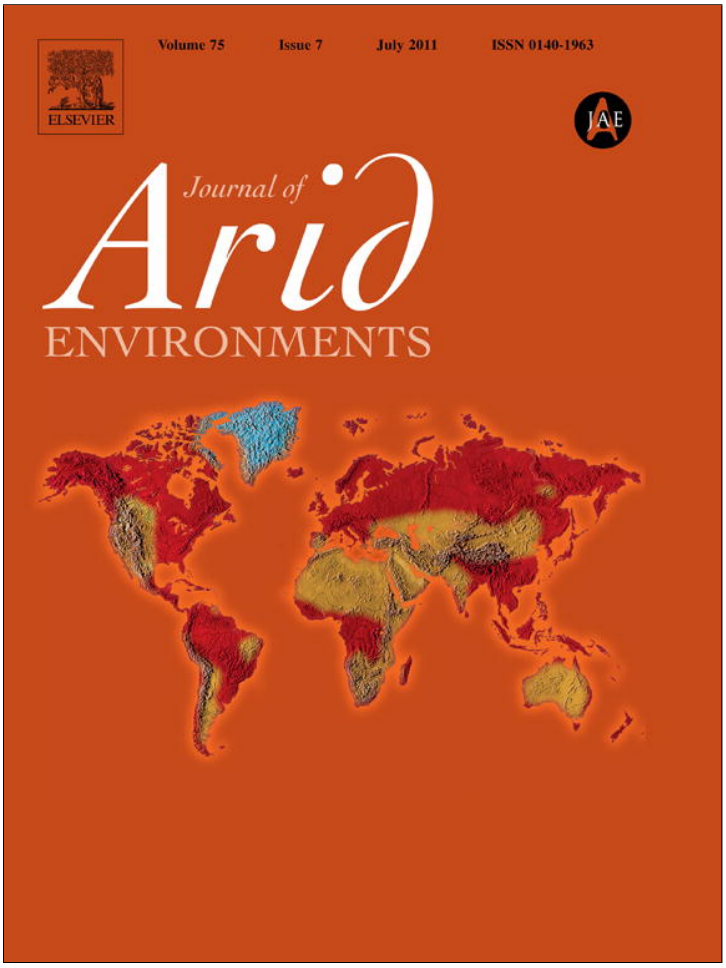
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Effects of soil degradation on infiltration rates in grazed semiarid rangelands of northeastern Patagonia, Argentina

M.P. Chartier [a,*], C.M. Rostagno [b], G.E. Pazos [b]

[a] Cátedra de Ecología General, FCEFyN, Universidad Nacional de Córdoba – CONICET, Vélez Sarsfield 229, CP 5000 Córdoba, Argentina
[b] Unidad de Investigación Ecología Terrestre, CENPAT – CONICET, Boulevard Brown 2825, CP 9120 Puerto Madryn, Chubut, Argentina

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In grazed semiarid ecosystems, considerable spatial variability in soil infiltration exists as a result of vegetation and soil patchiness. Despite widespread recognition that important interactions and feedbacks occur between vegetation, runoff and erosion, currently there is only limited quantitative information on the control mechanisms that lead to differences in infiltration from different vegetation types. In this paper, we determine (i) the relationship between vegetation and soil surface characteristics and (ii) the soil infiltration rate 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 vegetation and soil infiltration are closely coupled. Total infiltration was significantly higher in the GS (69.6 mm) compared with the DGS and DSS (42.9 and 28.5 mm, respectively). In the GS, soil infiltration rate declined more slowly than the others communities, reaching a terminal infiltration rate significantly greater (57.7 mm) than those of DGS and DSS (25.7 and 12.9 mm, respectively). The high rate of water losses via overland-flow may limit the possibilities for grass seedling emergence and establishment and favor the persistent dominance of shrubs.

