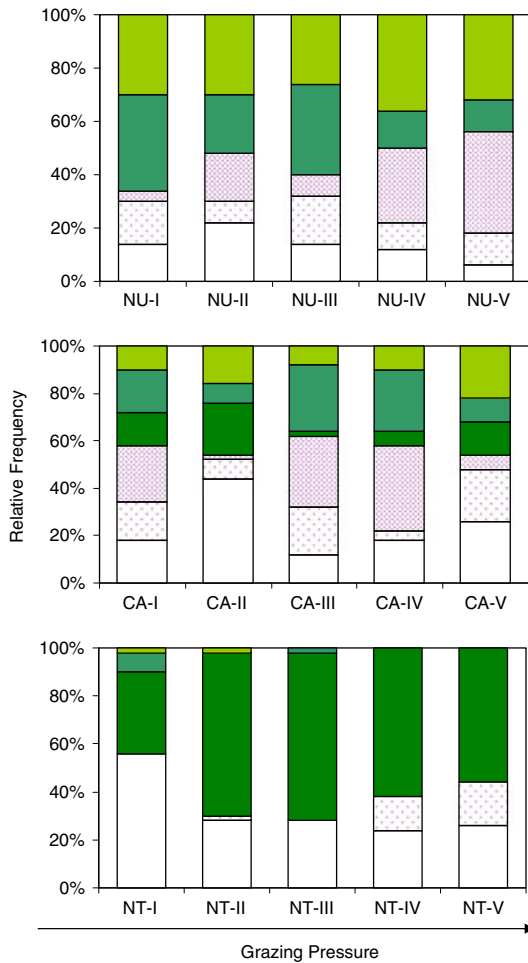


Figure 1. Relative frequency of soil surface types in each study site. Soil surface types: \square = bare soil, \square (dotted) = bare soil with gravel, \square (cross-hatched) = desert pavement, \square (green) = grass, \square (dark green) = intact mound and \square (light green) = degraded mound. Site acronyms as in Table I. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr.

Table III. Superficial (0–2 cm) soil texture, organic matter content and soil erodibility (K) in every study site

Site	% sand	% clay	% silt	% OM	K
NU-I	78.71	4.27	17.01	1.65	0.56
NU-II	67.49	9.24	23.07	1.88	0.48
NU-III	72.97	3.86	23.16	1.78	0.49
NU-IV	71.36	9.25	19.38	1.58	0.55
NU-V	60.46	8.36	31.18	1.47	0.44
CA-I	68.53	5.05	26.42	1.88	0.49
CA-II	70.73	5.55	23.72	1.54	0.48
CA-III	67.62	11.4	20.98	1.93	0.52
CA-IV	67.58	11.2	21.23	1.53	0.52
CA-V	73.17	4.66	22.17	1.47	0.53
NT-I	69.44	7.64	22.92	1.90	0.54
NT-II	62.67	7.87	29.45	2.29	0.48
NT-III	61.68	7.76	30.56	2.46	0.49
NT-IV	60.41	7.72	31.87	2.55	0.45
NT-V	57.88	8.72	33.41	2.10	0.46

Site acronyms as in Table I. OM, organic matter.

their grazing pressure, exhibited significant soil losses during the studied period, with z_r values ranging between -0.127 and -0.652 (Figure 2). Only the CA-V site exhibits net soil deposition ($z_r = 0.219$), whereas the remaining sites belonging to this community exhibited soil losses (z_r values ranging between -0.188 and -0.526 , Figure 2). Finally, three grass steppe sites did not show significant changes in soil level. Only NT-I and NT-III sites showed significant soil losses ($z_r = -0.241$ and -0.312 , respectively).

Soil Erosion at a Microsite Scale

At a microsite scale, some interesting patterns emerged. In the *N. ulicina* community, all non-vegetated soil surface types (one to three) exhibited significant soil losses in at least one site (z_r values ranging between -0.260 and -0.962 , Figure 3). In

contrast, among vegetated soil surface types, we did not find evidence of soil level changes in intact mounds ($p > 0.05$), although in four sites degraded mounds significantly lost soil, with z_r values ranging between -0.269 and -0.677 (Figure 3).

In the *C. avellanadae* community, we found evidence of significant soil losses from non-vegetated soil surface types (one to three) but only at intermediate grazing pressures (sites CA-II to CA-IV, z_r values ranging between -0.355 and -0.647 , Figure 3). In contrast, vegetated soil surface types exhibited significant losses at low grazing pressures (sites CA-I and CA-II, z_r values -0.449 and -0.537 , respectively), whereas at sites CA-III and CA-V, significant soil deposition was found (z_r values 0.325 and 0.413 , respectively).

