J Insect Conserv (2016) 20:929–944
DOI 10.1007/s10841-016-9918-2

ORIGINAL PAPER

Above-ground arthropod community structure and influence of structural-retention management in southern Patagonian scrublands, Argentina

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Received: 16 May 2016 / Accepted: 15 September 2016 / Published online: 18 October 2016

Abstract Southern Patagonia's landscape hosts several semi-natural habitats, traditionally used for sheep production, such as *Mulguraea tridens* and *Lepidophyllum cupressiforme* scrublands. *Mulguraea* scrublands are managed via mechanical shredding to remove shrubs and increase grass availability, alternating with structural-retention strips. We analyzed the influence of structural-retention management (with cut and retention strips) in *Mulguraea* scrublands with regards to above-ground arthropod community structure, as well as differences between the two natural scrubland types. We worked in Santa Cruz Province (Argentina) with pitfall traps during two summers in the first 2 years after mechanical shredding. Richness, abundance, occurrence frequency, Shannon–Wiener diversity and Pielou evenness indices, and similarity among assemblages were evaluated using univariate and multivariate statistical tests. Complementarily, we described vegetation ground cover and microclimate. We collected 3279 individuals from 38 species belonging to Insecta and Arachnida Classes. Shannon–Wiener diversity and Pielou evenness indices, as well as the overall assemblages, differed significantly between managed cut strips and natural *Mulguraea* scrublands, mainly due to the loss and introduction of species from surrounding environments; abundance also differed in the first sampling year compared to the second year. Likewise, managed retention strips allowed the partial maintenance of arthropod community structure and had a microclimate that was similar to natural *Mulguraea* scrublands, although assemblages in managed cut and retention strips became more similar among themselves in the second sampling year. On the other hand, richness, abundance and assemblage of both natural scrubland types differed significantly, with 87% more indicator species in *Mulguraea* than in *Lepidophyllum* scrublands. Greater dissimilarity occurred between both natural scrubland types in dryer years, which could be related with an El Niño Southern Oscillation event. If arthropod community structure changes prove stable over time, mechanical shredding with structural-retention management would allow for an increase in the sheep carrying capacity, while reducing impacts to the arthropod community, thus providing a viable compromise between productivity and conservation in a fragile arid environment. More studies are necessary to evaluate long term changes in above-ground arthropod community structure of scrublands in arid zones of southern Patagonia.

Keywords Ants · Beetles · Camel spiders · Conservation · Scorpions · South Patagonia · Xeric landscape

Introduction

Argentinean southern Patagonia comprises two provinces: Santa Cruz, at the extreme austral tip of the South American mainland, and Tierra del Fuego, an archipelago separated from the continent by the Magellan Strait. The landscape of southern Patagonia hosts several semi-natural habitats, varying from *Nothofagus* forests to arid steppes. The

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steppes, including grasslands and scrublands, cover 85 % of the total area in Santa Cruz and 25 % in Tierra del Fuego.

Two shrub species are common in the Patagonian steppe: *Mulguraea tridens* (Lag.) N. O'Leary and P. Peralta 2009 and *Lepidophyllum cupressiforme* (Lam.) Cass. 1816, which dominate separately in both scrubland types. *M. tridens* (Verbenaceae) is a woody endemic species, 0.70 m in height, which grows continuously or intermixed with xeric steppe dominated by *Stipa speciosa* or *Festuca pallescens*, and covers 2.83 million hectares in the southeastern portion of Santa Cruz. *L. cupressiforme* (Asteraceae) is also a woody endemic species, 0.50 m in height, which develops in saline and sandy depressions on seashores and river beds, jointly with halophytic steppe species (León et al. 1998), and occupies small areas at the southernmost edge of Santa Cruz and northern Tierra del Fuego (Oliva et al. 2001). These two scrubland types develop close to each other in many zones of Santa Cruz, and it is usually assumed by ranchers and policymakers that they form part of the same habitat. Very little information exists to confirm whether faunal diversity, function, ecosystem services and conservation value are the same in both scrubland types.

Patagonian steppe has been grazed by domestic livestock (mainly sheep) for more than 100 years, with stocking rates ranging at present from 0.13 to 0.75 head·ha^{-1}·year^{-1} (Peri et al. 2013). Livestock range freely in very large and heterogeneous paddocks, with year-long continuous grazing (Golluscio et al. 1998). Recently, to promote grass growth for sheep consumption, the above-ground portions of shrubs were removed via mechanical shredding as part of a large-scale management assay in some *Mulguraea* scrublands. Removal of shrubs was applied in strips (8 m wide), alternated with structural-retention strips (4 m wide). An added practical benefit of retention strips is protection for livestock and newly installed vegetation against the exceptionally strong Patagonian winds. Furthermore, structural retentions (preserved individuals or patches, of variable size and shape, without intervention for long periods of time) are also proposed for in situ conservation in managed landscapes (Franklin et al. 1997). In forests, retentions also have proven to preserve some of the microclimatic characteristics, structural complexity, biological legacies and original heterogeneity of ecosystems (Gustafsson et al. 2012).

Management via mechanical shredding is applied in several other scrublands around the world, mainly for fuel reduction, and the effects studied have been mainly over the shrubs themselves (e.g., Potts et al. 2010; Fernández et al. 2015), plant communities (e.g., Rollins and Bryant 1986; Allegretti et al. 1997; Daryanto and Eldridge 2010), and soil microbes (Fontúrbel et al. 2016). In *Mulguraea* scrublands, some effects of shrub removal have been evaluated, such as changes in soil water content and vegetation growth in the cut area (Peri et al. 2011; Rivera et al. 2011). However, there is a lack of information about other effects of shrub removal or structural-retention on scrubland ecosystem structure, processes, functions and biodiversity components, such as above-ground arthropod community structure.

Currently, arthropods have become increasingly popular as environmental and ecological impact indicators (Niemelä 2001; Underwood and Fisher 2006; Gerlach et al. 2013). Their usefulness for studying the impact of environmental changes and management on biodiversity derives from their numerical abundance, diversity of life strategies, and being involved in most ecosystem functions. Their small size and short life cycles also make them physically and numerically reactive to changing conditions (Werner and Raffa 2000; Sackmann and Flores 2009; McGeoch et al. 2010; Gerlach et al. 2013). They are, therefore, ideal to study changes in species composition and turnover among habitats, particularly in managed systems. Likewise, arthropod diversity responds to land use changes in many ecosystem types. For example, in agricultural landscapes, the natural habitat islands inside arable land were found to host many unique ground-dwelling arthropod species that were not presented within the surrounding crops (Knapp and Řezáč 2015); in grasslands, different graminoid species supported distinct invertebrate assemblages and less complex host plants supported fewer invertebrate individuals and species (Reid and Hochuli 2007); in southern Patagonia forests, harvesting reduces old-growth forest insect richness and significantly changes the original insect community's assemblage (Lencinas et al. 2014). Finally, scrubland arthropod community structure and distribution in desertified steppe ecosystems are related to shrub species identity and cover (Mazía et al. 2006; Liu et al. 2012, 2015a, 2016; Li et al. 2013; Zhao and Liu 2013; Kwok and Eldridge 2016), as well as shrub microhabitat differences (Liu et al. 2015b).

The objective of this study was to analyze above-ground arthropod community structure and the influence of structural-retention management in scrublands in southern Patagonia, Argentina. We evaluated the effect of structural-retention management in *Mulguraea* scrublands, as well as differences between natural *Mulguraea* and *Lepidophyllum* scrublands (continuous patches without removal intervention). We pursued answers to these driving questions: (i) how does above-ground arthropod community structure (richness, abundance, diversity and evenness indices, and assemblage) change after structural-retention management in *Mulguraea* scrublands (cut and retention strips), compared with natural continuous patches without removal intervention; and (ii) is above-ground arthropod community structure different between natural *Mulguraea* and *Lepidophyllum* scrublands?