1. Introduction

In arid and semiarid landscapes, soil infiltration is recognized as a fundamental ecological process affecting not only the water budget of plant communities but also the amount of surface runoff and the attendant danger of erosion (Ludwig et al., 2005; Michaelides et al., 2009). In these environments, the amount of runoff and where it infiltrates are important determinants of vegetation patterns; conversely, vegetation patterns also directly modify the amount and spatial variability of infiltration (Arnau-Rosalén et al., 2008; van Schaik, 2009). These ecological and hydrological processes are tightly coupled and the complex ways in which they interact are the focus of the emerging field of ecohydrology (Wilcox et al., 2003; Newman et al., 2006).

Many studies in the arid and semiarid environments have widely documented that vegetation patterns affect the redistribution of water, sediment, seeds and nutrients within the landscape leading to further changes in the reorganization of the ecosystem structure (Schlesinger et al., 1990; Peters et al., 2005; Bestelmeyer et al., 2006). Despite widespread recognition that important interactions and feedbacks occur between vegetation, runoff and erosion over a range of scales (Scheffer et al., 2001; Wainwright et al., 2002; Peters et al., 2005), currently there is only limited quantitative information on the control mechanisms that lead to differences in water infiltration from different vegetation types (Michaelides et al., 2009).

In general, it is proposed that grazing disturbance, by changing vegetation and/or soil properties, can trigger persisting alterations in soil hydrology and eventually change a functional landscape that efficiently captures, retains, and utilizes water and nutrients into a dysfunctional one that no longer can efficiently capture these resources (Bestelmeyer et al., 2004; Briske et al., 2005). For example, in grazed semiarid rangelands of Patagonia, when grasslands degrade into shrublands, the soil provides less opportunity for water retention, allowing accelerated erosion rates (Rostagno, 1989; Parizek et al., 2002; Chartier and Rostagno, 2006). These changes negatively affect soil quality because the removal of fine soil particles and litter by erosion reduces, in turn, organic matter and nutrient concentration in the soil (Palis et al., 1990; Schiettecatte et al., 2008). This reduced nutrient availability, along with reduced soil seed bank and degraded soil physical conditions, limit plant growth and establishment, hindering the regeneration of the vegetation matrix (Bisigato and Bertiller, 2004). Recent

* Corresponding author. Tel.: +54 (0) 2965 451024; fax: +54 (0) 2965 451543.
E-mail addresses: marcechartier@yahoo.com.ar (M.P. Chartier), rostagno@cenpat.edu.ar (C.M. Rostagno), gpazos@cenpat.edu.ar (G.E. Pazos).

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conceptual advances in community and landscape ecology suggest that severe disturbance triggers a positive plant-soil feedback that limits the potential return of runoff and erosion to predisturbance levels, which, in turn, favors the further rangeland desertification (Scheffer et al., 2001; van de Koppel et al., 2002; Briske et al., 2008).

Differing spatial pattern changes associated with soil hydrologic deterioration mechanisms suggest that different monitoring strategies and interpretations are required to detect vegetation and soil changes in arid and semiarid lands. Understanding of the complex interactions between ecology and hydrology is essential to effectively address landscape change resulting from climate change and land use (Ludwig et al., 2005; Newman et al., 2006). The objective of this study was to assess the spatial variability in the infiltration rate and its relationship with vegetation and soil surface characteristics in a grazed rangeland of northeastern Patagonia. We use this information to discuss range management implications for sustainable land use of these semiarid ecosystems.

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).

We selected two contiguous ecological sites: a pediment-like plateau and a flank pediment. Beeskow et al. (1987) described a pediment-like plateau, locally called "mesetas" or plateaus, 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 covered by a gravel mantle and a lower zone with a base level controlled by a playa lake. The dominant soil in the study area 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%.