Finally, in the *N. tenuis* community, only bare soil and grasses (soil surface types 1 and 4) exhibited temporal changes in soil level. Bare soil areas showed soil losses in sites NT-I and NT-III (z_r values -0.647 and -0.434 , respectively). In contrast, grasses entrapped soil at both extremes of the grazing pressure gradient (NT-I and NT-V, z_r values 0.326 and 0.196) but lost it at intermediate grazing pressure (site NT-III, $z_r = -0.287$).

DISCUSSION

At the community level, most of the grass steppe sites did not show signs of soil erosion/deposition, whereas the opposite was true for shrub steppes. In general, erosion rates are lower in grasslands than in shrublands (Breshears *et al.*, 2003). Grasslands exhibit lower soil erosion rates because grasses are more effective at entrapping sandy loess (Pye, 1995) and protecting surface soil against rain drops. As well pitted microtopography and reduced connectivity of bare soil patches favours water infiltration preventing the transport of fine sediments (Okin *et al.*, 2009; Turnbull *et al.*, 2010; Jin *et al.*, 2013), which are first removed by erosion (Pye, 1987). Accordingly, our results showed that superficial soil in the grass steppe sites has more silt and less sand than the soil of shrub steppes, more OM content, and that desert pavements are absent from the former. It indicates that historical erosion rates were higher in shrub steppes than in grass steppes sites, although plant cover is lower in the latter.

At a microsite scale, our results showed that, regardless of the plant community, non-vegetated soil surface types (bare soil, bare soil with gravel and desert pavement) and degraded mounds never accumulated sediments (i.e. these microsites remained stable or underwent erosion). In contrast, when the response of intact mounds and grasses is analysed, a clear pattern did not emerge: in some sites, these microsites accumulated sediments, but they lost them in others. Some authors suggest that during the life of the shrub, the mound is raised or increased in size by the addition of organic and inorganic materials (Boeken & Shachak, 1994; Bochet *et al.*, 1999). Soil particles mobilized by wind

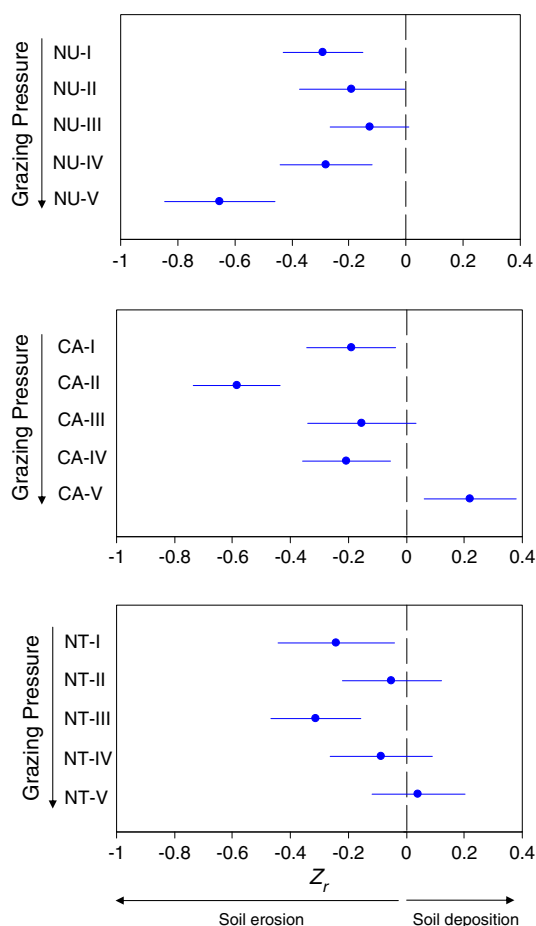


Figure 2. Soil erosion/deposition at community scale at every site. Effect sizes z_r are z -transformed Pearson correlation coefficients (r) between soil level and date, following the DerSimonian–Laird approach. A negative value of z_r indicates a significant loss of soil during the study period. Points are means $\pm 95\%$ confidence intervals. Means with confidence intervals overlapping the dotted reference line (0% change) show no significant trend of soil level in time. Site acronyms as in Table I. $n = 50$ in all cases. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr. Correction added on 22 May 2014, after first online publication. The first and second panels of Figure 2 have been interchanged to correct an error in the original published version.