Materials and methods

Study area

The study was conducted at two sheep ranches (51°07′23″ SL, 70°58′38″ WL; 51°38′00″ SL, 69°38′00″ WL) in the province of Santa Cruz, Argentina, where xerophytic *Mulguraea* and *Lepidophyllum* scrublands are common. Local weather is cold and dry, with mean annual temperatures between 6.5 and 8.5 °C. Rainfall fluctuates between 150 and 200 mm per year, with a peak in winter. Soils are Aridisols and Molisols, deep and sandy, with good drainage and stones in the profile (Godagnone and Salazar Lea Plaza 2004).

In three different patches of *Mulguraea* scrublands (approximately 1.5 ha each), ranch owners created 3–6 strips (8 m width × approx. 200 m length each), where the shrub layer was removed [managed cut strips in *Mulguraea* (MCM)], using a hydraulic shredder (FALC, CONDOR Model, with a working width of 1.80 m) to cut plants to a height of 2 cm. The standard management practice leaves shrub debris inside the cut strip, to minimize evapotranspiration and wind erosion, which represented (mean ± SD) 7995 ± 4990 kg dry matter.ha^{-1}. MCM strips were separated by retention strips [4 m width; managed retention strips in *Mulguraea* (MRM)] (Fig. 1), where the shrub layer was maintained without removal. Strips were oriented north–south, perpendicular to the main wind direction. Removal was implemented in winter 2009 (August), when soil was frozen and disturbance by machine wheels would be minimized. Near each patch of managed *Mulguraea* scrublands, there were also continuous patches (approximately 2 ha each) of *Mulguraea* [natural *Mulguraea* (NM)] and *Lepidophyllum* [natural *Lepidophyllum* (NL)] scrublands, which were maintained without removal intervention. Grazing was prevented by fencing off each studied patch during 2009–2013 to isolate any effects of vegetation removal/retention.

Vegetation and microclimatic characterization

Floristic descriptions were done for MCM, NM and NL in December 2009 (plant species list presented in Appendix 1). MRM could not be floristically surveyed due to limitations in time/field assistant availability during fieldwork. Ground cover was estimated by categories (total vegetation, soil without vegetation, litter, dead standing shrubs, lichens and shrub debris) in MCM, NM and NL.

Likewise, microclimatic characterization was done in MCM and NM during two consecutive growing seasons from November 2009 to May 2010, and from November 2010 to May 2011 (Appendix 2), allowing us to evaluate microclimate variability in the two most contrasting situations within *Mulguraea* scrublands. MRM and NL could not be characterized due to limitations in instrument availability. We evaluated soil and air temperature, and relative air humidity with HOBO H8 data loggers (Onset Computer Corporation, USA). Temperatures were evaluated monthly by mean, maximum and minimum values. On the other hand, rainfall was measured in an open area near the scrublands with a pluviometer (Cavadevices, Argentina). We also calculated MEI (multivariate El Niño Southern Oscillation-ENSO index) in each period (January–February and February–March in 2010 and 2011), using available data at http://www.esrl.noaa.gov/psd/enso/mei/, which indicates the occurrence of El Niño conditions when MEI is positive, and La Niña conditions when MEI is negative (Wolter and Timlin 1998).

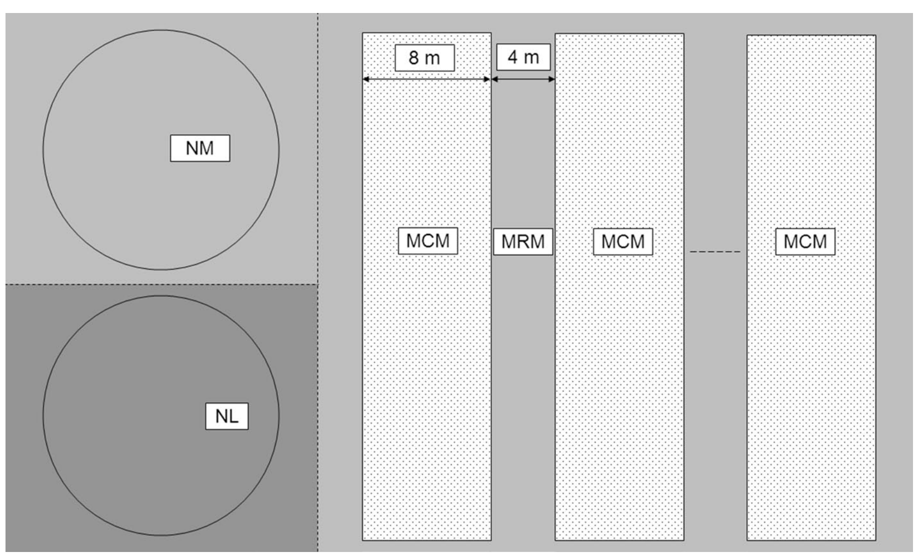

Fig. 1 Schematic representation of mechanical shredding implementation and location of scrubland types (*MCM* managed cut strips in *Mulguraea, MRM* managed retention strips in *Mulguraea, NM* natural *Mulguraea, NL* natural *Lepidophyllum*) in each of three managed/natural patches. MCM were 3–6 in each managed patch. NM and NL are near each managed patch

Arthropod sampling

We randomly selected plots in each managed and natural scrubland type (MCM, MRM, NM and NL), evenly distributed in the three managed/natural patches described in the "Study area" section. Differences in soil characteristics, slope, exposition (data not shown) and vegetation among the three managed/natural patches were not enough to be considered blocks; therefore, we made statistical analyses following a completely random design. Plots were at least 100 m apart one from each other and located in the central part of strips or continuous patches to minimize any edge effect.

Based on previous experience (unpublished data) and other works in southern Patagonia (Cheli and Corley 2010), contents of individual traps would have resulted in low catch counts, given low arthropod density in the xeric steppe. Therefore, we used pitfall traps arranged in sets of five in each plot, and contents of the five traps in a plot were pooled and used as a single sample to raise total capture per sample (Cheli and Corley 2010). There were located either: (i) aligned and 5 m apart in the strips (MCM, MRM), or (ii) one central and four positioned orthogonally at 5 m from the center in the continuous patches (NM, NL). Each trap consisted of a plastic pot (12 cm diameter and 14 cm height) buried in the ground until the opening was level with the surface, and filled to a third of its volume with soapy water (300 ml) as the killing agent. Traps were set in summer (February), a thermally relevant period for insect activity in southern Patagonia (Niemelä 1990), and remained active for 1 week before contents were collected. Samples were taken in two consecutive years (2010–2011). A total of 49 samples were obtained: 25 in 2010 (replicates were MCM = 6, MRM = 6, NM = 4, NL = 9), and 24 in 2011 (MCM = 6, MRM = 6, NM = 6, NL = 6). There was a loss of NM samples in 2010 caused by wild animals (trampling by guanacos and scratching by foxes) and weather (extremely strong winds). We sampled more plots in NL in the first sampling year to compensate for what we interpret as unfavorable sampling conditions at this site (e.g., low apparent arthropod densities, traps susceptible to being lost). When we were able to collect all NL samples, we did not wish to discard the extra data; therefore, the nine samples were conserved for the analyses. The favorable sampling efficacy was enough to equilibrate replicates with other scrubland types in the second sampling year.

