ORIGINAL PAPER

Structure of vegetation patches in northwestern Patagonia, Argentina

C. A. Busso · G. L. Bonvissuto

Abstract Vegetation of arid and semiarid environments has in general a patchy distribution. Our objective was to (a) determine several qualitative and quantitative analytical characteristics of vegetation patches in an arid zone of Patagonia, Argentina, and (b) investigate relationships between them. Annual precipitation in this area was 200 mm during 1999–2005. Eight transects involving ten patches each were studied within a 15 × 15 km area. Mean (±1 SE) values ($n = 80$) in the vegetation patches were 315 ± 25 and 207 ± 16 cm for the greatest and lowest patch diameter, respectively; 23 ± 2 cm for mound height; 113 ± 12 cm for maximum vegetation height; and 170 ± 18 cm and $58 \pm 2\%$ for distance to the next vegetation patch and vegetation patch cover within a transect, respectively. Correlations between greatest and lowest diameters, mound height, maximum plant height and distance to the closest vegetation patch were all significant ($P < 0.01$; $n = 80$). In all vegetation patches, the greatest and lowest frequencies were found for the grass *Stipa* spp. (71.2%) and the shrub *Grindelia chiloensis* (Cornel.) Cabrera (12.5%). *Stipa* spp. and the shrub *Atriplex lampa* (Moq.) D. Dietr. showed the highest simultaneous frequency (50%). A reasonable association among species (>45%) was found for *Stipa* spp., *Atriplex lampa* and the shrubs *Larrea divaricata* Cav., *Lycium chilense* Miers ex Bertero and *Junellia ligustrina* (Lag.) Moldenke. *Larrea divaricata* and *Atriplex lampa* contributed more than 84% of the total patch standing crop ($5,777 \pm 435$ g). Average patch size and specific diversity were 5.93 ± 0.33 m^2 and 1.31 ± 0.11, respectively. Aboveground standing crop of the two dominant shrubs decreased as plant species diversity increased ($P < 0.05$). Conservation of vegetation patches is crucial to prevent increased soil erosion and desertification in the study ecological system.

Keywords Arid and semiarid lands · Perennial grasses and shrubs · Qualitative and quantitative analytical plant characteristics · Regression analysis

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Introduction

The occurrence of patches of vegetation alternating with patches of (almost) bare soil is a common feature in arid and semiarid grazing systems throughout the world (Ludwig et al. 2004; Erickson et al. 2005). These systems cover nearly 30% of the earth's land surface. Vegetation patches vary in form, from irregular mosaics to regular stands, and size, from one to several hundred meters (Montaña 1992; Haase et al. 1996).

In environmentally harsh sites and in sites where levels of resources are chronically low, such as in Monte Austral Neuquino (Patagonia, Argentina), conditions suitable for rapid plant growth may occur infrequently. Natural selection under these conditions would favor adaptations conferring tolerance to prevailing forms of abiotic stress and the ability to conserve resources that have been acquired (Pyke and Archer 1991). Shrubs encompass an array of morphological and physiological traits that can contribute to vertical stratification of resources with grasses and forbs. These lifeforms are often considered undesirable on rangelands because they are presumed to reduce herbaceous production or because their presence increases the difficulty of livestock manipulation. Yet, in many regions or landscapes within a region, woody plants are well adapted to prevailing biotic and abiotic conditions, and can provide several ecological benefits. They play a key role in primary production and nutrient cycling while stabilizing soils; creating patches of fertility in bare, soil and vegetation degraded areas; allowing the presence of palatable, preferred grasses to domestic livestock in the plant community; providing essential temporary habitat for maintaining an animal metapopulation or for dispersing animals (Longland and Bateman 2002; Bonvissuto and Busso 2006, 2007a).

Quantifying vegetation structure is crucial to identifying wildlife habitat (Schulte and Niemi 1998; Sutter and Brigham 1998) and to characterize the spatial vegetation aggregation on the landscape (Schenk et al. 2003; Erickson et al. 2005). Many measures of vegetation structure have also been developed on rangelands as an index of plant species diversity (Connell 1978; Bates et al. 2000). Research to determine the floristic composition, quantitative plant measures and spatial arrangement of the vegetation patches in the Monte Austral Neuquino is then essential.

Vegetation patches are likely to enhance local biological diversity of plants and animals, because they harbor species that are lacking in disturbed areas, and because abrupt structural changes from disturbed to undisturbed vegetation provide a habitat mosaic that facilitates high levels of species turnover. Vegetation patches also provide more evenly dispersed seed sources for re-establishment of decimated vegetation within disturbed areas, and they may harbor animal species that provide seed dispersal services (Longland and Bateman 2002). Selective inclusion of woody plants in restoration plans offers several potential advantages including enhanced herbaceous production, reduced grazing pressure on grasses by providing a refuge for heavily utilized herbaceous species, enhanced soil nutrient status, mineralization and water infiltration (Pyke and Archer 1991).

Longland and Bateman (2002) reported that there are several species that are obligate associates of the shrub species that dominate undisturbed rangelands. These results are similar to those found by Morello (1956), Yeaton (1978) and West (1989), and include understory plant species that either utilize shrubs as nurse plants or are parasitic on them. Aarssen and Turkington (1985) demonstrated that niche differentiation and differential competitive abilities combine to establish distinct positive and negative associations between species as revegetated rangelands develop through time. For example, Schenk et al. (2003) found that negative effects of *Larrea tridentata* (Ses. et Moc ex DC.) Felger & Lowe on roots of *Ambrosia dumosa* (A. Gray ex Torr.) Payne kept plants of the latter

species segregated to a 3 m distance from those of *L. tridentata*. Research on associations between plant species are then important in the vegetation patches at the Monte Austral Neuquino.

Studies have demonstrated that succession toward woody species at the final stage, and community disturbance, can reduce plant species diversity. In central and eastern Oregon, northeastern California, and southwestern Idaho, succession to *Juniperus occidentalis* spp. *occidentalis* Hook. dominated communities has been accompanied by reductions in understory species diversity (Burkhardt and Tisdale 1969). Bates et al. (2000) also demonstrated that dominance by juniper in southeastern Oregon reduced understory diversity. They attributed this response to the high belowground interference for soil water and nitrogen conducted by juniper trees. Anyhow, these authors agree that it may take many years of woodland dominance before understory species are effectively removed from a site. Connell (1978) pointed out that plant species diversity begins to decline at some point following disturbance because either the most competitive plants eliminate the rest or the plants most resistant to damage by physical extremes or natural enemies eventually occupy most of the space.

Vegetation patches present a high both horizontal and vertical heterogeneity in the shrubby steppe of the Monte Austral of Chubut, Argentina. This is because of the large architectural diversity that there exists among its plant species (Bertiller et al. 2004). Biotic and abiotic characteristics have been described by Soriano et al. (1994) for vegetation patches in the Patagonian steppe. Similarly to that described for the Patagonian steppe, vegetation also has a patchy distribution in the shrubby steppe of the Monte Austral Neuquino, within the Province of Neuquén, Argentina. However, the structural characteristics of the dominant species in this ecological region have ever been studied, and are different from those reported for the Patagonian steppe.

Rangelands in the shrubby steppe of the Monte Austral Neuquino, surrounding the town of Picún Leufú (39°53′S, 69°25′W), have been exposed to continuous, uncontrolled grazing by goats, sheep and cattle since the early 1900s (Bonvissuto 2006). Forage production of these rangelands is often limited partially due to the scarce precipitation in these areas [long-term mean annual precipitation: 137.2 mm (1928–1950) (Bonvissuto 2006)]. Thus, these rangelands have been exposed to similar biotic and abiotic conditions for many decades (Bonvissuto 2006). It is then expected a similar structure and spatial arrangement of the vegetation in the study region, leaving plant-uncovered interspaces among vegetation patches which need immediate restoration efforts.

The shrubby steppe in the Monte Austral expands 3,816,025 ha. It is within the Phytogeographical Province of the Monte (Cabrera 1971). The main economical activity in this area is grazing by cattle, sheep and goats on natural vegetation. More than 90% of farmers live in their ranches, and for most of them, rangeland utilization represents their unique income. Eighty-six percent of the ranches have an economy just for subsistence. Seventy-eight percent of the farmers occupy federal lands, and just a few have a wire fence all around their ranch. The level of desertification in this highly disturbed system is from partial severe to severe (INTA-GTZ 1996). This urgent need for land reclamation and range improvement in the vegetation-uncovered areas has also been a concern in other rangelands of the world (Thorsteinsson et al. 1971). This paper will shed light on the structural characteristics and spatial arrangement, and relationships among species, of the vegetation patches in the Monte Austral Neuquino. It will also deal with some management and conservation issues to help preventing further vegetation and soil degradation of this ecological system.