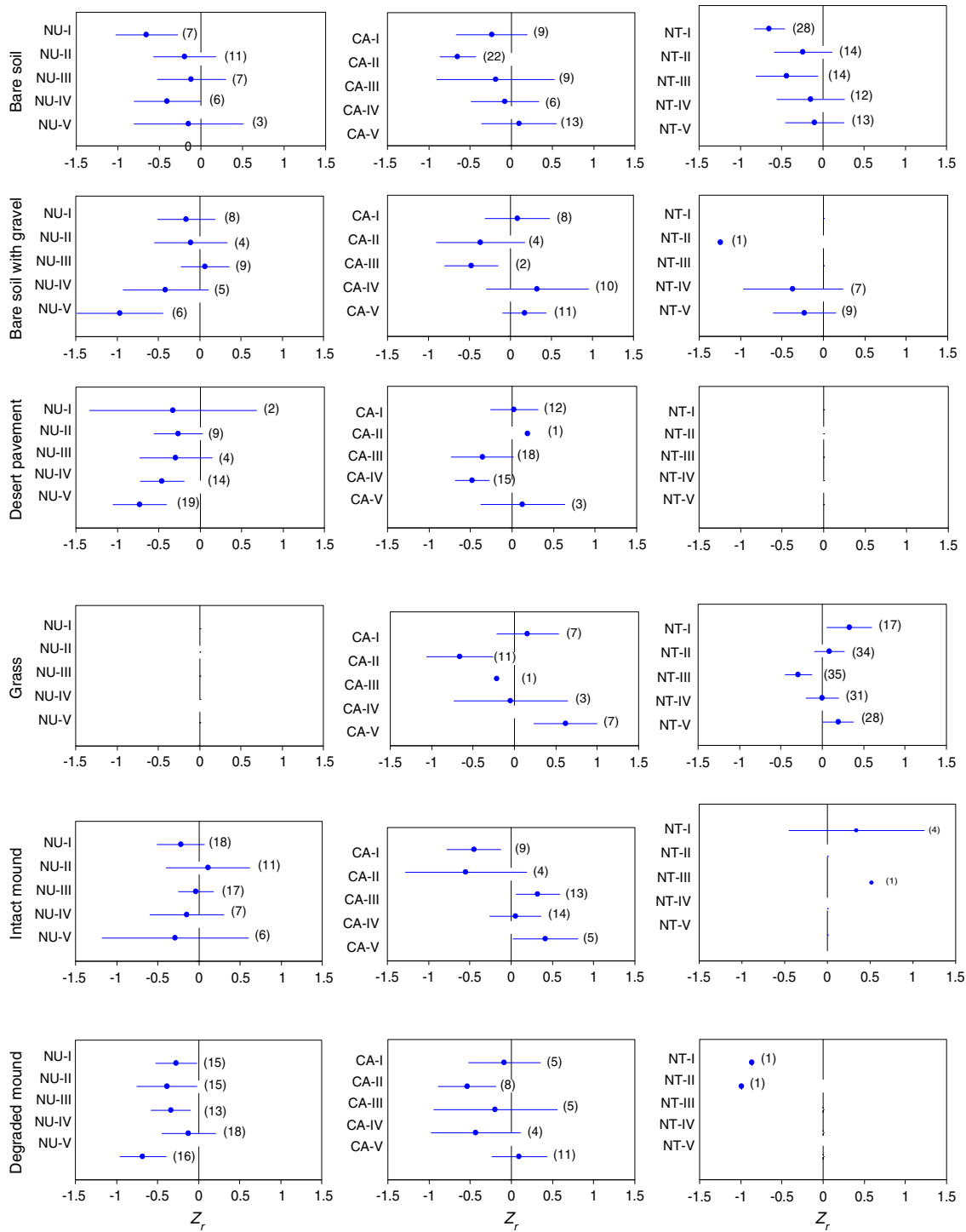


Figure 3. Soil erosion/deposition at microsite scale at every site. Effect sizes z_r are z -transformed Pearson correlation coefficients (r) between soil level and date, following the DerSimonian–Laird approach. A negative value of z_r indicates a significant loss of soil during the study period. Points are means $\pm 95\%$ confidence intervals, with number of points in parentheses. Means with confidence intervals overlapping the dotted reference line (0% change) show no significant trend of soil level in time. Site acronyms as in Table I. This figure is available in colour online at wileyonlinelibrary.com/journal/ldr.