In the laboratory, we classified and quantified individuals from Coleoptera, Formicidae (Hymenoptera) and Orthoptera in the Class Insecta, and Scorpionida and Solifuga in the Class Arachnida. We identified individuals to the genus or species level when possible, and classified them in recognizable taxonomic units or morphospecies (Oliver and Beattie 1993), when genus or species could not be determined (hereafter, "species"). Voucher specimens are deposited in the invertebrate collection at *Centro Austral de Investigaciones Científicas* (CADIC-CONICET) in Ushuaia, Argentina. Arthropod species list is presented in Appendix 2.

Data analysis

We estimated richness (S) (number of species), abundance (A) (number of individuals), occurrence frequency (%), Shannon–Wiener diversity (H′) and Pielou evenness (J) to characterize above-ground arthropod community structure. Richness calculations were made per plot, managed and natural scrubland type, and at the whole sampling levels, while relative abundance was calculated per plot only. Occurrence frequency for each species was obtained as a proportion of the occurrence in each plot relative to the total plots, for each managed and natural scrubland type, and for the whole study. Shannon–Wiener diversity index was obtained as $H' = -\sum p_i \ln p_i$, where p_i is relative abundance of i species at each plot; Pielou evenness index was obtained as $J = H'/H'_{max}$, where $H'_{max} = \ln(S)$, where S is from each plot (Pielou 1975).

After checking that statistical assumptions were met, we used two-way ANOVAs to evaluate the effects of structural-retention management in above-ground arthropod community structure of *Mulguraea* scrublands [driving question (i)], with managed scrubland types (MCM, MRM and NM) and sampling years (2010, 2011) as main factors. The interaction term (managed scrubland type x sampling year) was also analyzed. Similarly, we used two-way ANOVAs to evaluate the effect of dominant shrub species composition on above-ground arthropod community structure [driving question (ii)], with natural scrubland types (NM and NL) and sampling years (2010, 2011) as main factors. The interaction term (natural scrubland type × sampling year) was also analyzed. Including the sampling year as a main factor allowed exploring the variability between samplings of two consecutive years, which was then related to climatic variations produced by ENSO event. In these analyses, the response variables were: (i) average species richness per sample (number of species × trap set × sampling period), (ii) abundance per sample (number of individuals × trap set × sampling period), (iii) Shannon–Wiener diversity index, and (iv) Pielou evenness index. Post-hoc multiple comparisons were done by a posteriori contrasts, employing the Scheffé method (Sokal and Rohlf 1981). Statgraphics (Statistical Graphics Corp., USA) software was used for these analyses.

To graphically illustrate the similarity among assemblages in managed (MCM, MRM, NM) and natural (NM, NL) scrubland types in the two sampling years, we conducted non-metric multi-dimensional scaling (NMDS) ordinations (Minchin 1987) using the manual methodology, Bray-Curtis distance and with 250 iterations. A Monte Carlo test was used to evaluate stress in randomized data; probability was presented for each axis. This methodology is widely used to graphically analyze arthropod assemblages and composition (e.g., Grove and Forster 2011; Baker at al. 2015). Points representing samples for different scrubland types and sampling years were plotted in ordination space, with the distances between points proportional to the dissimilarity of their arthropod assemblages.

Subsequently, we performed quantitative analyses to evaluate differences in above-ground arthropod assemblages with Multi-Response Permutation Procedures (MRPP) based on Bray-Curtis distance and using T, p-value and A for evaluation (McCune and Grace 2002). The statistic "T" describes the separation between the groups (the more negative is T, the stronger the separation) and have an associated "p-value" determined by numerical integration of the Pearson type III distribution. "A" is the chance-corrected within-group agreement, which describes the within-group homogeneity compared to the random expectation, adopting values of $A=1$ when all items are identical within groups, $A=0$ when heterogeneity within groups equals expectation by chance, and $A<0$ if there is less agreement within groups than expected by chance. Subsequent pairwise groupings within variables were tested to determine where the differences were (Zimmerman et al. 1985). We tested the null hypothesis of no differences between groups of samples from different sampling years, and among groups of samples from scrubland types by sampling year. In MRPP the dependent variable was the abundance. This methodology is widely used to compare categorical variables for differences (e.g., Jacobs et al. 2007; Grove and Forster 2011).

Finally, we used Indicator Species Analysis (Dufrêne and Legendre 1997) to explore possible associations (in specificity and fidelity) of above-ground arthropods with scrubland types at each sampling year (e.g., Jacobs et al. 2007; Grove and Forster 2011). These analyses included a random reallocation procedure with 4999 permutations (Monte Carlo test) to evaluate the significance of the maximum indicator values (IndVal) provided ($p<0.05$). Following Tejeda-Cruz et al. (2008), we considered as "indicator species" those species with IndVal >50 and p values lower than 0.05. Indicator Species Analysis was performed for managed and natural scrubland types, jointly and separately for the two sampling years.

We used the software PC-ORD (McCune and Mefford 1999) to conduct NMDS, MRPP and Indicator Species Analysis. Singletons and doubletons (species represented by one or two individuals, respectively) were omitted from these multivariate analyses.

Results

Floristic characterization found 38 vascular plant species in managed and natural scrubland types, with 24 species in MCM, 20 in NM and 13 in NL (Appendix 1). Total vegetation ground cover was 20–30 % in MCM, 50–75 % in NM, and >75 % in NL. Complementarily, soil without vegetation (10–30 %), litter (1–5 %), and dead standing shrubs and lichens (<1 % each) presented similar ground cover values in MCM, NM and NL. Shrub debris was only present in MCM, accounting for 50–75 % of ground cover.

Microclimatic characterization showed air temperatures in February 2011 were 2.5–3.5 °C higher than in 2010, in both MCM and NM, while soils temperatures were 1.9–3.3 °C higher (Appendix 2). Similarly, rainfall was higher in February 2011 than in February 2010, but November and December 2010 had no rainfall at all. Relative humidity presented similar values in February 2010 and 2011. MEI was positive in January–February and February–March 2010 (1.52 and 1.39 respectively), and negative in the same periods in 2011 (−1.56 in both).

Above-ground arthropod samplings collected 3279 individuals, in 38 species and 12 families (Table 1), of which ten were singletons and six doubletons. Most species belonged to Coleoptera (31 spp.), particularly Curculionidae (17 spp.) and Tenebrionidae (7 spp.). Ants (Formicidae) were the most abundant group (2375 individuals). The overall richness values varied from 27 to 19 species, following the order MRM > MCM > NL > NM, while abundance fluctuated from 1272 to 269 individuals per scrubland type, being MRM > NM > MCM > NL. Occurrence frequency also varied with scrubland type (Appendix 2), with only one species (*Dorymyrmex antarcticus*, Formicidae) present in all samples. Another ant species (*Pogonomyrmex vermiculatus*) was also very frequent, present in all *Mulguraea* scrubland samples (100 % occurrence frequency) and at lower frequencies in *Lepidophyllum* scrublands (13 % occurrence frequency). The only other species with a frequency >50 % was *Emallodera multipunctata* (Tenebrionidae), while 12 species presented occurrence frequencies between 10 and 50 %, and 23 species occurred in <10 % of samples of the whole study.

