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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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A B S T R A C T

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.

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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 $P \leq 0.05$.

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