In the study area, the vegetation cover varies from 40 to 60% and presents a patchy structure where three discrete plant associations or community units (Whittaker, 1975) are clearly recognizable: i) a grass steppe with scattered shrubs (GS), ii) a degraded grass steppe with scattered shrubs (DGS), and iii) a degraded shrub steppe (DSS). These communities correspond to three states or stages of range degradation, identified along a gradient of grazing intensity (Beeskow et al., 1995). Grass dominated steppe represents the most desirable state in terms of livestock production and soil stability, while shrub steppe represents the most degraded and least productive state.

In the grass with scattered shrub steppe, the perennial grasses *Nassella tenuis* (Phil.) Barkworth and *Piptochaetium napostaense* (Speg.) Hackel ap Stuckert are the dominant species, while *Mulinum spinosum* (Cav.) Pers. is the dominant shrub. In the shrub steppe *Chuquiraga avellanedae* Lorentz is the dominant shrub species but isolate patches of *Nassauvia fuegiana* (Speg.) Cabrera are present. 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}) in paddocks commonly exceeding 2500 ha in size (Parizek et al., 2002).

2.2. Experimental procedure

During spring of 2003 and 2004, we randomly selected homogeneous vegetation patches at both ecological sites: four in GS, seven in DGS, and four in DSS (60 plots in total). In the DGS the number of patches was incremented due to the greatest surface occupied by this community respect to the others. Inside each selected patch, the infiltration rate was estimated using experimental plots measuring 0.60 × 1.67 m (1 m^{-2}), which 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 across the two ecological sites 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. In the study area, high-intensity rainfall can occur from December to March. A rainfall event with the intensity and duration of the simulated rainfall has a return period of 100 years in northeastern Patagonia (Vicenty et al., 1984). Runoff was collected at 5 min intervals in separate containers and determined by volume. Infiltration rate was calculated as the difference between the applied rainfall and the runoff collected for each interval. Time-to-ponding and time-to-runoff were recorded for each plot. Ponding was arbitrarily considered to be reached when approximately 10 per cent of the surface had attained this state.

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 determined. The A horizon thickness was determined by the depth to the Bt horizon in a pit opened adjacent to each plot. Undisturbed soil core samples were taken at the 0–5 cm depth adjacent to each plot for bulk density estimation (Blake, 1982). Soil samples from this same depth were collected and analyzed for texture by the hydrometer method (Bouyoucos, 1965), organic carbon content by wet combustion (Nelson and Sommers, 1982), and field capacity by centrifuging saturated soil samples (10 min; 2440 rpm). A particle density of 2.65 g cm^{-3} was used to calculate the total soil porosity (Blake, 1982).

2.4. Statistical analysis

A principal component analysis (PCA) was performed to identify associated vegetation and soil surface characteristics using the matrix of binary correlations between variables. For ordination of rangeland plant communities according to these characteristics, we further calculated the loading coefficients of each runoff plot on the first two principal components (Norusis, 1997).

Differences in total infiltration and terminal infiltration rate (at 25–30 min time interval) among plant communities, between ecological sites (pediment-like plateau versus flank pediment), and between study year (2003 versus 2004) were assessed by a three-way ANOVA. Mean separation with the Fisher's protected LSD test was used (Sokal and Rohlf, 1981). Furthermore, non-linear regression analysis was employed to determine the relationship between infiltration and soil degradation gradient. The independent variable measured for use in regression analysis was the first principal component of the PCA analysis. Soil infiltration rate was used as dependent variable. Significance levels were determined at $P \leq 0.05$.

3. Results

3.1. Soil and soil surface characteristics

The three plant communities presented contrasting soil and soil surface characteristics (Table 1). The GS had greater cover of perennial grasses and litter than the DGS and DSS communities. In contrast, the values of gravel and bare soil cover, the distance between perennial grasses, and the diameter of the largest bare soil patches increased from GS to DSS. A finer soil texture and decreasing amounts and size of perennial grass plants was recorded from GS to DSS. Field capacity was lower, and organic carbon content, total porosity, and A horizon thickness were higher in the GS as compared with the shrub interspaces of the DSS.