Managed scrublands harbored 82 % (31 species) of the total observed richness, with four exclusive species in MCM and five in MRM (Appendix 2). Significant differences in Shannon–Wiener diversity and Pielou evenness indices between NM and MCM were observed (Table 2), with greater indices for MCM (mean = 1.47 ± 0.61 standard deviation, and 0.38 ± 0.16, respectively) than for NM (1.13 ± 0.53 and 0.26 ± 0.13), while MRM did not differ from NM. Richness did not significantly vary among managed scrubland types (MCM = 8.5 ± 2.9, MRM = 8.8 ± 4.3 and NM = 7.2 ± 3.0

Table 1 Above-ground arthropod abundance (A) (number of individuals) for the whole sampling and managed/natural scrubland types, taxonomically classified by class, order and family

Class	Order	Family	Whole sampling			MCM	MRM	NM	NL
			A (S)	SI	D	A (S)	A (S)	A (S)	A (S)
Insecta	Coleoptera	Curculionidae	70 (17)	9	4	20 (6)	18 (9)	24 (5)	8 (7)
		Tenebrionidae	497 (7)			176 (6)	173 (6)	117 (6)	31 (4)
		Carabidae	66 (2)		1	26 (2)	30 (1)	7 (1)	3 (2)
		Chrysomelidae	4 (1)			4 (1)			
		Cleridae	2 (1)		1		1 (1)		1 (1)
		Ptinidae	7 (1)			2 (1)	1 (1)		4 (1)
		Scarabaeidae	130 (1)			62 (1)	37 (1)	30 (1)	1 (1)
		Staphylinidae	1 (2)	1			1 (1)		
	Hymenoptera	Formicidae	2375 (2)			414 (2)	980 (2)	798 (2)	183 (2)
	Orthoptera	Acrididae	19 (3)			6 (2)	5 (3)	5 (2)	3 (1)
Arachnida	Scorpionida	Bothriuridae	43 (1)			19 (1)	17 (1)	4 (1)	3 (1)
	Solifuga	Mummucidae	65 (1)			10 (1)	9 (1)	14 (1)	32 (1)
Total	5	12	3279 (38)	10	6	739 (23)	1272 (27)	999 (19)	269 (21)

Species richness (S) (number of species) is in parenthesis, with singletons (SI) and doubletons (D) discriminated for the whole sampling

MCM managed cut strips in *Mulguraea, MRM* managed retention strips in *Mulguraea, NM* natural *Mulguraea, NL* natural *Lepidophyllum*

Table 2 Two-way ANOVA results of richness (number of species), abundance (number of individuals), Shannon–Wiener diversity and Pielou evenness, for evaluation of managed and natural scrubland types.

Study	Factor	Level	Richness	Abundance	Shannon–Wiener	Pielou
Managed scrublands	M: management type	MCM	8.5	61.6 a	1.47 b	0.38 b
		MRM	8.8	106.0 b	1.30 ab	0.29 a
		NM	7.2	103.6 b	1.13 a	0.26 a
	F (p)		2.24 (0.128)	6.18 (0.006)	3.36 (0.049)	4.02 (0.029)
	Y: sampling year	2010	5.3 a	94.5	0.76 a	0.20 a
		2011	11.0 b	86.3	1.84 b	0.42 b
	F (p)		81.32 (<0.001)	0.47 (0.499)	102.61 (<0.001)	36.24 (<0.001)
	M×Y: *F (p)*		1.49 (0.243)	5.98 (0.007)	2.48 (0.102)	3.01 (0.065)
Natural scrublands	N: natural shrub type	NM	7.2 b	103.6 b	1.13	0.26
		NL	4.7 a	17.8 a	1.02	0.40
	F (p)		34.63 (<0.001)	56.02 (<0.001)	0.69 (0.414)	3.92 (0.062)
	Y: sampling year	2010	3.4 a	51.1	0.52 a	0.17 a
		2011	8.4 b	70.3	1.63 b	0.49 b
	F (p)		138.50 (<0.001)	2.82 (0.108)	76.60 (<0.001)	17.38 (<0.001)
	N×Y: *F (p)*		0.62 (0.441)	2.46 (0.132)	2.52 (0.127)	2.07 (0.166)

In managed scrublands, management type (*MCM* managed cut strips in *Mulguraea, MRM* managed retention strips in *Mulguraea, NM* natural *Mulguraea*) and sampling years (2010 and 2011) were the main factors. In natural scrublands, shrub type (*NM* and *NL* natural *Lepidophyllum*) and sampling years (2010 and 2011) were the main factors. Interaction between the main factors is also shown for each analysis

F (p) = Fisher test and significance between parenthesis. Different letters indicate significant differences ($p < 0.05$) using a posteriori contrasts between groups

species). On the other hand, sampling years affected richness, Shannon–Wiener diversity and Pielou evenness indices, with higher values in 2011 than in 2010 (11.0 ± 2.1 vs. 5.3 ± 1.7 species, 1.84 ± 0.29 vs. 0.76 ± 0.39, and 0.42 ± 0.08 vs. 0.20 ± 0.15, respectively). Abundance presented a significant interaction between the main factors (Fig. 2), which occurred by higher and similar values for MRM and NM (124.0 ± 43.7 and 122.2 ± 64.4 individuals, respectively) than in MCM (37.2 ± 27.2 individuals) in 2010, meanwhile intermediate and similar values were recorded in 2011 for the three managed scrubland types (88.0 ± 29.6 in MRM, 86.0 ± 19.2 in MCM and 85.0 ± 13.7 individuals in NM).

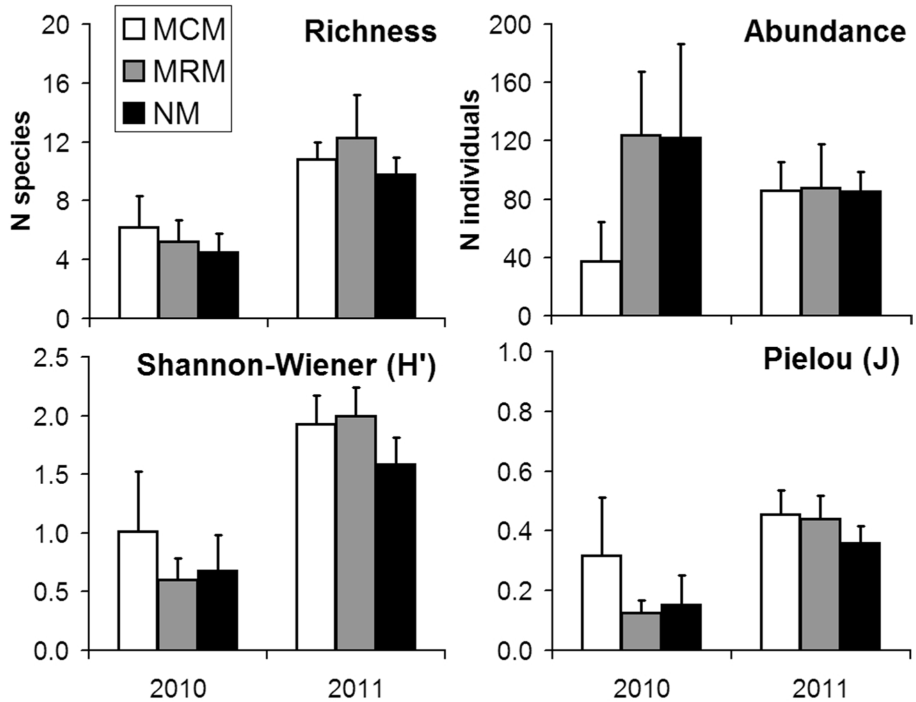

Fig. 2 Graphical representation of interactions among managed scrubland types and sampling years for richness (number of species), abundance (number of individuals), Shannon–Wiener diversity and Pielou evenness indices, according to Table 2 (*MCM* managed cut strips in *Mulguraea, MRM* managed retention strips in *Mulguraea, NM* natural *Mulguraea, NL* natural *Lepidophyllum*). Error bars indicate standard deviation