erosion can be caught by shrub canopies and accumulate below shrubs (Schlesinger & Pilmanis, 1998; Kröpfel *et al.*, 2013; Lozano *et al.*, 2013). In contrast, water erosion does

not contribute to the deposit of sediments in the soil beneath the shrub canopies because the microtopography diverts water flow away from the higher elevation microsites under

shrubs (Ravi *et al.*, 2010). These results highlight the importance of plant cover to prevent or reduce soil erosion at a microsite scale (Breshears *et al.*, 2003; Puigdefábregas, 2005), because all non-vegetated microsites exhibited evidence of sediment loss. On the other hand, our results showed that soil erosion prevails in degraded mounds where plant cover is fragmented and soil surface is exposed, as previously reported by Tongway *et al.* (2003), Golodets & Boeken (2006) and Li *et al.* (2008), among others.

Our results also highlight the fact that soil erosion measurements are scale dependent, because results at microsite and community scales often differed. This scale dependence was previously reported in studies where the linear extrapolation of small-scale results underestimates or overestimates large-scale measurements of soil erosion (e.g. Tongway *et al.*, 2003; Puigdefábregas, 2005; Parsons *et al.*, 2006). Moreover, our results support higher scale processes because, although at a community scale most sites lost sediments, CA-V showed evidence of deposition (i.e. this site traps sediments from outside the site).

These results agree with previous studies carried out in NE Patagonia, which found that bigger bare soil spaces accelerate erosion rates (Beeskow *et al.*, 1995; Parizek *et al.*, 2002). Likewise, Chartier & Rostagno (2006) applied simulated rainfall on runoff experimental plots (1 m² in area) and found that soil erosion rates were higher in shrub steppes (*C. avellanadae* community) than in grass steppes (*N. tenuis* community). This was partially attributed to a greater erodibility in the former community (Chartier & Rostagno, 2010). In addition, Parizek *et al.* (2002) found that infiltration rates in the *N. tenuis* community were higher than those of the *C. avellanadae* community. However, infiltration rates strongly differed between microsites inside the latter community: non-vegetated areas showed the lowest infiltration rate, while infiltration rates at soil mounds below shrubs exceeded that of the grass steppe.

Soil erosion was present in all NU sites, but it was highest in the site exhibiting the most intense grazing pressure (NU-V). Coincidentally, this site had the lowest plant cover among NU sites and showed the highest proportion of soil superficial type 3 (desert pavement) and the lowest proportion of soil superficial type 5 (intact mounds). At a microsite scale, soil erosion was evident (and highest among NU sites) in bare soil with gravel, desert pavement and degraded mounds (soil superficial types 2, 3 and 6). These results indicate that in some circumstances, gravel cover can be ineffective to prevent soil erosion, as previously reported by Poesen *et al.* (1994). In contrast, soil level changes were highly variable among CA sites. Although most of the sites exhibited significant soil losses, the site with the highest grazing pressure showed soil deposition. This difference among sites cannot be related to changes in the frequency of superficial soil types or total plant cover because the

remaining CA sites showed similar values of these variables. Among microsites, soil deposition took place in grass and intact mounds (soil superficial types 4 and 5). Finally, although NT sites exhibited the lowest plant cover among plant communities, most of NT sites did not show evidence of soil erosion/deposition. Only two lightly grazed sites (NT-I and NT-III) exhibited signs of soil erosion. Coincidentally, silt content (an indicator of historical soil erosion) and number of bare soil areas were highest at sites exhibiting the most intense grazing pressure. At a microsite scale, soil losses are mainly the consequence of bare soil erosion. Surprisingly, we did not find a relation between soil erosion and soil erodibility within communities.

The present study, despite the fact that it is limited by a lack of true replication of grazing level within communities, indicates complex interactions among vegetation heterogeneity, grazing intensity and soil erosion. Persistence of desertified ecosystems around the world has been mainly attributed to the occurrence of two practically irreversible processes in human terms and within practical economic limitations: shrub invasion and soil erosion (Friedel, 1991; Dodd, 1994). Although it is not the case in NE Patagonia, it should be mentioned that if encroaching shrubs are palatable, a combination of grazing by goats and managerial practices (e.g. burning, trimming and clearing) can reverse the process (Álvarez-Martínez *et al.*, 2013). When comparing among vegetation communities, our results indicate that NT is the most intensely grazed, but simultaneously, it is the one that shows less evidence of past and present erosion. Shrub cover decreases and grass cover increases with grazing in this community, suggesting that vegetation heterogeneity controls sheep grazing behaviour. In both shrub steppes, the relationship between grazing intensity and plant cover was not as clear as in the grass steppe. However, grass cover was higher at CA-I and CA-II than at more intensely grazed sites belonging to this community, suggesting that in this case, preferred species (i.e. grasses) are excluded from intensely grazed sites (i.e. vegetation heterogeneity is due to grazing). Nevertheless, we did not find a relationship between grazing pressure and soil erosion in the CA community. Finally, the NU community exhibited a direct relationship between grazing pressure and soil erosion, and at the same time, a general trend of shrub cover reduction was evident. These results indicate a direct relationship among grazing, plant cover and soil erosion.