The PCA axis 1 and 2 explained 43% and 21% of the total variance in soil and soil surface characteristics of the runoff plots, respectively ($n = 60$) (Fig. 1). PCA axis 1 discriminated the three plant communities through gravel cover, distance between perennial grasses, clay content, and field capacity (positively correlated to DSS) and perennial grass cover, litter cover, sand content, and A hz thickness (positively correlated to GS) (Fig. 1; Table 2). PCA axis 1 can be interpreted as a soil degradation gradient from resource-conserving GS community to soil degraded condition in the DSS. PCA axis 2 discriminated GS from DGS mainly through soil physical characteristics (Table 2), with field capacity, total porosity and organic carbon content associated with GS (Fig. 1).

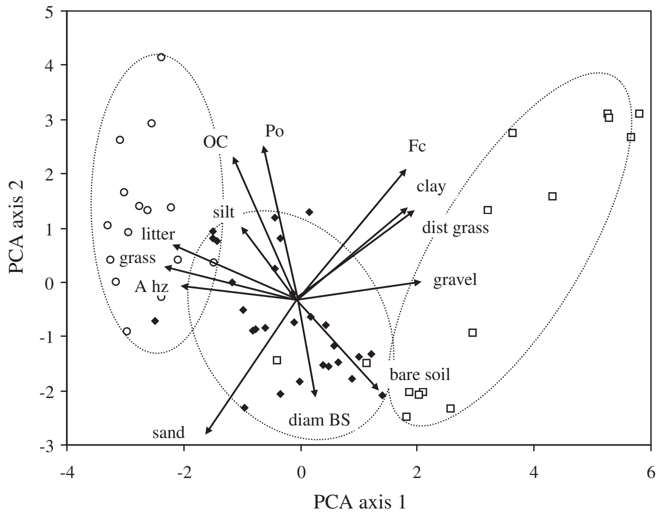

Fig. 1. Multivariate ordination of runoff plots ($n = 60$) according to their soil and soil surface characteristics (arrows) (see acronyms in Table 2). PCA axes 1 and 2 explained 43% and 21% of the total variance, respectively. Runoff plots in different plant communities: grass (○), degraded grass (◆), and degraded shrub steppes (□) are encircled by dotted lines.

3.2. Soil infiltration

Total soil infiltration recorded after 30 min of simulated rainfall was significantly different among plant communities ($F_{2,48} = 72.80$, $P < 0.0001$, $n = 60$) (Fig. 2A). The highest infiltration was recorded in the GS (69.6 mm) and decreased from this community to DGS and DSS (42.9 and 28.5 mm, respectively). Total soil infiltration did not show significant differences between both ecological sites ($F_{1,48} = 0.12$, $P > 0.73$, $n = 60$) and between years ($F_{1,48} = 0.18$, $P > 0.67$, $n = 60$).

Soil infiltration curves were similar in the DGS and DSS, and they were lower than in the GS (Fig. 2B). In the GS, soil infiltration rate declined more slowly than the others communities, reaching a terminal infiltration rate significantly greater (57.7 mm) than those of DGS and DSS (25.7 and 12.9 mm, respectively) ($F_{2,57} = 61.48$, $P < 0.0001$, $n = 60$). In these two later communities, the infiltration curves were similar after 15 min of the rainfall initiation (Fig. 2B). Time-to-ponding was significantly different among plant communities, recording 2.03 (S.E. 0.07), 1.38 (S.E. 0.08), and 1.04 (S.E. 0.04) minutes in the GS, DGS, and DSS respectively ($F_{2,57} = 33.60$, $P < 0.0001$, $n = 60$). Plant communities also differed in time-to-runoff starting at 4.23 (S.E. 0.31), 3.50 (S.E. 0.25), and 2.39 (S.E. 0.22) minutes after rainfall initiation in the GS, DGS, and DSS respectively ($F_{2,57} = 9.81$, $P < 0.001$, $n = 60$).