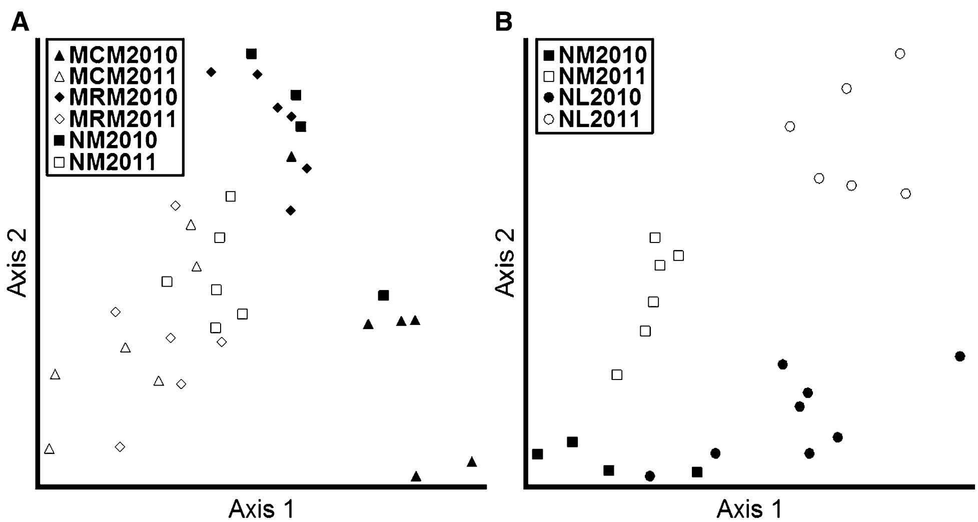

Fig. 3 Non Metric Multidimensional Scaling (NMDS) analyses for arthropod assemblages in (**A**) managed and (**B**) natural scrubland types. Points represent samples, according to scrubland types (*MCM* managed cut strips in *Mulguraea, MRM* managed retention strips in *Mulguraea, NM* natural *Mulguraea, NL* natural *Lepidophyllum*) and sampling years (2010 and 2011). In (**A**) Axis 1 presented 44.69 stress (p = 0.128), while Axis 2 showed 16.189 stress (p = 0.004), in (**B**) Axis 1 presented 47.68 stress (p = 0.004), while Axis 2 showed 16.145 stress (p = 0.004)

Natural scrublands harbored 76% (29 species) of the total observed richness, with 10 exclusive species in NL and eight in NM (Appendix 2). Significant differences in species richness and abundances between natural scrubland types and between sampling years were detected (Table 2). Both richness and abundance were higher in NM compared to NL (7.2 ± 3.0 vs. 4.7 ± 2.5 species; 103.6 ± 43.1 vs. 17.8 ± 13.4 individuals). However, neither Shannon–Wiener diversity nor Pielou evenness showed significant differences between natural scrublands (H′ = 1.02 ± 0.74 and J = 0.40 ± 0.30 for NL, H′ = 1.13 ± 0.53 and J = 0.26 ± 0.13 for NM). Furthermore, richness, Shannon–Wiener diversity and Pielou evenness were significantly different between the two sampling years, all of which were higher during the first sampling year (3.4 ± 1.4 to 8.4 ± 1.8 species, 0.52 ± 0.37 to 1.63 ± 0.22, and 0.17 ± 0.21 to 0.49 ± 0.17, respectively). Meanwhile, abundance was similar for both sampling years (51.1 ± 61.0 individuals at 2010 and 70.3 ± 51.1 individuals at 2011). No interactions were observed in natural scrubland types.

NMDS analysis (Fig. 3) confirmed grouping for sampling year and scrubland type according to similarity among above-ground arthropod assemblages. For managed scrublands, NMDS analysis (10.23868 final stress for two-dimensional solution; 0.00039 final instability) showed a clear split between sampling years (Fig. 3A). In the 2010 group, MCM samples were greatly separated from the others. Contrary to this, scrubland types did not show clear differences in the 2011 group, although NM samples were more concentrated and partially separated from MRM and MCM. For managed scrublands, MRPP showed statistically significant differences between groups of samples from different sampling years, and in almost all comparisons among groups of samples from each scrubland type and year (Table 3). However, many of these groups presented some heterogeneity (A near 0) (e.g., NM2011 vs. MRM2011, NM2011 vs. MCM2011). Moreover, MRPP showed no differences for comparisons NM2010 versus MRM2010 and MRM2011 versus MCM2011, although with less agreement within groups than expected by chance (A < 0).

In natural scrublands (Fig. 3B), NMDS (9.17392 final stress for two-dimensional solution; 0.00392 final instability) clearly split between sampling years, but also between natural scrubland types. For 2010, there was a small partial overlapping between NL and NM, while 2011 showed no overlapping. NL2011 was the more distinct group. For natural scrublands, MRPP showed statistically significant differences between groups of samples from different sampling years, and in all comparisons among groups of samples from each scrubland type and year (Table 3). High A values were observed in these comparisons, e.g. A = 0.365 in NM2011 versus NL2011.

Finally, Indicator Species Analysis (Table 4) for comparison among managed scrubland types showed only one indicator in NM for 2011 samplings (one species of Tenebrionidae) and none in MRM and MCM. On the other hand, comparison by Indicator Species Analysis between natural scrubland types detected one indicator species in NL for 2011 samplings (Mummucidae family), while NM presented eight indicator species (two Formicidae and one Curculionidae from 2010 samplings, and four Tenebrionidae

Table 3 Multi-response permutation procedure (MRPP) results to evaluate differences among above-ground arthropod assemblages in managed and natural scrublands

MRPP	Overall	Group comparison	T	A	p	Significance
Managed scrublands	Sampling years		−13.184	0.181	<0.001	**
	Managed scrubland type by sampling year		−8.512	0.280	<0.001	**
		NM2010 versus MRM2010	**0.889**	**−0.049**	**0.825**	**NS**
		NM2010 versus MCM2010	**−1.999**	**0.136**	**0.048**	*
		NM2010 versus NM2011	−3.794	0.220	0.006	**
		NM2010 versus MRM2011	−4.159	0.256	0.004	**
		NM2010 versus MCM2011	−4.336	0.261	0.003	**
		MRM2010 versus MCM2010	**−4.443**	**0.234**	**0.003**	**
		MRM2010 versus NM2011	−5.243	0.635	0.001	**
		MRM2010 versus MRM2011	−6.129	0.330	<0.001	**
		MRM2010 versus MCM2011	−6.027	0.327	<0.001	**
		MCM2010 versus NM2011	−4.91	0.209	0.001	**
		MCM2010 versus MRM2011	−5.201	0.201	<0.001	**
		MCM2010 versus MCM2011	−5.476	0.221	<0.001	**
		NM2011 versus MRM2011	**−2.332**	**0.063**	**0.023**	*
		NM2011 versus MCM2011	**−2.838**	**0.094**	**0.018**	*
		MRM2011 versus MCM2011	**1.292**	**−0.034**	**0.936**	**NS**
Natural scrublands	Sampling years		−4.383	0.083	0.003	**
	Natural scrubland type by sampling year		−10.414	0.364	<0.001	**
		NM2010 versus NL2010	**−4.664**	**0.247**	**0.002**	**
		NM2010 versus NM2011	**−3.794**	**0.220**	**0.006**	**
		NM2010 versus NL2011	−5.244	0.330	0.001	**
		NL2010 versus NM2011	−6.946	0.322	<0.001	**
		NL2010 versus NL2011	**−6.343**	**0.189**	<0.001	**
		NM2011 versus NL2011	**−3.943**	**0.365**	<0.001	**