Previous studies have suggested that the most important change in the study area as a consequence of grazing is the transformation of grass steppes into shrub steppes (Beeskow *et al.*, 1995; Chartier & Rostagno, 2006). Our results showed that grass steppes exhibited lower soil erosion rates than shrub steppes, although they are under the highest grazing pressure. Also, we did not find a relationship between soil erosion and grazing pressure among grass steppe sites.

This indicates that grass steppes are more resistant to grazing disturbance than shrub steppes and suggests that this resistance could slow down the conversion of grasslands into shrublands. On the other hand, the fact that most shrub steppe sites showed evidence of soil erosion suggests that the reversion of the process is very improbable because a substantial fraction of the superficial soil is lost. Thus, our results indicate that managerial practices to prevent the grass steppe conversion into shrub steppe should be developed.

CONCLUSION

We found evidence of a direct relationship among grazing, plant cover reduction and/or soil erosion in the case of shrub steppes. In contrast, although they are under the highest grazing pressures, grass steppes seem to be more resistant to grazing. Our results also highlight the fact that soil erosion measurements are scale dependent, because results at microsite and community scales often differ.

ACKNOWLEDGEMENTS

This work was supported by CONICET PIP-1122008-0101238 and a CONICET fellowship to R.G.P. Lina Videla, Claudia Saín, Iván Tolaba, Ana Casalini and María V. Campanella gave us important support during field work.