3.3. Relationship between infiltration and soil degradation gradient

Non-linear regression analysis demonstrated a negative exponential relationship between infiltration rate and the soil degradation gradient, represented by PCA axis 1:

Infiltration rate (mm h^{-1}) = 42.97 (S.E. 1.45) exp −0.16 (S.E. 0.01) PCA axis 1; $R^2 = 0.59$, $P < 0.0001$, $n = 60$ (Fig. 3).

4. Discussion

4.1. Relationship between vegetation and soil surface characteristics

In arid and semiarid ecosystems, vegetation and soil surface characteristics are spatially heterogeneous and variable in time (Mergen et al., 2001; van Schaik, 2009). Accordingly, in the

Table 1
Mean ± 1 S.E. values ($n = 60$) of soil and soil surface characteristics for the grass (GS), degraded grass (DGS), and degraded shrub steppe (DSS).

	Rangeland plant communities		
	GS	DGS	DSS
Perennial grass cover (%)	46.8 ± 2.5	17.9 ± 1.5	6.7 ± 1.9
Litter cover (%)	36.7 ± 2.7	21.3 ± 2.5	5.5 ± 2.2
Gravel cover (%)	0.3 ± 0.1	6.9 ± 1.5	44.4 ± 3.4
Bare soil cover (%)	16.4 ± 2.4	53.8 ± 2.9	43.4 ± 2.0
Distance between perennial grasses (cm)	9.6 ± 0.9	25.7 ± 1.7	83.8 ± 5.9
Diameter of the largest bare soil patch (cm)	9.3 ± 1.3	20.5 ± 2.7	16.3 ± 4.7
Sand (%)	73.0 ± 1.0	71.9 ± 0.7	66.1 ± 1.2
Silt (%)	21.0 ± 0.7	19.2 ± 0.5	18.4 ± 0.9
Clay (%)	6.1 ± 0.8	8.8 ± 0.7	15.5 ± 2.4
Organic carbon (%)	1.0 ± 0.05	0.8 ± 0.03	0.7 ± 0.05
Total porosity (%)	53.8 ± 0.8	49.3 ± 0.6	49.3 ± 1.2
Field capacity (%)	18.1 ± 0.8	19.4 ± 0.5	28.1 ± 1.1
A horizon thickness (cm)	26.6 ± 1.7	15.7 ± 2.0	4.2 ± 0.8

Table 2
Correlation coefficients (r) between soil and soil surface characteristics in the runoff plots ($n = 60$) and the first two PCA axes (PCA axis 1: 43% and PCA axis 2: 21% of the total variance).

Variables	Acronyms	Axis 1	Axis 2
Perennial grass (%)	Grass	−0.36	0.15
Litter (%)	Litter	−0.33	0.21
Gravel (%)	Gravel	0.36	−0.01
Bare soil (%)	Bare soil	0.22	−0.29
Distance between perennial grasses (cm)	Dist grass	0.36	0.20
Diameter of the largest bare soil patch (cm)	Diam BS	0.04	−0.29
Sand (%)	Sand	−0.31	−0.36
Silt (%)	Silt	−0.14	0.25
Clay (%)	Clay	0.34	0.19
Organic carbon (%)	OC	−0.14	0.44
Total porosity (%)	Po	−0.06	0.45
Field capacity (%)	Fc	0.34	0.31
A horizon thickness (cm)	A hz	−0.31	0.09