Comparisons were calculated between groups of samples from different sampling years, and among groups of samples from each scrubland type and year (*MCM* managed cut strips in *Mulguraea, MRM* managed retention strips in *Mulguraea, NM* natural *Mulguraea, NL* natural *Lepidophyllum;* sampling years = 2010 and 2011)

T is the statistic of MRPP, A is the chance-corrected within-group agreement, p is the probability associated with T. *NS* not significant differences (p > 0.05); *significant differences (p ≤ 0.05); **highly significant differences (p < 0.01). Main comparisons are in bold

Table 4 Values from Indicator Species Analysis (IndVal and probability) for comparisons among managed scrubland types (*MCM* managed cut strips in *Mulguraea, MRM* managed retention strips in *Mulguraea, NM* natural *Mulguraea*), and between natural scrubland types (*NM* and *NL* natural *Lepidophyllum*)

Species	Managed scrublands			Natural scrublands	
	MCM	MRM	NM	NM	NL
Nyctelia sp. 2			51.1 (0.002)**	59 (0.008)**	
Nyctelia sp. 1				100 (<0.001)**	
Taurocerastes patagonicus				97.8 (<0.001)**	
Nyctelia multicristata				93.6 (<0.001)**	
Emallodera multipunctata				82.3 (<0.001)**	
Curculionidae 1				75 (0.002)*	
Pogonomyrmex vermiculatus				65.3 (0.015)*	
Dorymyrmex antarcticus				59.3 (0.003)*	
Mummucidae 1					51.7 (0.034)**

Super-index corresponded to sampling years: *2010; **2011

and one Scarabeidae from 2011 samplings), although some of them had rather low indicator values (Table 4).

Discussion

Mulguraea scrubland management

Sheep are frequently set to pasture on *Mulguraea* scrublands, even though the presence of the shrubs reduces the productivity of these areas (Cibils and Borrelli 2005). Shrub removal improves the availability of grasses and edible plants for sheep, and also the site's conditions for the growth of new seedlings (Billoni et al. 2014). However, this practice represents a significant modification of the environment, which was demonstrated by changes in above-ground arthropod diversity in this study.

Although average species richness did not significantly change with management, shrub removal affected above-ground arthropod community structure, as was shown by differences in Shannon–Wiener diversity, Pielou evenness and abundance. Moreover, assemblages also changed, as was demonstrated by NMDS and MRPP, mainly in the first year following shredding. Many differences found in MRPP comparisons among groups of managed scrubland types by year presented moderately large chance-corrected within-group agreement ($A \geq 0.3$), being therefore ecologically significant (McCune and Grace 2002). The increase in richness and Shannon–Wiener diversity and Pielou evenness in MCM is mainly due to the incursion of opportunistic species that prefer the impacted conditions (Koivula et al. 2003; Gerlach et al. 2013), or species associated with debris, which had a very high ground cover and biomass in this managed treatment, in addition to the species associated with the nearby shrubs. There is much evidence about debris housing a specific invertebrate assemblage, such as saproxylic fauna of coarse woody debris in forests (e.g., Groove and Forster 2011), though more studies are necessary to identify arthropod species associated to litter, coarse and fine woody debris in scrublands. However, there is also the possibility that our sampling method might have been more effective in the open conditions as opposed to the sheltered ground under intact shrubs (Greenslade 1964). In the second year after removal, MCM above-ground arthropod species recovered their abundance to similar values than in NM and MRM, which could be a rapid adaptation of the arthropod community to the new site conditions.

Jointly with the shrub removal treatment to promote the growth of more palatable vegetation for sheep, structural-retention strips provide some protection to the original arthropod community inside these strips. In our analyses, MRM showed greater similarities with NM than with MCM, because the MRM arthropod community maintained most of the species found in the natural *Mulguraea* scrubland, mainly in the first year after removal. This is clearly confirmed by MRPP analysis, which showed no differences between NM2010 and MRM2010, although the within-group heterogeneity was greater than expected by chance ($A < 0$, Table 3). This is also clearly observed in the NMDS graph (Fig. 3A), where MRM2010 samples presented some dispersion, and NM2010 samples were clearly separated in two groups, with one sample more similar to MCM2010. This finding implies that natural *Mulguraea* scrublands had some internal variability that could influence the response of arthropod assemblages to shredding management, and which should be investigated with more specific studies.

On the other hand, arthropod species proportions in MRM probably depend on the proximity to MCM, because these cut areas were next to the scrubland retention strips, and arthropods could move from one strip to the other.

This could be considered as an indirect effect of removal (MCM) on the retention strip (MRM), allowing for the introduction of new species associated with MCM to MRM, which is more evident in the second year after removal. In this sense, MRPP analysis showed no differences between MRM2011 and MCM2011, again with within-group heterogeneity greater than expected by chance ($A < 0$, Table 4). Likewise, the changes in above-ground arthropod assemblages among sampling years were greater in MRM than in NM (T from MRPP was −6.129 between MRM2010 and MRM2011, and −3.794 between NM2010 and NM2011, Table 3). Therefore the arthropod community of the retention strip was initially similar to the natural scrubland 1 year after mechanical shredding, but resembled the more strongly modified cut area (MCM) a year later. This was also expressed in the ANOVA interaction on the abundance (Table 2; Fig. 2), which greatly diminished in the second year after shredding in MRM (as well as in NM), but incremented in MCM. This could be an incremental effect generated by management practices added to climatic influence, or to more susceptibility in MRM due to proximity to cut areas. It will be necessary to continue monitoring changes in arthropod communities for longer time periods and different seasons to better evaluate stability in MRM over time, as well as the effect of different strip widths. Further samplings might show a progressive shift in the assemblage, changing it completely (Sinclair et al. 2006).

Finally, exploratory Indicator Species Analysis among managed scrublands also showed that it was not possible to detect indicator species for different management types, except one Tenebrionidae for NM2011 (*Nyctelia* sp. 2) with a low indicator value (51.1, Table 4). Species specificity and fidelity were not sufficiently different in managed scrubland types at least in the first 2 years after shredding, but limited time span of sampling in this work could be insufficient to describe the community in enough detail to detect this level of specificity. More research is needed to completely assess above-ground arthropod diversity along the whole year in managed *Mulguraea* scrublands.

Natural *Mulguraea* and *Lepidophyllum* scrublands

The xeric Patagonian landscape is generally dominated by grasslands, but scrublands are typical in river valleys, in the lowlands near plateaus or terraces, or in lake and sea shores (Roig 1998). Due to similar physiognomies, NM and NL are usually considered in the same administrative category ("scrublands") and as similar ecosystems by ranch owners, resource managers, authorities and legal administrators, without considering if these are inhabited by different flora and fauna. This work provides evidence for differences between the two natural scrubland types in their above-ground arthropod and vascular plant communities. This was also highlighted by ANOVAs for species richness and abundance, and by the exploratory results of the Indicator Species Analysis (Table 4), which detected nine indicator species between both NM and NL, despite the limited time span of sampling in this work. On the other hand, NL could be interpreted as a more homogeneous environment than NM, due to vegetation structure highly dominated by few shrub species in NL (Appendix 1) while NM forms mosaics with grasses, chamaephytes, etc. (Roig 1998). Therefore, NM could support different niches for different species. As was stated before, heterogeneity within NM groups is also highlighted by MRPP results.