Three hypotheses were tested in this research: (1) structural characteristics of the vegetation patches dominated by *Larrea divaricata* Cav. and *Atriplex lampa* (Moq.) D.

Dietr. are similar within the whole study site (15 × 15 km), leaving extensive bare, eroded areas among vegetation patches, (2) within each patch, some species are associated with one another, but not with others. Even more, species associations are in some cases positive and in others negative, and (3) as size of the dominant shrubs (expressed as aerial standing crop) increases, plant species diversity decreases. To test these hypotheses, we determined several qualitative and quantitative analytical characteristics of one sub-group of the vegetation patches in the shrubby steppe of the Monte Austral Neuquino, and investigated relationships between them.

Methodology

Study site

This research was conducted in the shrubby steppe of *Larrea divaricata* and *Atriplex lampa*, nearby the town of Picún Leufú, within the Monte Austral Neuquino (Fig. 1). Initially vegetation patches were sampled within a 15 × 15 km area (39°20–32′S; 69°19–30′W). Further, more detailed studies were conducted at two sites, distant 1.5 km straight line from one another.

Climate

In Picún Leufú, long-term (1928–1950) average annual precipitation is 137.2 mm (Ital-consult Argentina 1966), and average annual precipitation during 1999–2005 was 200 mm.
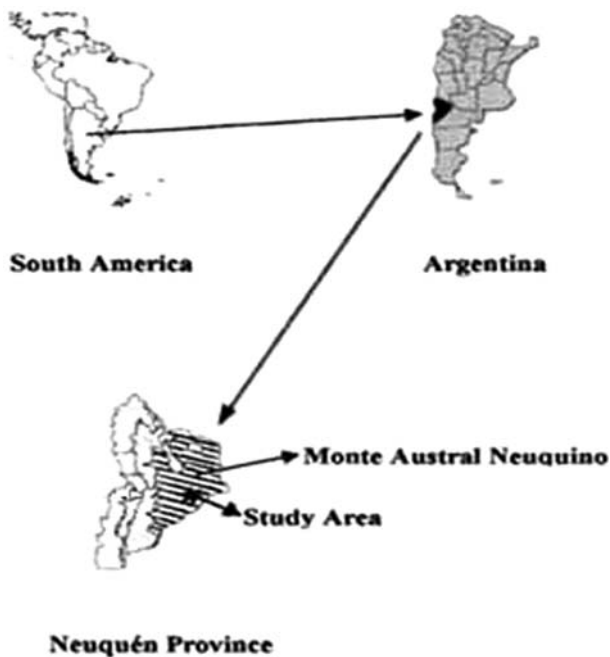

Fig. 1 Location of the study area in the Monte Austral Neuquino region and its relation to South America and Argentina

Extreme values of precipitation during 1928–1950 have been 62.7 and 245.3 mm (Morello 1958). No precipitation data are available at this site between 1950 and 1999. Maximum and minimum precipitations occur during winter and spring, and summer, respectively. Monthly average values for diurnal and nocturnal temperatures are 28 and 15°C in January, respectively, and 7 and 2°C in July, respectively. Mean annual wind speed is 13 km h^{-1} (Servicio Meteorológico Nacional 1958). November, December and January (mid-spring to early summer) have the greatest wind speeds, with a mean of 17 km h^{-1} at 2 m height from the soil level.

Vegetation

The shrubby steppe of the dominant *Larrea divaricata* and the co-dominant *Atriplex lampa* occupies ∼1,080,000 ha in the Monte Austral Neuquino. The average height of this steppe is 1.20 m. It is usually present either in plain areas or in those with very little slope (<5%) (Movia et al. 1982). Three strata can be found in this steppe (Movia et al. 1982): (1) *Larrea divaricata*, *Bougainvillea spinosa* (Cav.) Heimerl, *Monttea aphylla* (Miers) Bentham et Hooker and *Larrea cuneifolia* Cav. are present with a mean height of 1.50 m, but which can be reduced to 0.80 m on stony or fine-textured soils, (2) the second stratum varies between 0.70 and 1.00 m and is characterized by the presence of *Atriplex lampa* (0.5–0.8 m height). This species is present either on sandy mounds together with *Prosopis alpataco* Philippi, *Schinus polygamus* (Cav.) Cabr., *Lycium chilense* Miers (ex Bertero), *Chuquiraga erinacea* D. Don and *Fabiana peckii* Niederlein or alone in the interspaces, (3) small shrubs form the third stratum, which is 0.05–0.60 m height [*Acantholippia seriphioides* (A. Gray) Moldenke, *Gutierrezia solbrigii* Cabr., *Verbena seriphioides* Gilles & Hook, *Senecio aff. filaginoides* DC., *Cassia aphylla* Cav. and *Grindelia chiloensis* (Corn.) Cabr.] and perennial grasses [*Stipa neaei* (Nees ex Steudel), *Poa lanuginosa* (Poiret ap. Lamarck), *Poa ligularis* (Nees ap. Steudel), *Stipa speciosa* (Trinius et Rupretch), and *Elymus erianthus* Philippi]. There are ephemeral species (*Schismus barbatus* Thellung and *Plantago patagonica* Jacquin) 0.05 m height which do not have a plant cover greater than 15%.

Soil

Aridisols dominate the area. There are mostly old structural plains. Landscape is controlled for the horizontal disposition of the subjacent mantle. It includes Typic Paleortids, Petrocalcic Calciortids, Petrocalcic Paleargids and Typic Torriorthents, with shallow and saline phases (Ferrer and Irisarri 1989). According to the soil classification (Soil Survey Staff 1975) no water is available during long periods. A light tendency to the Mediterranean type can be observed, with greater humidity during the winter (Morello 1958). When temperature is sufficient for plant growth, soil water potentials are ≤−1.5 MPa most of the time. During the warm season, a continuous period of three of more months with available moisture does not exist (Soil Survey Staff 1975). In addition, the scarce rain has a low infiltration, especially in the corridors between vegetation patches, and most of it is loss by runoff (Bonvissuto, personal observation, EEA INTA Bariloche, Argentina).

Soil profile measurements

In each of the two study sites, a vegetation patch was selected to (a) analyze the soil profile and (b) to determine soil physico-chemical characteristics below plants of *Larrea divaricata*. There were four different locations within each patch: (1) windward, (2) below
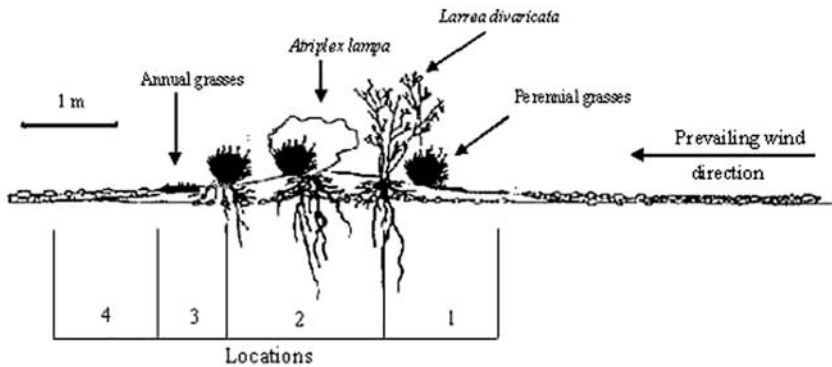
Fig. 2 Vegetation patch description. The position of annual and perennial grasses, *Larrea divaricata* and *Atriplex lampa* is indicated in the patch. The prevailing wind direction comes from the side nearby location 1

Larrea divaricata, (3) under the lee, and (4) in the bare corridors among vegetation patches (Fig. 2). These four study locations have a wind-related ontogeny.

The soil profile and physico-chemical characteristics were described determining the following parameters: (1) texture [pipette method (INTA EEA Pergamino 1998) and the Soil Survey Staff (1975) classification was applied]; (2) structure and hydric constants (Black et al. 1965); (3) bulk density (INTA EEA Pergamino 1998); (4) organic matter (Walkley and Black 1934); (5) infiltration (Lyford and Qashu 1969); (6) C/N ratio; (7) total nitrogen (Kjeldahl method: Carter 1993); (8) extractable K using a spectrophotometer of atomic absorption by flame emission (Carter 1993); (9) soluble P in sodium bicarbonate (Olsen and Sommers 1982) and (10) pH in water (in suspension with a relationship 1:2.5: SSSA 1996). In the same suspension (1:2.5) the following were evaluated; (11) electric conductivity using a conductimeter Schott Gerate; (12) percentage of interchangeable sodium (USDA 1954); (13) carbonates (gasovolumetric method); and (14) initial soil moisture content (Lascano and Landivar 1997). Infiltration versus initial moisture content measurements were conducted on 6 October and 7 November 2000, and 7 February 2001.