REFERENCES

- Abensperg-Traun M, Smith GT, Arnold GW, Steven DE. 1996. The effects of habitat fragmentation and livestock-grazing on animal communities in remnants of gimlet *Eucalyptus salubris* woodland in the Western Australian Wheatbelt I. Arthropods. *Journal of Applied Ecology* **33**: 1281–1301.
- Abrahams AD, Parsons AJ, Wainwright J. 1995. Effects of vegetation change on interrill runoff and erosion, Walnut Gulch, southern Arizona. *Geomorphology* **13**: 37–48.
- Adler P, Milchunas D, Sala O, Burke I, Lauenroth W. 2005. Plant traits and ecosystem grazing effects: comparison of U.S. Sagebrush steppe and Patagonian steppe. *Ecological Applications* **15**(2): 774–792.
- Álvarez-Martínez J, Gómez-Villar A, Lasanta T. 2013. The use of goats grazing to restore pastures invaded by shrubs and avoid desertification: a preliminary case study in the Spanish Cantabrian Mountains. *Land Degradation & Development*. DOI: 10.1002/ldr.2230.
- Archer SR, Schimel DS, Holland EA. 1995. Mechanisms of shrubland expansion: land use, climate or CO₂? *Climatic Change* **29**: 91–99.
- Ares J, Beeskov AM, Bertiller M, Rostagno CM, Irisarri M, Anchorena J, Defossé G, Merino C. 1990. Structural and dynamic characteristics of overgrazed lands of northern Patagonia, Argentina. In *Managed grasslands: regional studies*, Breymeyer A (ed.). Elsevier: Amsterdam, Netherlands; 149–175.
- Ares J, Del Valle HF, Bisigato AJ. 2003. Detection of process-related changes in plant patterns at extended spatial scales during early dryland desertification. *Global Change Biology* **9**: 1643–1659. DOI: 10.1046/j.1529-8817.2003.00690.x.
- Beeskov AM, del Valle HF, Rostagno CM. 1987. Los sistemas fisiográficos de la región árida y semiárida de la Provincia del Chubut. Secretaría de Ciencia y Tecnología: Bariloche, Argentina.
- Beeskov AM, Elissalde N, Rostagno CM. 1995. Ecosystem changes associated with grazing intensity on the Punta Ninfas rangelands of Patagonia, Argentina. *Journal of Range Management* **48**: 517–522.
- Belnap J, Gillette DA. 1998. Vulnerability of desert biological soil crusts to wind erosion: the influences of crust development, soil texture, and disturbance. *Journal of Arid Environments* **39**: 133–142.
- Bestelmeyer BT. 2006. Threshold concepts and their use in rangeland management and restoration: the good, the bad, and the insidious. *Restoration Ecology* **14**: 325–329. DOI: 10.1111/j.1526-100X.2006.00140.x.
- Bisigato AJ, Bertiller MB. 1997. Grazing effects on patchy dryland vegetation in northern Patagonia. *Journal of Arid Environments* **36**: 639–653. DOI: 10.1006/jare.1996.0247.
- Bisigato AJ, Bertiller MB, Ares JO, Pazos GE. 2005. Effect of grazing on plant patterns in arid ecosystems of Patagonian Monte. *Ecography* **28**: 561–572. DOI: 10.1111/j.2005.0906-7590.04170.
- Bochet E, Rubio JL, Poesen J. 1999. Modified topsoil islands within patchy Mediterranean vegetation in SE Spain. *Catena* **38**: 23–44.
- Boeken B, Shachak M. 1994. Desert plant communities in human-made patches-implications for management. *Ecological Applications* **4**: 702–716.
- Bouza PJ, Del Valle HF. 1998. Propiedades micromorfológicas del suelo superficial y subsuperficial en un ambiente pedemontano árido de Patagonia, Argentina. *Ciencia del Suelo* **16**: 30–38.
- Breshears DD, Whicker JJ, Johansen MP, Pinder JE. 2003. Wind and water erosion and transport in semi-arid shrubland, grassland and forest ecosystems: quantifying dominance of horizontal wind-driven transport. *Earth Surface Processes & Landforms* **28**: 1189–1209. DOI: 10.1002/esp.1034.
- Burri K, Gromke C, Graf F. 2013. Mycorrhizal fungi protect the soil from wind erosion: a wind tunnel study. *Land Degradation & Development* **24**: 385–392. DOI: 10.1002/ldr.1136.
- Canfield RH. 1941. Application of the line interception method in sampling range vegetation. *Journal of Forestry* **39**: 388–394.
- Cerdà A. 1997. The effect of patchy distribution of *Stipa tenacissima* L. on runoff and erosion. *Journal of Arid Environments* **36**: 37–51.
- Cerdà A. 1998. The influence of aspect and vegetation on seasonal changes in erosion under rainfall simulation on a clay soil in Spain. *Canadian Journal of Soil Science* **78**: 321–330.
- Cerdà A. 2001. Effects of rock fragment cover on soil infiltration, interrill runoff and erosion. *European Journal of Soil Science* **52**: 59–68.
- Chambers JM, Hastie TJ. 1992. Statistical models in S. Wadsworth & Brooks/Cole: Pacific Grove, California.
- Chartier MP, Rostagno M. 2006. Soil erosion thresholds and alternative states in northeastern Patagonian rangelands. *Rangeland Ecology & Management* **59**: 616–624.
- Chartier MP, Rostagno CM. 2010. Pastizales semiáridos del noreste de la Patagonia. *Ciencia del Suelo* **28**: 91–103.
- Chartier MP, Rostagno CM, Roig FA. 2009. Soil erosion rates in rangelands of northeastern Patagonia: a dendrogeomorphological analysis using exposed shrub roots. *Geomorphology* **106**: 344–351. DOI: 10.1016/j.geomorph.2008.11.015.
- Darokh MBK. 1998. The nature, causes and consequences of desertification in the drylands of Africa. *Land Degradation & Development* **9**: 1–20. DOI: 10.1002/(SICI)1099-145X(199801/02)9:1<1::AID-LDR263>3.0.CO;2-8.
- Del Valle H, Elissalde NO, Gagliardini DA. 1998. Status of desertification in the Patagonian region: assessment and mapping from satellite imagery. *Arid Soil Research & Rehabilitation* **12**: 1–27.
- Dodd JL. 1994. Desertification and degradation of Africa's rangelands. *Rangelands* **16**: 180–183.
- Dregne H. 1996. Desertification: challenges ahead. *Annals of Arid Zone* **35**: 305–311.
- Fidalgo F, Riggi JC. 1970. Consideraciones geomórficas y sedimentológicas sobre los Rodados Patagónicos. *Revista de la Asociación Geológica Argentina* **25**: 430–443.
- Friedel MH. 1991. Range condition assessment and the concept of thresholds: a viewpoint. *Journal of Range Management* **44**: 422–426.
- Gabarrón-Galeote MA, Martínez-Murillo JF, Ruiz-Sinoga JD. 2012. Relevant effects of vegetal cover and litter on the soil hydrological response of two contrasting Mediterranean hillslopes at the end of the dry season (south of Spain). *Hydrological Processes* **26**: 1729–1738. DOI: 10.1002/hyp.8254.