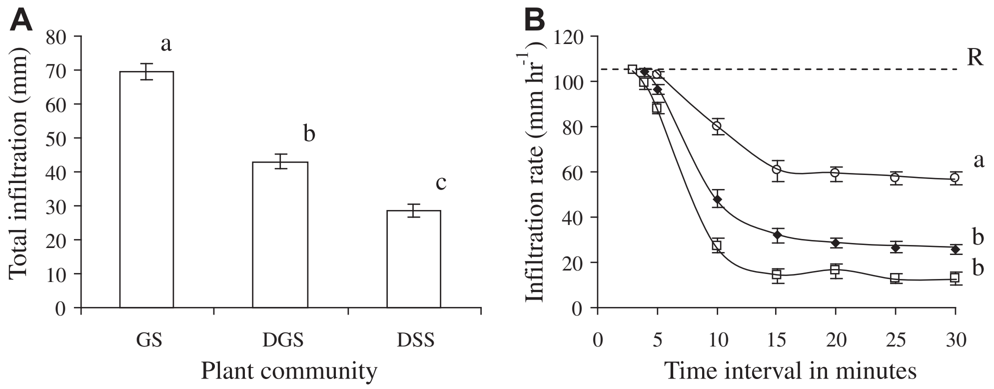

Fig. 2. Mean ± 1 S.E. values ($n = 60$) of (A) total infiltration and (B) infiltration rate at each 5-min interval across the 30 min of simulated rainfall for each plant community: GS grass (○), DGS degraded grass (◆), and DSS degraded shrub steppe (□). R is the applied rainfall intensity. Different lower case letters indicate significant differences ($P \leq 0.05$) among plant communities in total infiltration and terminal infiltration rate at the 25–30 min interval.

semiarid lands of northeastern Patagonia, we recognized a mosaic of patches of three plant communities co-occurring in the study area: grass with scattered shrubs (GS), degraded grass with scattered shrubs (DGS), and degraded shrub steppes (DSS). Our results showed that variation in plant composition and soil surface characteristics are closely related in this ecosystem (Table 1 and Fig. 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. In addition, the soils of these plant communities varied greatly in terms of soil physical and chemical properties. Soil texture composition, total porosity, field capacity, and organic carbon were more favorable, in terms of soil hydrologic condition, in the GS and DGS as compared with the DSS. These results agree with others showing that in arid and semiarid environments, vegetation plays a significant role in controlling the soil physical and chemical properties, which in turn determine the plant composition (Dunkerley, 2002; Ludwig et al., 2005).

Besides these physical and chemical properties, other soil properties (total nitrogen, ammonium, nitrates) can also contribute to fully characterize the impact of soil degradation on soil quality. In a previous work in the same study area, Rostagno and Beeskow (2000) found that degraded soils were characterized by lower soil organic matter and total nitrogen contents. A subsequent study (Videla et al., 2004) confirmed this finding. These authors found that total nitrogen decreased, in average, from 0.95 g kg^{-1} in GS to 0.69 g kg^{-1} in DSS, with DGS showing an intermediate value. Indeed, total nitrogen was highly correlated with organic carbon, as has been extensively reported for other soils elsewhere (e.g., Tongway et al., 2003; Carrera et al., 2005; Bai et al., 2005). Thus, the variation in total nitrogen across plant communities found in these studies encompass our results on the pattern of variation of organic carbon, confirming a decrease in soil quality with increasing degradation. Our results are 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 (Palis et al., 1990; Schiettecatte et al., 2008). It is also expected that inorganic carbon mirror the pattern of variation of total nitrogen, although the relative concentration of ammonium and nitrate are highly variable in time in arid regions depending on the occurrence of rain pulses (Mazzarino et al., 1998).