Soil profile and physico-chemical characteristics at sites 1 and 2 are summarized in Table 1. At site 1, shallow horizons were generally clear and had a soft consistency when they were dry. A high stoniness was observed in the shallower horizons in the soil profile. The regime of soil moisture was aridic and that of temperature was mesic. The whole profile reacted to HCl indicating carbonate presence. A layer of broken sand with abundant fissures was observed in the R horizon. It was a water-insoluble material, covered in part with gypsum salts that, when submerged in water, presented a greater solubility.

In the first 5 cm of the soil profile, soil moisture retained at -1.5 MPa was $5.46 \pm 0.18\%$ (mean $\pm$ 1 SE) on average (4 vegetation patches $\times$ 4 locations within each vegetation patch, $n = 16$). Soil moisture ranged from 0.8 ± 0.1 (March 1999; locations 1, 3 and 4) to $14.6 \pm 0.3\%$ (June 2000; location 2).

Vegetation patches

Eight transects of variable length (Fig. 3; Canfield 1941) were randomly located in the 15×15 km research site, within which the two study sites were later selected. Each

Table 1 Soil physico-chemical properties at sites 1 and 2 in the study area

Soil physicochemical properties	Site 1	Site 2
Horizon characteristics with increasing soil depth	4-10 cm = eolic accumulation 25-30 cm = A$_1$/AC horizon 30-35 cm = Cca (calcic horizon) 35-70 cm = R horizon (very dense, consolidated material)	
Texture (0-5 cm soil depth)	84.5% coarse material 15.5% silty material	
Bulk density (g/cm^3)	1.16±0.04*	
Organic matter (%)	Surface soil = 1.00 30 cm soil depth = 0.75	Surface soil = 1.40 30 cm soil depth = 1.20
Infiltration** (0-5 cm soil depth)	Date 6 Oct 2000 Infiltration rate (mm/h) Initial soil moisture content (%) 7 Nov 2000 Infiltration rate (mm/h) Initial soil moisture content (%) 7 Feb 2001 Infiltration rate (mm/h) Initial soil moisture content (%)	Location 1 2 3 4 158 214 41 13 8.5 9.8 12.4 15.4 631 641 288 28 1.8 2.3 3.4 5.5 843 797 164 23 1.3 1.2 1.2 2.6
C/N ratio (surface soil)	14	11/12
Total N (%) (surface soil)	0.04	0.07
Extractable K (ppm) (0-5 cm soil depth)	389.7 ± 17.5***	
Soluble P (ppm) (0-5 cm soil depth)	10.8 ± 1.3***	
pH	Surface soil = 7.8 A$_1$/AC horizon = 8.3	Surface soil = 7.7 30 cm soil depth = 8.9
Electric conductivity (mmho/cm)	Mostly 0.2-0.4 Calcic horizon = 4.1	0.2-0.3 in the whole profile
Percentage interchangeable Na (me/100 g soil)	Above the R horizon = 3 Below the R horizon = 44	Above the R horizon = 3 Below the R horizon = 39
Carbonate equivalents (%)	Surface soil = 1.9 30 cm soil depth = 3.3 Below the R horizon = 13.6	Surface soil = 5.0 30 cm soil depth = 2.5 Below the R horizon = 6.9
Cation exchange capacity (me/100 g soil)	11-13	12

* Mean ± 1 SE, $n = 10$; ** infiltration and soil moisture content were only evaluated at site 1; *** mean ± 1 SE, $n = 6$

transect intercepted ten vegetation patches (8 transects × 10 vegetation patches/transect = 80 vegetation patches) to characterize these patches throughout their species at the end of winter 1998. Transects were of different length because the distance between any two vegetation patches was unequal (Fig. 3). As many quantitative and qualitative variables as possible were measured to determine patch structure. Observations included: (a)
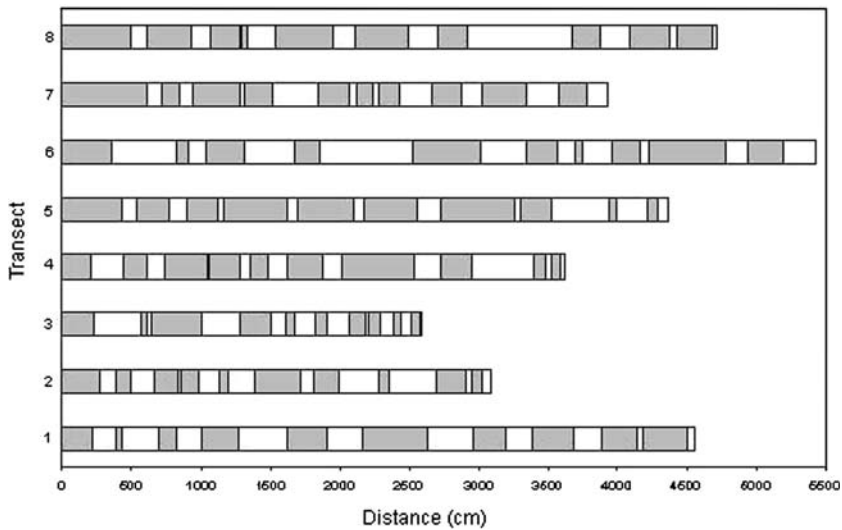
Fig. 3 Vegetation patch (*shaded areas*) distribution along each of eight transects (*horizontal bars, shaded + white areas*). Transects were of variable length (centimeters, *X* axis) as to include ten vegetation patches each. *White areas* represent bare, uncovered zones among vegetation patches

patch diagram where the interception within the transect was detailed, and orientation of the greatest patch diameter; (b) greatest and lowest mound diameter (mound refers to the rounded area of topographically higher elevation on any vegetation patch); (c) interception length of the vegetation patch with the transect; (d) mound height (height difference between the tallest part of the mound and the plant-uncovered corridors among mounds); (e) maximum vegetation height from the soil surface and name of the tallest species; (f) floristic composition of the species intercepted by the transect; (g) determination of lineal cover per species following Frangi (1978); and (h) distance to the next vegetation patch.

The relationship between plant species was studied to determine either positive or negative associations between them (Silvertown and Wilson 1994). Similarity between vegetation patches was also evaluated (Russell and Rao 1940).

Because structural variables measured in the vegetation patches were similar ($P > 0.05$) among the eight transects, it was decided to work with a smaller number of sites ($n = 2$) and various vegetation patches within each site ($n = 8$). Vegetation patches used for this research were excluded from domestic herbivory during the duration of this investigation, since the experimental study area was under grazing at that time. Herbivory exclusion was performed using 1.5 m height iron fences made of 15×15 cm quadrats. At the end of the study, the eight vegetation patches were harvested within each site (total $n = 16$ vegetation patches) to determine patch aerial standing crop. With this purpose, entire plants of each species were harvested at the soil surface level, dried in a forced oven at 60°C until constant weight, and weighed. Since the greatest and smallest diameters were measured within each of the sixteen vegetation patches, standing crop could be expressed per unit surface area.

Species diversity was calculated from the patch floristic composition data using the Shannon (1948) index: $H' = -\Sigma \ p_i \ \ln \ p_i$, where H' is the diversity index, p is the

proportional abundance of i species which is the number of individuals of i species divided by the total number of individuals in the community.

Statistical analysis

When eight transects involving ten vegetation patches each were studied, frequency histograms were made for longest and shortest diameters, relationship between them, mound height, maximum plant height and distance to the next vegetation patch. Correlations between these variables were calculated. An analysis of species grouping (Sorensen) was realized, and results were represented using a dendrogram. A one-way ANOVA was conducted in blocks [13 treatments (species) × 8 blocks (transects)] to analyse the relative frequency of species occurrence in the vegetation patches. Simultaneous frequency was analysed following Pielou (1977).