- Geist H, Lambin EF. 2004. Dynamic causal patterns of desertification. *BioScience* **54**: 817–829.
- Golodets C, Boeken B. 2006. Moderate sheep grazing in semiarid shrubland alters small-scale soil surface structure and patch properties. *Catena* **65**: 285–291. DOI: 10.1016/j.catena.2005.12.005.
- Haller M, Meister CM, Monti AJ, Weiler N. 2005. Hoja Geológica 4366-II, "Puerto Madryn", Provincia del Chubut. Instituto de Geología y Recursos Minerales (SEGEMAR), Boletín 289, Buenos Aires. 39.
- Jin Z, Dong YS, Qi YC, Liu WG, An ZS. 2013. Characterizing variation in soil particles-size distribution along a grass-desert shrub transition in the Ordos Plateau in Inner Mongolia China. *Land Degradation & Development* **24**: 141–146. DOI: 10.1002/ldr.1112.
- Kakembo V, Ndlela S, Cammeraat E. 2012. Trends in vegetation patchiness loss and implications for landscape function: the case of *Pteronia incana* invasion in the Eastern Cape Province, South Africa. *Land Degradation & Development* **23**: 548–556. DOI: 10.1002/ldr.2175.
- Kilmer VJ, Alexander LT. 1949. Methods of making mechanical analysis of soils. *Soil Science* **68**: 15–24.
- Kröpfl AI, Cecchi GA, Villasuso NM, Distel RA. 2013. Degradation and recovery processes in semi-arid patchy rangelands of northern Patagonia, Argentina. *Land Degradation & Development* **24**: 393–399. DOI: 10.1002/ldr.1145.
- Laliberté E, Wells J, Declerck F, Metcalfe DJ, Catterall CP, Queiroz C, Aubin I, Bonser SP, Ding Y, Fraterrigo JM, McNamara S, Morgan JW, Merlos DS, Vesik P, Mayfield MM. 2010. Land-use intensification reduces functional redundancy and response diversity in plant communities. *Ecology Letters* **13**: 76–86. DOI: 10.1111/j.1461-0248.2009.01403.x.
- Lange RT. 1969. The piosphere: sheep track and dung patterns. *Journal of Range Management* **2**: 396–400.
- Larreguy C, Carrera AL, Bertiller MB. 2011. Production and turnover rates of shallow fine roots in rangelands of the Patagonian Monte, Argentina. *Ecological Research* **27**: 61–68. DOI: 10.1007/s11284-011-0869-5.
- Li XJ, Li XR, Song WM, Gao YP, Zheng JG, Jia RL. 2008. Effects of crust and shrub patches on runoff, sedimentation, and related nutrient (C, N) redistribution in the desertified steppe zone of the Tengger Desert, northern China. *Geomorphology* **96**: 221–232. DOI: 10.1016/j.geomorph.2007.08.006.
- Lozano FJ, Soriano M, Martínez S, Asensio C. 2013. The influence of blowing soil trapped by on fertility in Tabernas District. *Land Degradation & Development* **24**: 575–581. DOI: 10.1002/ldr.2186.
- Ludwig JA, Wilcox BP, Breshears DD, Tongway DJ, Imeson AC. 2005. Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. *Ecology* **86**: 288–297. DOI: 10.1890/03-0569.
- Ludwig JA, Bartley R, Hawdon AA, Abbott BN, McJannet D. 2007. Patch configuration non-linearly affects sediment loss across scales in a grazed catchment in north-east Australia. *Ecosystems* **10**: 839–845. DOI: 10.1007/s10021-007-9061-8.
- Maestre FT, Reynolds JF, Huber-sannwald E, Herrick J, Smith MS. 2006. Understanding global desertification: biophysical and socioeconomic dimensions of hydrology. In *Dryland ecohydrology*, D'Odorico P, Porporato A (eds). Springer: Dordrecht, The Netherlands; 315–332.
- Okin GS, Parsons AJ, Wainwright J, Herrick JE, Bestelmeyer BT, Peters DC, Fredrickson EL. 2009. Do changes in connectivity explain desertification? *BioScience* **59**: 237–244. DOI: 10.1525/bio.2009.59.3.8.
- Parizek B, Rostagno CM, Sottini R. 2002. Soil erosion as affected by shrub encroachment in north-eastern Patagonia. *Journal of Range Management* **55**: 43–48.