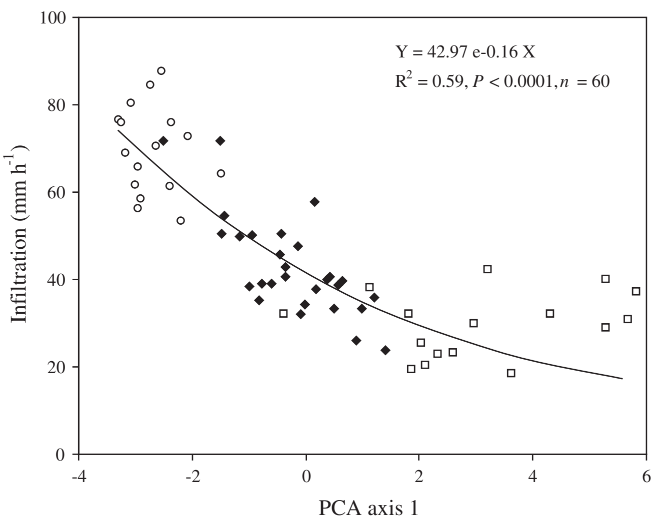

Fig. 3. Relationship between infiltration and soil degradation gradient, represented by PCA axis 1. Symbols indicate different plant communities: grass (○), degraded grass (◆), and degraded shrub steppe (□).

Changes in vegetation and soil surface characteristics may be directly produced by grazing or indirectly following a positive feedback mechanism including cover reduction, soil erosion and losses of soil organic matter, nutrients, and seeds from the upper soil in the bare soil patches (van de Koppel et al., 2002; Newman et al., 2006; Michaelides et al., 2009). At the landscape scale, the extensive replacement of these grasslands by arid shrublands, characterized by a more widely spaced resource accumulation points (Schlesinger et al., 1990; Bisigato and Bertiller, 2004), has resulted in desertification through increased spatial heterogeneity of soil resources (Chartier and Rostagno, 2006; Bisigato and Lopez Laphitz, 2009).

4.2. Vegetation and soil surface characteristics as determinants of infiltration rate

As has been largely documented in arid and semiarid ecosystems, the type and spatial distribution of vegetation have spatial influence on soil surface characteristics and infiltration (Mergen et al., 2001; van Schaik, 2009). Accordingly, our field results clearly showed that infiltration rates are not uniform within the Punta Ninfas rangelands. The transformation of grassland to degraded grassland to shrubland resulted in progressively lower quantities of soil infiltration (Fig. 2). Soil protection factors, mainly cover of perennial grasses and litter, affect infiltration/runoff and soil detachment by raindrops (Chartier and Rostagno, 2006). However, structural components of vegetation, such as plant height, density, canopy characteristics, etc. can also affect the amount of rainfall intercepted (Weltz et al., 1998).

Both perennial grass and litter cover were the highest in the GS community. The high herbaceous plant density, as indicated by the short distance between perennial grasses (Table 1), usually confers an interwoven made by crisscrossing strips of leaves and litter material creating barriers or obstacles that hold back the runoff water flow (Chartier and Rostagno, 2006). Indeed, the larger perennial tussocks grasses produce a marked microtopography with high surface water retention capacity in small closed micro-basins that form in the plant interspaces (Davenport et al., 1998). This high surface water retention/interception increase both the time-to-ponding and the time-to-runoff and, consequently, also increase the likelihood that the water will infiltrate rather than runoff into the surrounding bare intershrub area (Fig. 2B) (Herrick et al., 2005). Eventually, a reduction in perennial grass cover and density and an increase of size of bare soil patches (Fig. 1) will lead to a more degraded DGS community with significantly reduced soil infiltration rates (Figs. 2,3).

On the contrary, perennial grass cover in the DSS communities was about 85% lesser than that of GS (Table 1). Indeed, it was concentrated in small patches in the shrub interspaces, as evidenced by the large distance between perennial grasses and size of the bare soil patches. Under these conditions, there is an increase in the connectivity of intercanopy areas (Davenport et al., 1998; Parizek et al., 2002; Chartier and Rostagno, 2006). Runoff is concentrated downslope in the intershrub areas, increasing the velocity of the flow and exceeding the critical threshold for sediment transport (Michaelides et al., 2009). The higher energy of the runoff will accelerate erosion, deflating the intercanopy areas, which further diminishes the opportunities for canopy patches to capture runoff (Puigdefábregas, 2005; Arnau-Rosalén et al., 2008). In fact, the size and connectivity of the bare soil patches may be more important than the absolute amount of bare soil in determining potential runoff and soil erosion rates (de Soyza et al., 2000). Accordingly, de Soyza et al. (1998) defined a bare patch index based on the mean size of bare patches and proportion of bare soil as sensitive early warning indicators of desertification for the Chihuahuan desert. Thus, not only cover but structural and spatial components of vegetation can explain the variation of infiltration rates observed across our study area.