The Phi-Pearson index, utilized to determine either positive or negative associations between plant species, measures departure from independence (is the correlation coefficient for binary data). The greater the index value, the greater the association between species. This allowed to interprete if the existence of any given species (i.e., perennial grasses) was associated, at least in part, to the conditions which can provide other species (i.e., shrubs). Absence between any two species pairs was considered in the analysis; otherwise all correlations would be negative. It is true that an exaggerated double absence between any species pairs could determine high, spurious correlations. Because of this, all species with low frequency (<12.5%) were discarded to calculate the Phi-Pearson correlation index. The level of similarity at which the stems were cut in the dendrogram was 0.45 (close to half of the maximum possible association).

When 16 vegetation patches (8 patches × 2 sites) were harvested at the end of the study, a one-way ANOVA was conducted for the variables obtained at the two sites: total standing crop per patch, greatest diameter, lowest diameter, patch surface, standing crop per unit surface area and species diversity. Regression analysis was developed between species diversity and size of the dominant shrub/s (expressed as aerial standing crop) within each vegetation patch.

Results

Patch structure

This plant community was quite homogeneous. Structural characteristics of the vegetation patches were similar ($P > 0.05$) among transects in the whole study area (15 × 15 km). There was lower heterogeneity between than within transects. About 90% variability occurred within transects. When variation components were detected between sites, these were much lower than variation components between vegetation patches within each site. This justified a greater labour effort within each site.

The greatest and smallest diameter of the vegetation patches were 315 ± 25 and 207 ± 16 cm (Fig. 4; mean ± 1 SE, $n = 8$ transects of ten vegetation patches each: Fig. 3). Sixty-one out of the 80 vegetation patches had a greatest diameter between 140 and 540 cm, and 57 of these had a smallest diameter between 90 and 330 cm.

Shape of the vegetation patches, defined as the relationship between the smallest and the greatest diameters, was 66.8 ± 13.5 (Fig. 4). This relationship was between 0.5 and 0.9 in 68 of the vegetation patches (Fig. 4).
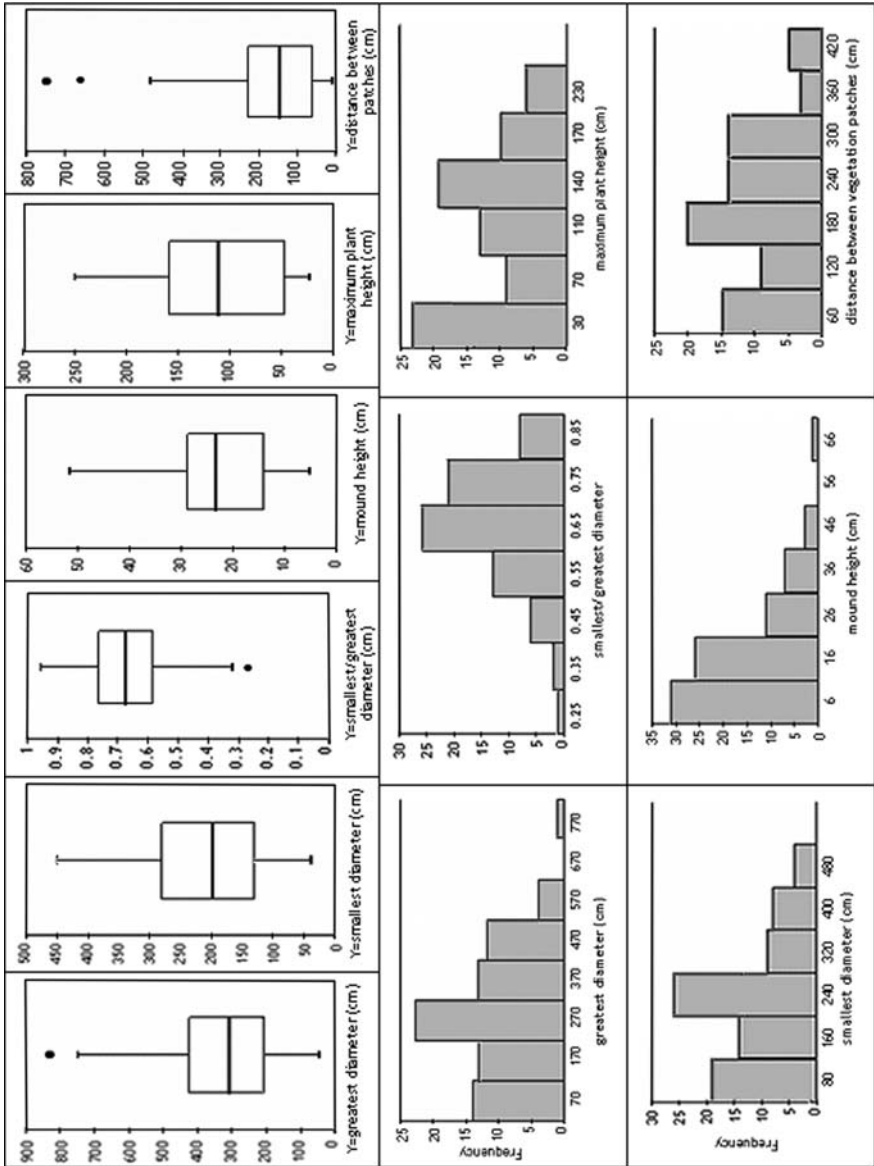
Fig. 4 *Upper panel set*: Diagram of boxes and arms for the greatest diameter, smallest diameter, the relationship smallest/greatest diameter, mound height, maximum plant height and distance to the next vegetation patch. The *X* axis was not drawn because there are no divisions on it. Within each *panel*, the *Y* variable is depicted at the panel's bottom. The *solid line* within each *box* represents the median. *Horizontal hyphens* at the *top* and *bottom* of *vertical lines* represent the maximum and minimum values, respectively. The lower Q1 and the upper Q3 quartiles are represented at the *bottom* and *top horizontal lines* within each *box*. This is where 50% of observations are found (25% between Q1 and the median and 25% between the median and Q3). *Vertical bars* indicate the distance between the minimum and maximum values. A point located outside this range represents an outlier. *Lower panel set*: Absolute frequency distribution of the greatest diameter, smallest diameter, the relationship smallest/greatest diameter, mound height, maximum plant height and distance to the next vegetation patch of vegetation patches (*n* = 8) in the Monte Austral Neuquino

Mound height was 23 ± 2 cm (Fig. 4). Height was between 0 and 36 cm in 68 out of the 80 mounds (Fig. 4). Maximum vegetation height was 113 ± 12 cm (Fig. 4), being the shrub *Larrea divaricata* the tallest species. Sixty-four of the vegetation patches had plants with a maximum height of up to 170 cm (Fig. 4). Distance to the next vegetation patch within a transect was 170 ± 18 cm (Fig. 4). Distance between vegetation patches was less than 310 cm in 68 of the patches (Fig. 4).

Vegetation patches covered $57.73 \pm 2.34\%$ of the transects ($n = 8$). *Larrea divaricata* showed the greatest percentage cover ($19.06 \pm 1.76\%$) in these transects. At the same time, percentage cover of the remaining species was less than 5%.

Correlations between variables (greatest and smallest diameters, mound height, maximum plant height, and distance to the next, closest vegetation patch) measured in the vegetation patches were all significant at $P < 0.01$. Greatest correlations were observed between (1) the smallest patch diameter and maximum plant height ($r = 0.9515$), (2) greatest and smallest patch diameters ($r = 0.8584$), (3) mound height and greatest vegetation patch diameter ($r = 0.8278$), and (4) mound height and distance to the next, closest vegetation patch ($r = 0.8242$).

In all vegetation patches, the greatest and lowest frequencies were found for *Stipa* spp. (71.2%) and *Grindelia chiloensis* (12.5%). *Stipa* spp. and *Atriplex lampa* showed the highest simultaneous frequency (50%). Simultaneous frequencies for *Atriplex lampa* and *Larrea divaricata*; *Stipa* spp. and *Larrea divaricata*; *Poa* spp. and *Atriplex lampa*; *Lycium chilense* and *Atriplex lampa*; and *Poa* spp. and *Stipa* spp. were between 32 and 37%. All other copresence of each species pair was $\leq 30\%$ and, in 42 out of 78 comparisons, it was $\leq 10\%$.