Influence of annual microclimatic variability

Inter-annual differences in above-ground arthropod diversity in natural scrublands could be related to wide scale changes, such as differences in climate (mainly temperature and rainfall) between years. Monthly mean air temperature and rainfall in 2010 were similar to those modeled for the region (Kreps et al. 2012), but while monthly mean air temperature in 2011 was 2–3 °C higher than modeled data, mean maxima air temperature was 7–9 °C higher, both in NM and MCM (Appendix 2). This increase in temperatures could be determining higher richness and diversity indices in 2011 compared to 2010 in both natural scrublands conditions. Contrarily, mean temperature increases seem to negatively influence NM arthropod abundance, but almost do not affect NL, which could be related to the specific tolerance to high temperatures of arthropod species. Observed differences in rainfall and temperature patterns between sampling years is strongly explained by the ENSO event, showing a positive MEI in 2010 (lower temperatures and rainfall) that indicates the occurrence of El Niño conditions, and a negative MEI in 2011 (higher temperatures and rainfall) that indicates the occurrence of La Niña conditions (Wolter and Timlin 1998). Differences in above-ground arthropod diversity in both NL and NM could be related with variations in temperature and rainfall, which were more dissimilar in 2011 and more similar in 2010.

Ecological and conservation implications

Arthropods have been suggested as potential environmental indicators for Patagonian ecosystems (Sackmann et al. 2006; Lencinas et al. 2008, 2014), and our arthropod data suggest that beetles, ants and camel spiders are sensitive to environmental changes. In xeric environments such as the Patagonian steppe, Tenebrionidae, Curculionidae and Formicidae are especially suited to these conditions (Sømme 1995). Above-ground arthropod diversity in NL and NM could potentially provide environmental and ecological indicators, but their utility as biodiversity indicators (correlation among arthropods and other group diversities) in agricultural landscapes must be further evaluated in combination with other arthropod groups (e.g., Heteroptera and Hymenoptera) and vegetation inventories (Duelli et al. 1999).

There are seven state protected areas in the Argentine province of Santa Cruz. Three are national parks and four are provincial natural reserves. However, none of them include *Lepidophyllum* or *Mulguraea* scrublands in their jurisdictions. Furthermore, pressure by overgrazing in these scrublands is evident, mainly in areas with more than a 100 years of uninterrupted use. The conservation status of scrublands is critical, and it is important to develop strategies for their preservation, as well as research that offers more information about ecological characteristics and functions of these habitats. For example, *Mulguraea* branch structure makes this species highly resistant to wind and snow, therefore it is a pioneer and constructor species, which favors grasslands development and protects herb species under its canopy influence. Moreover, it is able to recolonize and repopulate eroded areas by seeds, because bare and removed soils favor its multiplication (Roig 1998). On the other hand, *Lepidophyllum* scrublands control erosion in rivers, lakes and sea coasts, mainly where saline sediments are deposited, due to its capacity to tolerate high soil salt contents (Faggi 1985).

Management strategies with "retention areas" are appropriated to offset the lack of natural protected areas in southern Patagonia that include characteristic scrublands, according to the aim of in situ conservation. The proximity of structural-retention to cut areas could allow above-ground arthropod species to survive near the managed areas, permanently or until the scrubland structure recovers. Also, managing landscapes for a greater range of habitat conditions may be essential for some organisms. More studies are necessary to evaluate effects of different size, shape and distribution of retention areas in managed scrublands in southern Patagonia, as well as different size, shape, cut intensity and spatial distribution of shredding areas. The effect of successive management cycles over the same scrubland area, as well as changes in the original community assemblages by annual variability and management must be further analyzed by long term studies.

Conclusions

Structural-retention management in *Mulguraea* scrublands modified above-ground arthropod diversity, by both loss and introduction of species from surrounding habitats. Nevertheless, retention strips maintain some of the original microclimate, complexity, heterogeneity and legacies of the original scrubland structure, providing potential *in situ* conservation. Likewise, above-ground arthropod diversity differed in natural *Lepidophyllum* and *Mulguraea* scrublands. Differences in arthropod diversity for the two consecutive years of our study could be related to inter-annual climatic variability, which was strongly influenced by an ENSO event. If arthropod diversity changes prove stable over time, mechanical shredding with structural-retention management would allow for an increase in loading capacity for sheep production while reducing impacts on the arthropod community, thus providing a viable compromise between productivity and conservation in a fragile arid environment. Undoubtedly, some completely uncut areas would need to be maintained in proximity to provide reservoirs of scrubland specific species (Mazía et al. 2006). Further follow-up sampling will be required to determine the outcome of this experimental manipulation.

Acknowledgments We appreciate the collaboration of Emilio Rivera, María Vriginia Sturzenbaum, Francisco Milicevic and Sabrina Billoni in field measurements. The authors also gratefully thank the *Centro Austral de Investigaciones Científicas* and *Instituto Nacional de Tecnología Agropecuaria* for their support during the realization of this work.

Appendix 1

See Table 5.

Table 5 Vascular plant species list for floristically surveyed scrubland types (*MCM* managed cut strips in *Mulguraea; NM* natural *Mulguraea, NL* natural *Lepidophyllum*), classified by life form, origin and class cover (>75, 75-50, 50-5, 5-1, <1%)

Life form	Species	Author	Origin	MCM (%)	NM (%)	NL (%)
Shrub	*Lepidophyllum cupressiforme*	(Lam.) Cass. 1816	Endemic			>75
	Nardophyllum brioides	(Lam.) Cabrera 1954	Endemic	1–5	<1	1–5
	Senecio filaginoides	DC. 1838	Endemic			<1
	Mulguraea tridens	(Lag.) N. O'Leary and P. Peralta 2009	Endemic	5–50	50–75	
	Berberis microphylla	G. Forst. 1787	Endemic	1–5	1–5	
Subshrub	*Nassauvia glomerulosa*	(Lag. ex Lindl.) D. Don 1832	Endemic	1–5	<1	<1
	Ephedra frustillata	Miers 1863	Endemic	<1	<1	<1
	Atriplex vulgatissima	Speg. 1897	Native			<1
	Clinopodium darwinii	(Benth.) Kuntze	Endemic	5–50	5–50	
	Nassauvia aculeata	(Less.) Poepp. and Endl. var. aculeata 1835	Endemic	1–5	1–5	
	Colobanthus lycopodioides	Griseb. 1854	Endemic	1–5	<1	
Gramm	*Pappostipa chrysophylla*	(E. Desv.) Romasch. 2008	Endemic	5–50	5–50	1–5
	Poa spiciformis	(Steud.) Hauman and Parodi 1929	Endemic	5–50	5–50	<1
	Pappostipa ibarii	(Phil.) Romasch. 2008	Endemic	1–5	<1	<1
	Festuca pyrogea	Speg. 1896	Endemic	1–5	1–5	
	Bromus setifolius	J. Presl 1830	Endemic	1–5	1–5	
	Carex argentina	Barros	Endemic	<1	<1	
	Trisetum spicatum ssp. cumingii	(Nees ex Steud.) Finot 2010	Endemic		<1	
Herb	*Acaena sericea*	J. Jacq. 1816	Endemic			<1
	Armeria maritima	(Mill.) Willd. 1809	Endemic			<1
	Azorrella trifurcata	(Gaertn.) Pers. 1805	Endemic			<1
	Juncus balticus ssp. mexicanus	(Willd. ex Roem. and Schult.) Kirschner 2002	Native			<1
	Plantago patagonica	Jacq. 1795	Native			<1
	Rumex crispus	L. 1753	Exotic			<1
	Tripleurospermum inodorum	(L.) Sch. Bip. 1844	Exotic			<1
	Valeriana carnosa	Sm. 1791	Endemic			<1
	Burkartia lanigera	(Hook. and Arn.) Crisci 1976	Endemic			<1
	Perezia recurvata	(Vahl) Less. 1830	Endemic	<1	<1	<1
	Acaena poeppigiana	Gay 1847	Endemic	<1	<1	
	Cerastium arvense	L. 1753	Exotic	<1	<1	
	Azorella fuegiana	Speg. 1896	Endemic	<1	<1	
	Calandrinia caespitosa	Gillies ex Arn. 1831	Endemic	<1	<1	
	Polygala darwiniana	A.W. Benn. 1879	Endemic	<1		
	Nassauvia darwinii	(Hook. and Arn.) O. Hoffm. and Dusén 1901	Endemic	1–5		
	Leucheria purpurea	(Vahl) Hook. and Arn. 1836	Endemic	<1		
	Senecio magellanicus	Hook. and Arn. 1841	Endemic	<1		
	Oxalis sp.		Native	<1		

Appendix 2

See Fig. 4.