On the other hand, the magnitude of soil texture variability resulting from different rangeland plant communities was also associated with different infiltration rates (Table 1 and Fig. 2). Thus, compared with the DSS, the greater infiltration in the GS might also be caused by coarser soil texture, which prevents soil compaction by trampling and provide greater hydraulic conductivity (Rostagno et al., 1991). In a sandy soil, most of the pores are relatively large. Oppositely, soils enriched in clay content, mainly smectitic (Bouza et al., 2007), are associated with lower soil infiltration rate. Clay-rich soils were locally found in the *C. avellanedae* shrub interspaces of the DSS, where the A horizon has been largely striped by erosion and the underlying restrictive argillic horizon exposed (Rostagno, 1989; Chartier and Rostagno, 2006). When the argillic horizon is close to soil surface, it is generally associated to desert pavements where vesicular layers, sedimentary crusts, and surface clay seals are present (Súnico et al., 1996; Parizek et al., 2002). The low hydraulic conductivity of argillic horizons as well as clay seals and vesicular layers has a direct impact on the infiltration rate (Rostagno et al., 1991).

Our results provide evidences that support the hypothesis that vegetation exerts a first-order control on infiltration/runoff dynamics through direct interactions with soil quality and surface characteristics. Perennial grass cover was found to be strongly related to soil infiltration process (Fig. 3). Vegetation stabilizes the soil aggregates against erosion as has been well documented elsewhere (Ludwig et al., 2005; Michaelides et al., 2009), and it is also associated with surface accumulation of biologically mediated elements, such as carbon and nitrogen (Tongway et al., 2003). In this sense, the net effect of perennial grasses will be complex, and will include the soil stabilization and nutrient conservation and the positive effect on water infiltration.

In degraded areas, the water availability and soil seed bank are the main limiting factors of seedling recruitment of perennial grasses (Bisigato and Bertiller, 2004). Moreover, the cumulative effect of accelerated soil erosion may result in enough soil loss to trigger irreversible changes in hydrologic function and prompt the transition from the DGS to the DSS community (Chartier and Rostagno, 2006). In this context, the DGS may be considered as an at-risk community defined by the increased vulnerability to catastrophic erosion (Scheffer et al., 2001; van de Koppel et al., 2002). Commonly, the transition to a degraded state is triggered by a high-intensity rainfall event affecting a community phase characterized by low perennial grass cover and large patches of bare soil (Stringham et al., 2003; Briske et al., 2008). In water-limited ecosystems, understanding of vegetation dynamics associated with the main hydrological processes (e.g., infiltration and runoff production) is crucial if we are to effectively address landscape change resulting from climate change and land use (Wilcox et al., 2003; Ludwig et al., 2005).

5. Conclusions

Our results illustrate that vegetation and soil infiltration are closely coupled, and grazing management affects both ecological and hydrological processes as well as their interactions. In the Punta Ninfas range site, vegetation has a first-order control on the local soil quality, and hence on the hydrological behavior. In particular, while the GS represents a resource-conserving plant community, desirable for both forage production and soil protection, the DSS represents an undesirable community with degraded soil hydrologic properties and low value for animal production. Alternatively, the DGS represents an unstable and transitional community that, without management intervention to halt soil erosion, will likely change into the DSS.

Results from this study indicate that changes over time in the relative proportion as well as in the spatial pattern of plant communities may be early warning indicators of the potential for soil hydrological deterioration. Since rangeland sustainability depends primarily on conservation of the soil and water resources, the fundamental goal of range management should be the maintenance of the integrity of the main ecological processes involved of these semiarid ecosystems.

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