Some species showed association among themselves, but not with others. Two major species groups can be observed in the dendrogram (Fig. 5). One group has a reasonable association among species (>45%; *Stipa* spp., *Atriplex lampa*, *Larrea divaricata*, *Lycium chilense*, and *Junellia ligustrina* (Lag.) Moldenke). The other group is rather diffuse and constituted by associations of two species (*Poa* spp. and *Acantholippia seriphioides*; *Grindelia chiloensis* and *Senecio aff. filaginoides*; and *Chuquiraga erinacea* and *Gutierrezia* spp.) or isolated species (*Cassia aphylla* and *Panicum urvilleanum* Kunth). In addition, there were positive and negative associations among species. The greatest ($P = 0.0001$) positive association was between *Senecio* spp. and *Grindelia chiloensis* (44.8%) using the Phi-Pearson's index (Table 2).

Relationship between parameters evaluated in the vegetation patches ($n = 16$)

There were no statistical differences ($P > 0.05$) between sites 1 and 2 regarding the variables standing crop (g patch^{-1} or g m^{-2}), greatest and smallest vegetation patch diameters (m), patch surface area (m^2), and plant species diversity. Because of this, values were grouped for both sites.

Larrea divaricata and *Atriplex lampa* contributed more than 84% of the total patch standing crop ($5{,}776.8 \pm 435.0$, Table 3). On a unit surface area measurement, patch standing crop was 961.8 ± 72.9 g m^{-2} (Table 3). *Acantholippia seriphioides* and *Stipa neaei* determined an important standing crop contribution (3.67 and 2.79%, respectively), comparatively to the remaining species in the patch (Table 3).

Average patch size and specific diversity were 5.93 ± 0.33 m^2 and 1.31 ± 0.11, respectively. Aboveground standing crop of the two dominant shrubs (*Larrea divaricata* and *Atriplex lampa*) decreased as plant species diversity increased ($P < 0.05$, Fig. 6).
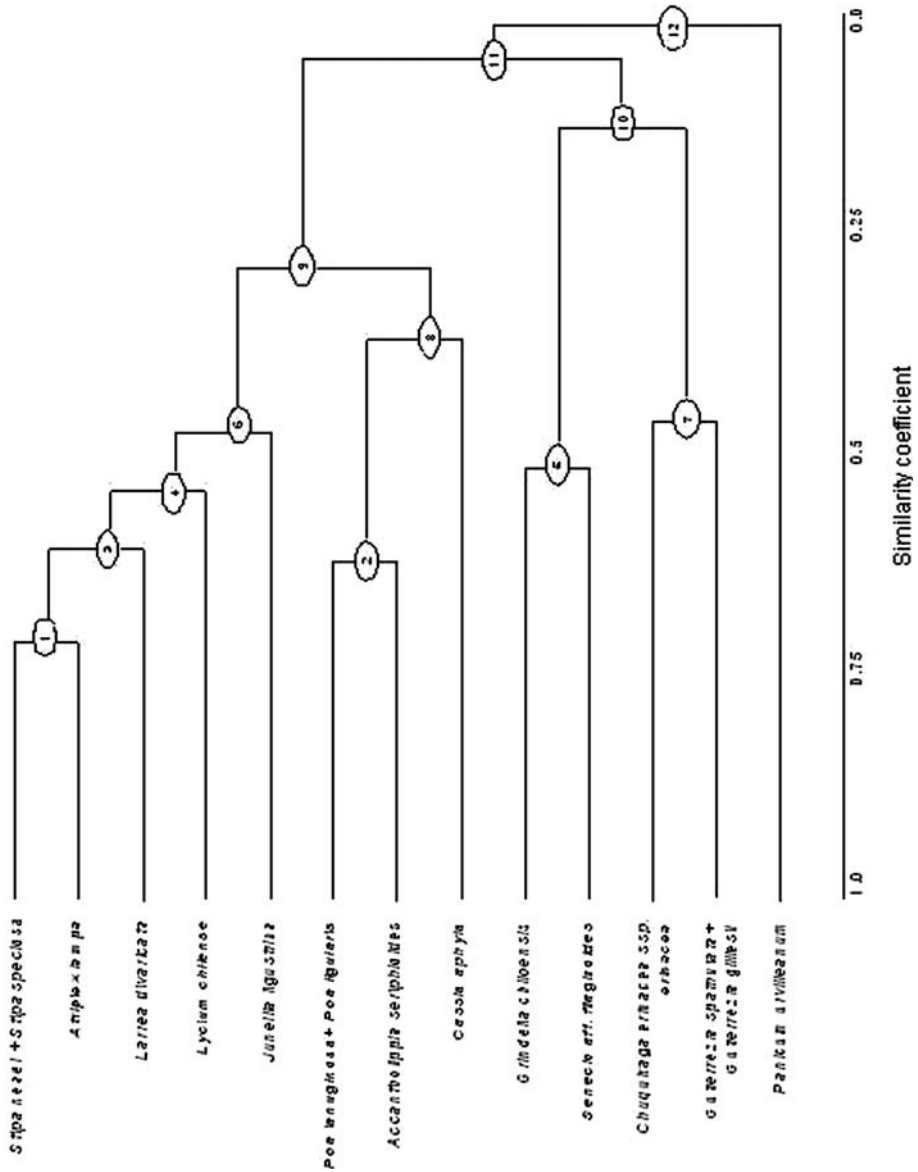
Fig. 5 Dendrogram derived from a grouping analysis (Sorensen) applied to vegetation data

Discussion

Patch structure

Differences in rangeland age and use history, among other factors, may influence comparison of our results with those obtained in other ecological systems. As stated in hyphotesis 1, vegetation patch structural characteristics were similar within the whole

Table 2 Phi-Pearson's indexes to show association between species

	Larrea divaricata	Atriplex lampa	Lycium chilense	Senecio aff. filaginoides	Acantholippia seriphioides	Junellia ligustrina	Gutierrezia spathulata + Gutierrezia gilliesii
Larrea divaricata	1.000						
Atriplex lampa	0.184	1.000					
Lycium chilense	**0.239**	**0.296**	1.000				
Senecio aff. filaginoides	−0.068	0.027	0.066	1.000			
Acantholippia seriphioides	−0.054	−0.069	0.105	0.128	1.000		
Junellia ligustrina	0.185	**0.253**	**0.238**	0.214	**0.223**	1.000	
Gutierrezia spathulata + Gutierrezia gilliesii	−0.169	**−0.225**	−0.184	−0.066	−0.163	−0.029	1.000
Chuquiraga erinacea ssp. Erinacea	**−0.274**	−0.209	−0.145	0.038	−0.181	−0.173	**0.412**
Cassia aphylla	0.008	0.112	**0.276**	0.102	0.161	**0.309**	0.126
Grindelia chiloensis	−0.199	−0.050	0.009	**0.448**	**0.221**	0.152	−0.082
Stipa neaei + Stipa speciosa	0.201	0.130	0.108	−0.019	−0.207	0.130	−0.126
Panicum urvilleanum	**0.298**	0.050	−0.028	−0.202	**−0.249**	−0.168	−0.086
Poa lanuginosa + Poa ligularis	−0.098	0.044	0.057	**0.294**	**0.409**	**0.387**	0.092

	Chuquiraga erinacea ssp. Erinacea	Cassia aphylla	Grindelia chiloensis	Stipa neaei + Stipa speciosa	Panicum urvilleanum	Poa lanuginosa + Poa ligularis
Larrea divaricata						
Atriplex lampa						
Lycium chilense						
Senecio aff. filaginoides						
Acantholippia seriphioides						
Junellia ligustrina						

Table 2 continued

	Chuquiraga erinacea ssp. Erinacea	Cassia aphylla	Grindelia chiloensis	Stipa neaei + Stipa speciosa	Panicum urvilleanum	Poa lanuginosa + Poa ligularis
Gutierrezia spathulata + Gutierrezia gilliesii						
Chuquiraga erinacea ssp. Erinacea	1.000					
Cassia aphylla	−0.011	1.000				
Grindelia chiloensis	0.200	0.080	1.000			
Stipa neaei + Stipa speciosa	0.073	0.059	0.073	1.000		
Panicum urvilleanum	−0.174	**−0.239**	−0.174	**0.219**	1.000	
Poa lanuginosa + Poa ligularis	−0.009	**0.262**	**0.368**	−0.177	**−0.406**	1.000

The sign of the index indicates if the association is either positive or negative

Values in bold are significant at $P < 0.05$

Table 3 Aboveground standing crop (g patch^{-1} and g m^{-2}) and percentage contribution to total plant standing crop of each of the species present in the patch