- Parsons AJ, Brazier RE, Wainwright J, Powell DM. 2006. Scale relationships in hillslope runoff and erosion. *Earth Surface Processes & Landforms* **31**: 1384–1393. DOI: 10.1002/esp.1345.
- Poesen J, Torri D, Bunte K. 1994. Effects of rock fragments on soil erosion by water at different spatial scales: a review. *Catena* **23**: 141–166.
- Puigdefábregas J. 2005. The role of vegetation patterns in structuring runoff and sediment fluxes in drylands. *Earth Surface Processes & Landforms* **30**: 133–147. DOI: 10.1002/esp.1181.
- Puigdefábregas J, Sole A, Gutierrez L, Del Barrio G, Boer M. 1999. Scales and processes of water and sediment redistribution in drylands: results from the Rambla Honda field site in southeast Spain. *Earth-Science Reviews* **48**: 39–70. DOI: 10.1016/S0012-8252(99)00046-X.
- Pye K. 1987. *Aeolian Dust and Dust Deposits*. Academic Press: London.
- Pye K. 1995. The nature, origin and accumulation of loess. *Quaternary Science* **14**: 653–667.
- Ravi S, Breshears DD, Huxman TE, D'Odorico P. 2010. Land degradation in drylands: interactions among hydrologic-aeolian erosion and vegetation dynamics. *Geomorphology* **116**: 236–245. DOI: 10.1016/j.geomorph.2009.11.023.
- Reid DA, Graham RC, Southard RJ, Amrhein C. 1993. Slickspot soil genesis in the Carrizo Plain, California. *Soil Science Society of America Journal* **57**: 162–168.
- Rostagno CM. 1989. Infiltration and sediment production as affected by soil surface conditions in a shrubland of Patagonia, Argentina. *Journal of Range Management* **42**: 382–385.
- Rostagno CM, Degorgue G. 2011. Desert pavements as indicators of soil erosion on arid soils in north-east Patagonia (Argentina). *Geomorphology* **134**: 224–231. DOI: 10.1016/j.geomorph.2011.06.031.
- Schlesinger W, Pilmanis AM. 1998. Plant-soil interactions in deserts. *Biogeochemistry* **42**: 169–187.
- Schlesinger WH, Raikes JA, Hartley AE, Cross AF. 1996. On the spatial pattern of soil nutrients in desert ecosystems. *Ecology* **77**: 364–374.
- Schulze R. 2004. *Meta-analysis: a comparison of approaches*. Hogrefe & Huber: Cambridge, Ashland.
- Soriano A, Movia CP. 1986. Erosión y desertización en la Patagonia. *Interciencia* **11**: 77–83.
- Tongway D, Sparrow A, Friedel M. 2003. Degradation and recovery processes in arid grazing lands of central Australia. Part 1: soil and land resources. *Journal of Arid Environments* **55**: 301–326. DOI: 10.1016/S0140-1963(03)00025-9.
- Turnbull L, Wainwright J, Brazier RE. 2010. Changes in hydrology and erosion over a transition from grassland to shrubland. *Hydrological Processes* **24**: 393–414. DOI: 10.1002/hyp.7491.
- Venables WN, Ripley BD. 2002. *Modern applied statistics with S*. 4th edn. Springer: New York.
- Walkley A, Black IA. 1934. An examination of Degtjareff method for determining soil organic matter, and a proposed modification of the chromic acid titration method. *Soil Science* **37**: 29–37.
- Wischmeier WH, Smith DD. 1978. Predicting rainfall erosion losses. USDA Agricultural Handbook No. 537.
- Yan X, Cai YL. 2013. Multi-scale anthropogenic driving forces of karst rocky desertification in Southwest China. *Land Degradation & Development*. DOI: 10.1002/ldr.2209.
- Zhao G, Mu X, Wen Z, Wang F, Gao P. 2013. Soil erosion, conservation and eco-environment changes in the loess plateau of China. *Land Degradation & Development*. DOI: 10.1002/ldr.2246.

SUPPORTING INFORMATION

Additional supporting information may be found in the online version of this article at the publisher's web-site.