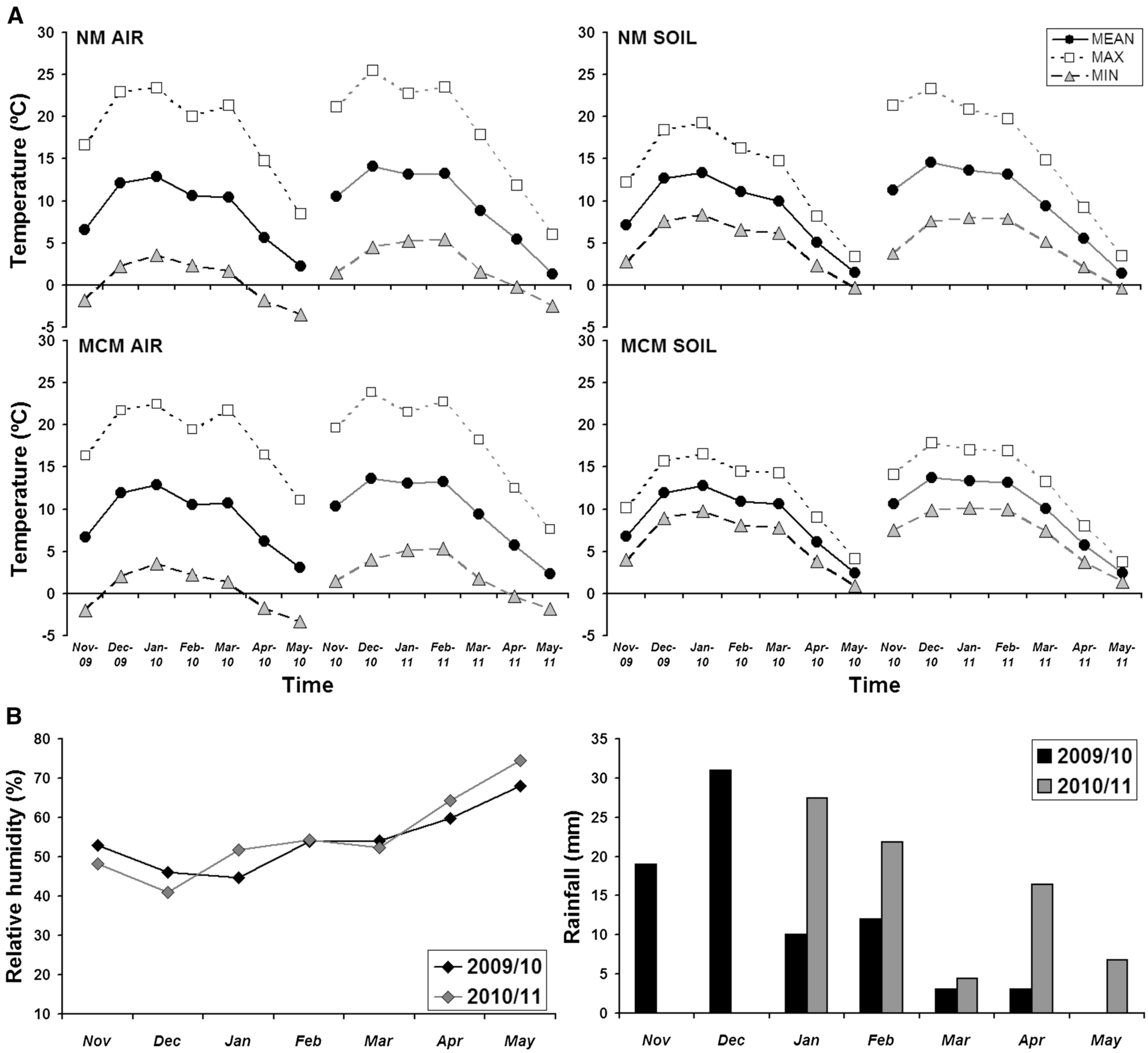

Fig. 4 Microclimatic characterization of managed and natural *Mulguraea* scrublands, presenting mean, maximum and minimum monthly air and soil temperature in managed cut strips (MCM) and natural (NM) scrublands throughout 2009–2010 and 2010–2011 growing seasons (**A**), and relative humidity and rainfall, measured in a open area near *Mulguraea* scrublands, throughout 2009–2010 and 2010–2011 growing seasons (**B**). The growing season extends from November to May

Appendix 3

See Table 6.

Table 6 Above-ground arthropod species list for scrubland types (*MCM* managed cut strips in *Mulguraea, MRM* managed retention strips in *Mulguraea, NM* natural *Mulguraea, NL* natural *Lepidophyllum*), showing occurrence frequency by site (without distinction of sampling year)

Class	Order	Family	Species	MCM (%)	MRM (%)	NM (%)	NL (%)	Total (%)
INSECTA	Coleoptera	Carabidae	*Barypus clivinoides*	50	50	30	13	36
			Metius spp.	8			7	4
		Ptinidae	Ptinidae 1	8	8		20	9
		Chrysomelidae	Chrysomelidae 1	25				6
		Cleridae	Cleridae 1		8		7	4
		Tenebrionidae	*Emallodera multipunctata*	67	83	60	40	63
			Nyctelia multicristata	58	58	60	13	48
			Nyctelia sp. 1	58	58	60		44
			Nyctelia sp. 2	25	42	60	33	40
			Tenebrionidae 1	33	42	10		21
			Tenebrionidae 2	25	17	10		13
			Tenebrionidae 3				20	5
		Curculionidae	*Caneorhinus lineatus*	8			7	4
			Caneorhinus sp. 1		8			2
			Caneorhinus sp. 2		8			2
			Cylydrorhinus angulatus	33	33	30		24
			Cylydrorhinus deltipennis		8	10		5
			Cylydrorhinus sp. 1		8		7	4
			Cylydrorhinus sp. 2		8			2
			Puranius nigrinus				7	2
			Curculionidae 1	17	33	30		20
			Curculionidae 2	17		10		7
			Curculionidae 3			10	13	6
			Curculionidae 4		8		7	4
			Curculionidae 5				7	2
			Curculionidae 6		8			2
			Curculionidae 7	8				2
			Curculionidae 8	8				2
			Curculionidae 9				7	2
		Scarabaeidae	*Taurocerastes patagonicus*	50	50	60	7	42
		Staphylinidae	Staphylinidae 1		8			2
	Hymenoptera	Formicidae	*Pogonomyrmex vermiculatus*	100	100	100	13	78
			Dorymyrmex antarcticus	100	100	100	100	100
	Orthoptera	Acrididae	Acrididae 1		8	20		7
			Acrididae 2	33	8	20	20	20
			Acrididae 3	8	17			6
ARACHNIDA	Scorpionida	Bothriuridae	*Urophonius granulatus*	67	58	40	20	46
	Solifuga	Mummucidae	Mummucidae 1	42	33	50	53	45

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