Species	Standing crop (g patch^{-1})	Standing crop (g m^{-2})	Contribution (%)
Dicotyledoneous shrubs			
Atriplex lampa	900.02 ± 173.57	145.70 ± 27.37	17.34 ± 3.77
Larrea divaricata	4,026.38 ± 444.41	670.92 ± 76.39	67.41 ± 3.73
Lycium chilense	72.65 ± 34.52	11.35 ± 5.43	1.29 ± 0.60
Gutierrezia spathulata	107.23 ± 21.24	18.75 ± 3.75	2.15 ± 0.52
Acantholippia seriphioides	201.31 ± 75.70	36.55 ± 15.53	3.67 ± 1.45
Cassia aphylla	39.38 ± 39.38	6.17 ± 6.17	0.47 ± 0.47
Junellia spp.	150.06 ± 94.72	26.04 ± 17.49	2.36 ± 1.41
Senecio aff. filaginoides	1.37 ± 1.05	0.22 ± 0.17	0.02 ± 0.01
Tetraglochin caespitosum	33.95 ± 12.66	5.59 ± 2.08	0.56 ± 0.18
Herbaceous dicots			
Boopis anthemoides	1.11 ± 0.93	0.18 ± 0.16	0.02 ± 0.02
Hoffmannseggia erecta	23.05 ± 6.30	3.87 ± 1.16	0.49 ± 0.18
Senecio goldsakii	10.69 ± 3.00	2.25 ± 0.53	0.26 ± 0.07
Tarasa sp.	3.96 ± 2.52	0.76 ± 0.48	0.12 ± 0.08
Grasses			
Elymus erianthus	7.38 ± 7.38	0.89 ± 0.89	0.13 ± 0.13
Poa ligularis	32.46 ± 14.65	5.87 ± 2.55	0.43 ± 0.21
Poa lanuginose	0.04 ± 0.03	0.01 ± 0.01	0.0 ± 0.0
Stipa neaei	149.63 ± 29.95	23.98 ± 4.66	2.79 ± 0.61
Stipa speciosa	15.28 ± 7.24	2.68 ± 1.25	0.33 ± 0.16

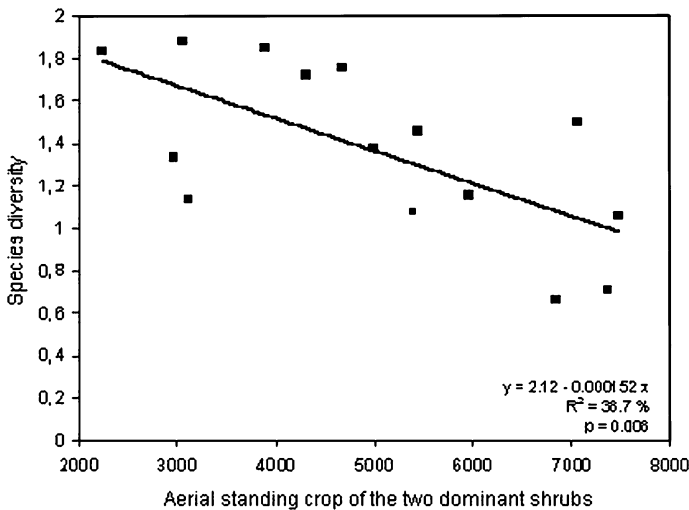Each value is the mean ± 1 SE of $n = 16$

Fig. 6 Relationship between aerial standing crop of the two dominant shrubs (*Larrea divaricata* and *Atriplex lampa*) in any vegetation patch and species diversity. Each *symbol* comes from $n = 1$

study area (15 × 15 km) leaving eroded, bare areas among vegetation patches. This is an indication of a broadly disturbed ecological system. Our values for average patch size (mean = 5.93 m^2) were intermediate than those reported by other authors. For example, Erickson et al. (2005) found minimum-size patches of 3 m^2 in low-stature shrub communities dominated by *Ceanothus cordulatus* Kellogg in the central California Sierras. In the 'Monte' of Río Negro, Cecchi (2000) found that the vegetation patch size (26 m^2) was greater than ours. These patches were composed by *Prosopis alpataco*, *Atriplex lampa* and *Larrea divaricata*, with an herbaceous layer dominated by *Schismus barbatus*.

Similar to our results, Rostagno and del Valle (1988) reported that the greatest and smallest patch average diameters were 355 and 266 cm, respectively, in a shrub-dominated community nearby Puerto Madryn-Chubut, Argentina. In other vegetation patches of the 'Monte' in the Province of Río Negro, Argentina, the mean reported diameter has been 580 cm (Cecchi 2000). Rostagno and del Valle (1988) found that the vegetation patch shape, defined as the relationship between the smallest to the greatest diameter, was 74.9%, a little higher value than the one we found (66.8%). These authors reported 41 cm as the average mound height in their system, which is superior to our value (23 cm). In an environment similar to that studied by Rostagno and del Valle (1988), Gile et al. (1998) described mounds of 4 cm height under plants of *Larrea tridentata* (Sessé & Moc. ex DC.) Coville at the south of New Mexico.

Rostagno and del Valle (1988) reported that height of shrubby vegetation, dominated by *Larrea divaricata*, ranged from 80 to 200 cm. In their study, Schenk et al. (2003) showed that *L. tridentata* reached heights over 200 cm. Gile et al. (1998), however, found individuals of this species of 90–120 cm height. These values are within the range we found for *Larrea divaricata* in our study.

Larrea tridentata and *Ambrosia* have been shown to be spaced between 60 and 260 cm on sandy soil, and between 100 and 260 cm over alluvial material (Schenk et al. 2003). Lateral dispersion of *Larrea*'s roots, which can average 300 cm, occurred in the interspaces (Brisson and Reynolds 1994).

Despite working in a similar ecological system, the vegetation studied by Rostagno and del Valle (1988) was lightly different from that found in our study. They reported *Larrea divaricata*, *Chuquiraga avellanedae* Lorentz, *Prosopidastrum globosum* Gill. ex Hook et Arn., *Schinus polygamus* and *Ephedra ochreata* Miers as the most prominent shrub species, and *Stipa speciosa*, *S. tenuis* Philippi and *S. humilis* Cav. as the major perennial grasses.

Plant cover is an important factor to control partitioning of precipitation and sediment transport. Its dynamics has been recently incorporated into erosion models. In the Monte Austral Neuquino, vegetation patches cover 58% of the soil; the remaining soil stays uncovered between them. Natural vegetation often shows some kind of spatial aggregation which determines the occurrence of the four study locations shown in Fig. 2. Existence of these locations has a wind-related ontogeny. In other arid and semiarid ecosystems, wind has been much stronger winward than under the lee (Wilson and Briske 1978; Mack and Pyke 1984). This had a desiccation effect on the exposed superficial soil layers. Seedlings growing in windy environments can also be particularly vulnerable to abrasive damage by wind-born soil particles (Fryrear et al. 1973). Mounds, which are the result of eolic material accumulation under the shrub canopies, are then always associated to shrubs, and they are apart from each other by bare, vegetation-uncovered areas. Wind velocity is low nearby and below shrubs in the patagonian steppe (Bustos 1995). However, wind is an important factor for propagule dispersal at the study site (Bonvissuto 2006). Vegetation can trap seeds during secondary dispersal (Aguiar and Sala 1997). In the patagonian steppe

grazing has reduced plant cover enhancing the occurrence of nude soil areas where mostly wind removes litter, plant propagules and in some cases the superficial soil layer (Soriano 1956; Ares et al. 1990). Recuperation of native vegetation in these locations uses to be insignificant (Defossé et al. 1997). Seeds reaching the bare interspaces among vegetation patches can be lost due to wind effects (Chambers and McMahon 1994).

Vegetation patches produce areas of vegetated mounds with higher fertility levels and water retention capacity, and uncovered areas with different erosion rates in arid and semiarid regions (Sánchez and Puigdefabregas 1994). Rostagno and del Valle (1988) estimated that mounds under *Larrea divaricata* and *C. avellanedae* were 40% vegetated in soils of northeast Patagonia, Argentina.

Larrea divaricata (∼20% cover) and *Atriplex lampa* (around 5% cover) appeared as the dominant shrubs in the transects. These species, together with other shrub and subshrub species, covered 30.4% of transects. Cover of perennial grass species did not exceed 2%. Bisigato and Bertiller (1997) observed that vegetation cover was 40–60% in a 'Monte' of eastern Chubut, with an annual average precipitation (188 mm) a little lower than ours during the study (200 mm). In their investigation, cover of *L. tridentata* and *Atriplex lampa*, together with that of the remaining shrub and subshrub species, almost reached 22% (to a distance of 600 m from a water source for domestic animals). At the same time, cover of individual grass species was less than 2%, except *Stipa speciosa* (3.5% at 3,000 m from that water source).

Relationship between parameters evaluated in the vegetation patches

Study of correlations between the variables measured at each vegetation patch allowed to learn that (1) the higher the plant, the thinner the vegetation patch, (2) the relationship between the greatest and smallest diameters was relatively constant, and (3) the higher the mound, the greater the distance to the next, closest vegetation patch.

In this study, and in agreement with hypothesis 2, some species were found associated among themselves, but not with others. We obtained two groups of species. The first major group was composed by *Atriplex lampa*, *Larrea divaricata*, *Lycium chilense* and *Junellia ligustrina* which appear to foster growth of the perennial grasses *Stipa* spp. and *Poa* spp. This group is composed by species characteristic of the vegetation patches found in fields best appropriate for utilization with grazing animals, with greater forage production. The second species group was rather diffuse and composed by the species pairs *Grindelia chiloensis* and *Senecio aff. filaginoides*, *Acantholippia seriphioides* and *Poa* spp., and *Gutierrezia* spp. and *Chuquiraga erinacea*. The latest four species can be found in highly deteriorated, rangeland areas (Bonvissuto, personal observation, INTA EEA Bariloche, Argentina). *Cassia aphyla* and *Panicum urvilleanum* did not associate with other species. *Panicum urvilleanum* grows in more sandy soils, outside the vegetation patches.

Most of the species forming the vegetation patches showed positive associations between them. However, species which were out of the vegetation patches demonstrated negative associations with respect to those present in them. These findings also agree with hypothesis 2. For example, positive associations were proved between *Larrea divaricata* and (a) *Lycium chilense* (0.239), (b) *Atriplex lampa* (0.184), and (c) *Stipa* spp. (0.201). Morello (1956) observed that *Larrea divaricata* lives associated to *Atriplex lampa* in places where the water table is between 10 and 30 m depth. Schenk (2004) emphasizes that a few species can access the total soil water availability (shallow depths: <0.4 m; medium depths: 0.4–1.2 m; deep soil: >1.2 m, and water table). This could explain dominance of these species over large areas in desertic zones. It is possible that this may occur with

Larrea divaricata, if its root system is similar to that of *L. tridentata*. Gile et al. (1998) found that roots of *L. tridentata* reached 5 m depth with a horizontal extension of more than 4 m. *Atriplex*, which is a C_4 shrub with roots distributed close to the soil surface and up to 2 m depth (Lee and Lauenroth 1994), appears to utilize as much the water stored deep in the soil profile (as a result of the snow melt in spring) as the water coming from summer rains (when spring precipitations are scarce and soil water recharge is poor).

Negative associations involved *Gutierrezia* spp. and *Chuquiraga erinacea* with most of the species that associate between them to form the vegetation patches. In the Mojave Desert, plants of *Ambrosia dumosa* (A. Gray) Payne are spatially segregated to a 3 m distance from those of *L. tridentata*. This is because of negative root effects of *Larrea* on roots of *Ambrosia* (Schenk et al. 2003).

The mechanisms which foster development of these either positive or negative associations are the subject of future research. Some species, for e.g., may need the microenvironment generated by shrubs, such as *L. tridentata* and *Atriplex lampa*, for a successful germination, emergence and seedling establishment. *Larrea divaricata* and *Atriplex lampa* contributed 4,026 and 900 g dry weight, respectively, per vegetation patch. In this manner, they could be acting as nurse plants. West (1989) reported that shrubs can serve as the nucleus of the successional change and ecosystem development. He emphasized that shrubs can serve as nurse plants because, for e.g., most of the seedlings of *Pinus* and *Juniperus* were under the canopy of *Artemisia* and other associated shrubs. Yeaton (1978) demonstrated that *L. tridentata* served as a nurse plant for *Opuntia leptocaulis* DC. in the Chihuahua desert.

Larrea tridentata, in turn, has used *Ambrosia dumosa* (shrub of the Asteraceae family) as nurse plant. Eighty-five percent of all young plants of *Larrea* were rooted under the canopy of *Ambrosia dumosa*. Most of the establishment of *Larrea* occurred close to the live individuals of *Ambrosia dumosa* (www.fs.fed.us/database/feis/plants.nical_and_ecological_characteristics.html).

Shrub contribution to average total standing crop in the vegetation patch was a little more than 900 g m^{-2}. This is about half the contribution of *L. tridentata* (2,000–2,057 g m^{-2}) in the area of Sevilleta (Gosz and Milne 2005).

Northrup et al. (1995) found that as size of the vegetation patch increased, its structural development also increased. In our study, high correlations were established between the (1) smallest mound diameter and the maximum plant height, (2) smallest and largest diameter of the patch, (3) mound height and greatest patch diameter, and (4) mound height and distance to the next, closest vegetation patch.

Average diversity of the vegetation patch was 1.31 in our research. In ten shrubby communities of northwestern Spain, Basanta et al. (1989) found diversity values from 0.8 to 2.0, with an average of 1.58. Standing crop of the two dominant shrubs (*Larrea divaricata* and *Atriplex lampa*) and species diversity were negatively correlated in our study, which agrees with hypothesis 3. These results agree with those of Burkhardt and Tisdale (1969), who found that succession to juniper-dominated communities in central and eastern Oregon, northeastern California, and southwestern Idaho, was accompanied by reductions in understory species diversity. Understory species diversity was also reduced by juniper dominance in southeastern Oregon (Bates et al. 2000). This response was attributed to the high juniper belowground interference for soil water and nitrogen. Plant species diversity begins to decline at some point following disturbance because either the most competitive plants eliminate the rest or the plants most resistant to damage by physical extremes or natural enemies eventually occupy most of the space (Connell 1978).

Management and conservation implications

Forty-two percent of the soil surface were bare, uncovered areas among vegetation patches in our study, and mounds under the shrubs were only 40% vegetated in soils of northeast Patagonia (Rostagno and del Valle 1988). The Soil Conservation Service in Iceland has been very concerned with protection and reclamation of denuded range areas (Thorsteinsson et al. 1971). In these areas, for e.g., they have re-seeded grasses for haymaking and grazing. They have also worked with livestock management. Different means for improving distribution of the livestock, such as increased fencing, distribution of salt on the range, herding, etc., are being used in some areas. This livestock management approach should also be taken in the Monte Austral Neuquino to avoid further degradation of the ecological system.

Rupture of the superficial soil layers in the interpatch spaces together with seeding of shrub species like *Larrea divaricata*, *Atriplex lampa* and *Gutierrezia* sp. could also be beneficial for vegetating the interspaces in years with above-average precipitation. It is recommended to wash the seeds of *Atriplex lampa* before seeding because of the presence of chemical inhibitors in its bracts (Bustos, unpublished data, EEA INTA Bariloche, Argentina). Germination of *L. tridentata* is also related to precipitation since a minimum of at least 25 mm appears necessary to induce its germination (Marshall 1995). Raven et al. (1986) also suggest that some seeds will not germinate after having some abrasion treatment such as that provided from the soil. Scarification of *Larrea divaricata* seeds is recommended before conducting germination trials with this species (Zappe, personal communication, EEA INTA Bariloche, Argentina). When seed germination of *Larrea divaricata* was conducted under controlled moisture and temperature, and its seeds were scarified, germination percentages of up to 25% were obtained (Bonvissuto and Busso 2007b). Norton and Bermant (1977) have demonstrated that *Larrea divaricata* seeds occasionally germinate in the infertile, bare vegetation interspaces.

Overgrazing by cattle in certain arid and semiarid landscapes has resulted in severe soil compaction and manipulations such as heavy discing are required to promote infiltration of water and reestablish herbaceous vegetation in the interspaces among vegetation patches (Fulbright 1991). Forb species richness was greater during the first year after discing than on untreated areas in southern Texas (Bozzo et al. 1992). Because herbaceous species diversity may temporarily increase, treatments such as discing and roller chopping may result in posttreatment plant communities that are temporarily greater in species diversity than nontreated communities (Fulbright 1996).

Under conditions of severe stress, where establishment of an independent propagule can be a long and risky process, vegetative reproduction present advantages which derives of its capacity to sustain the offspring (Grime 1981). It is very likely that the vegetation establishment in the corridors among vegetation patches (location 4) is dependent upon colonization of rhizomatous species such as *Poa ligularis* and *Stipa neaei*. These species could be occupying the empty places until generation of an adequate environment for the successful germination and seedling establishment of good forage, non-rhizomatous species such as *L. erianthus* and *Poa ligularis*. The increase in herbaceous cover in the interspaces is important from a hydrologic standpoint. Erosion rates in semiarid systems are greatest in interspace zones (Wilcox and Breshears 1994). Also, research in New Mexico suggests that herbaceous ground cover is more effective at reducing erosion rates than is cover of woody vegetation (Wilcox and Breshears 1994). Clonal propagation is ecologically important because (1) it enables plants in harsh environments to persist until suitable conditions for viable seed production or seedling establishment are encountered, (2) it enables plants to survive shoot damage resulting from grazing, fire, freezing, or

drought stress, and (3) the mode of vegetative growth determines how plants exploit space and resource heterogeneity (Jackson et al. 1985).

Colonization of empty places might also depend of shrub species highly resistant to severe stress like *Gutierrezia* sp. It appears that its ability to colonize a site depends mainly of its reproductive potential and possible early establishment, which was reported by Osman and Pieper (1988) in New Mexico where vegetation was dominated by *Gutierrezia sarothrae* (Pursh.) Britt. and Rusby, and annual precipitation was 220 mm. This latter species produces a large seed amount and present a rapid seedling establishment, which makes it an increasing species in the community (Osman and Pieper 1988). Its root system is quite superficial and provides stability to the soil against wind and hydric erosion (Tirmenstein 1999). Superficial root systems can take advantage of short-term rainfall events (lower than 5 mm) which are very common at the study site (Busso 1997). Rains of this magnitude have rapidly stimulated physiological processes in perennial grasses (Sala and Lahuenroth 1982) and might be good used by the superficial root system of *Gutierrezia*. Advance of *Gutierrezia* sp., possibly through seedling establishment from a large seed pool (e.g., Ladyman 2003 for *G. sarothrae*), and of *Poa lanuginosa* and *Stipa neaei* (both rhizomatous perennial grasses) to bare, unvegetated sites, was reported after 3 years (1999–2001) with precipitations 50% above the historical mean. This was observed by Bonvissuto (personal observation, EEA INTA Bariloche, Argentina) in exclosures to domestic livestock in the Monte Austral Neuquino.

During the summer 2002/2003, with precipitation values close to the historical mean in 2002, it was observed that the plant cover had diminished in the corridors. This was because the aboveground portion of the rhizomatous grasses did not prosper in the less favourable year (Bonvissuto, personal observation, EEA INTA Bariloche, Argentina). It is possible that in such rigorous areas as the Monte Austral Neuquino, vegetation can not definitively cover the corridors among vegetation patches because of stressful ecological conditions. However, it appears that in favourable years perennial vegetation has a greater cover among vegetation patches. As a result, it can be thought that recuperation of these environments would be based in improving the structure and diversity of the already existent vegetation patches. This could be achieved by implementing strategic resting periods which avoid vegetation grazing such as plants can recuperate its vigour. This would be highly favoured if these periods were coincident with precipitation years above the historical mean.

We supposed at the beginning of this work that formation of new vegetation patches would start from germination and subsequent establishment of *Larrea divaricata* and *Atriplex lampa*. Shrubs like *L. tridentata* are plants which present clonal subdivisions which can facilitate dispersion of the plant canopy and facilitate the formation of large clonal patches (Schenk 1999). Lateral branches of *Larrea* can be covered by sand, and they can root close to the point where they show up from the mound (Barbour 1969). Thus, shrub groups do not develop from seedlings which grow in closed association. Formation of vegetation patches could also start from seeds of shrubs species colonizing empty areas such as *Gutierrezia* spp. This species will later generate an environment adequate for establishment of other species.

Mismanagement of vegetation patches for domestic herbivory will eventually lead to reduce species richness, leaving only inedible plant species (mostly shrubs) in the community. This is because of the selective nature of grazing animals on the most palatable species. However, the magnitude of this impact on structure and function of a community can be minimised by managing the timing and intensity of paddock use (Ludwig et al. 1999).

One particularly important interaction between ecological and hydrological processes in patchy semiarid lands is how vegetation patches serve to obstruct runoff, and then how this retained water increases patch growth that, in turn, provides feedbacks to the system (Ludwig et al. 2004). Such ecohydrological interactions have been mostly demonstrated for semiarid landscapes with distinctly banded vegetation patterns. Their findings do support the conclusion that vegetation patches and runoff-erosion processes do strongly interact in many semiarid landscapes across the globe. For example, Ludwig and Tongway (1995) emphasized the importance of mounds in the vegetation patches because they collect water from rainfall and air-borne particulate matter from wind. Mounds create microhabitats where soil aeration and infiltration are improved due to microfauna and microflora (Tongway and Ludwig 1989). These patches of grass clumps and debris bridges act as filters allowing large water flows to pass through without damaging the patch (Ludwig and Tongway 1995). They capture and retain scarce resources within the landscape system rather than being carried out of the system. Loss of these patches alters the whole system of erosion and deposition. Each runoff volume increases with loss of patches, thereby transporting larger quantities of eroded materials. The interception zone may not be able to handle larger volumes of runoff, which may lead to a reduced infiltration rate. This will lead to degradation of soil quality, which could decrease to levels below what is necessary to support the microhabitat. HilleRisLambers et al. (2001) also reported that pattern formation can occur in semiarid areas given only the positive feedback between plant density and local water infiltration coupled with the spatial redistribution of runoff water.

Conservation of vegetation patches is then crucial to prevent increased soil erosion and desertification in those ecological systems. This is especially true when we look at the findings of Bonvissuto (2006), who studied the vegetation patches and the surrounding bare areas in northwestern Patagonia, Argentina. Research of Bonvissuto (2006) was conducted in a study site with a rough soil surface area which will tend to slow down run-off, hence increasing the time for water infiltration and soil water storage. She found that emergence and seedling survival of two perennial grass species, *Stipa neaei* and *Elymus erianthus*, were greater underneath the shrubs in the vegetation patches than in the interpatch, plant-uncovered interspaces. This was determined, at least in part, by lower apparent soil density, wind speeds and temperature amplitudes, and greater water infiltration rates, and gravimetric soil water and nutrient contents in the mounds below the shrubs than in the bare, surrounding areas. Experimental plots underneath the shrubs in her studies were covered by *Stipa neaei* and *Poa lanuginosa* through vegetative reproduction, after germination took place from the soil seed bank. An important seed rain reached the interspaces, but no individuals survived the high summer temperatures which used to reach 50°C. Her results leave no doubts that appropriate rangeland management is essential for maintaining the vegetation patches, and thus some vegetation cover and soil erosion control in that ecological system.

Tongway and Ludwig (1996) reported that several attempts to re-seed bare areas of the landscape using mainly grass species failed due to the poor match between introduced species and soil type. Also, re-seeding along with various mechanical soil reclamation treatments were also attempted with minimal success (Tongway and Ludwig 1996). Nelson (1999) recognized the need to take a more comprehensive approach. This entails understanding the natural vegetation patterning and nutrient cycling of a landscape. If the nutrient cycling of a soil system is understood, it is more likely that a reasonable strategy for restoration and conservation of the soil will be developed. Once the soil is restored in the interpatch areas, re-establishment of vegetation is more feasible, thereby making the re-establishment of vegetation patterns and the effective restoration of the landscape possible.

Tongway and Ludwig (1990) studied the patterning of the semiarid mulga lands of eastern Australia to determine the processes that make the existing natural vegetation patches fertile. The landscape was considered as a whole, instead of only concentrating in the bare patches. Once the landscape patterning was determined, it was possible to develop a strategy to restore the soil's characteristics and nutrients, increasing the likelihood of successfully reestablishing vegetation. Later on, Tongway and Ludwig (1996) treated bare soil areas with branches. Accumulated soil was a result of both fluvial and aeolian processes. The branches caught debris by the wind from as far as 100 m away. Once the branches caught the debris, it was dropped onto the soil, and was not easily re-suspended. Water also brought leaf litter from the surrounding areas. These authors stated that the only treatment to consistently rehabilitate bare sites, alone or in combination with fertilizer and litter treatments, was the application of branches. This restoration effort was successful in proving that a simple, inexpensive technique can be used to re-establish the landscape to pre-grazing conditions. This technique demonstrates a way to amend the nutrient cycling in the soil, improve water filtration and create a microhabitat for organisms. This is a very important aspect since, as in any arid or semiarid environment, the climate has unpredictable rainfall events. The restoration effort also shows the importance of understanding the natural patterning of the landscape. Previous restoration efforts had failed because the important landscape processes had been ignored (Nelson 1999).

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