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Spider communities in urban green patches and their relation to local and landscape traits

Carina I. Argañaraz[1] · Gonzalo D. Rubio[2] · Raquel M. Gleiser[1,3]

Abstract Urbanization and urban landscape characteristics greatly alter plant and animal species richness and abundances in negative and positive directions. Spiders are top predators, often considered to be sensitive to habitat alteration. Studies in urban environments frequently focus on ground-dwelling spiders or on spiders in built structures, leaving aside foliage spiders. Effects of habitat, landscape type and structure and local characteristics on spider species composition, richness and relative abundance were evaluated in urban green patches in a temperate city of South America. We also assess whether Salticidae could be an indicator group for the broader spider community in the urban environment. Spiders were sampled with a G-VAC (aspirator) in urban green patches in Córdoba city, Argentina, in urban, suburban and exurban habitats (18 sites; six per habitat) and local and landscape traits were assessed. Overall, the exurban was richer than the urban habitat, however, at the site level Salticidae richness and abundance (but not the total spider assemblage) were significantly lower in urban sites. Species composition moderately differed between urban and exurban sites. Results indicate that on urban green spaces a low impervious surface cover, a coverage of trees, herbaceous vegetation and a vertical structure of vegetation at least up to 1 m in height contribute to higher richness and abundance of spiders, Salticidae being more sensitive than the overall spider community to

✉ Carina I. Argañaraz
c.arg.bio@gmail.com

✉ Raquel M. Gleiser
raquel.gleiser@unc.edu.ar

1 Universidad Nacional de Córdoba- CONICET, IMBIV, Centro de Relevamiento y Evaluación de Recursos Agrícolas y Naturales (CREAN), Av. Valparaíso S/N. CC 509, 5000 Córdoba, Argentina

2 Estación Experimental Agropecuaria Cerro Azul (EEA Cerro Azul), INTA, Ruta Nacional 14, Km 836, Cerro Azul, CP 3313 Misiones, Argentina

3 Universidad Nacional de Córdoba, Facultad de Ciencias Exactas, Físicas y Naturales, Av. Vélez Sársfield 299, Córdoba, Argentina

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local effects. In addition, Salticidae richness can predict 74% of the total spider richness recorded and may be used as spider diversity bio-indicators in this climatic region.

Introduction

Urbanization phenomena are relatively recent in human history and while most natural ecosystems are reduced, cities continue to grow every year (United Nations 2008). As cities grow and become industrialized, natural habitats are fragmented and the remnants are altered and become isolated, surrounded by a matrix of built structures (buildings, roads, etc.). Patterns of city growth (such as increased impervious surface) and side effects (e.g. pollution, changes in natural biogeochemical cycles and climate conditions) are similar in different parts of the world, and thus cities tend to be more similar to each other than compared to the environment where they are immersed (Alberti 2005; Pickett et al. 2011). Homeowner choice and socioeconomic condition also influence landscaping aesthetic of urban green spaces, in turn producing different patterns of plant diversity and density within the urban ecosystems (Walker et al. 2009).

Although urbanization is broadly related with biodiversity loss, plant and animal species diversity and abundances are greatly altered in both negative and positive directions (Faeth et al. 2011; Aronson et al. 2014; Lowe et al. 2017; Meineke et al. 2017). For example, an analysis of bird and plant diversity data from cities of varied human population sizes and establishment dates on six continents, showed that the density of species (i.e. the number of species per km^2) in cities was substantially lower compared with non-urban levels (Aronson et al. 2014). Another study of all spontaneously occurring vascular plant species in 45 settlements of three different sizes and disturbance regimes in Eastern Europe indicated that larger urban settlements had higher richness due to native and human introduced species, but settlement centers with intense regular disturbances held the lowest species richness as opposed to other habitat types with irregular and weaker disturbances (Ceplová et al. 2017).

We decided to study the effects of urbanization on spiders because they are top predators, which are often considered to be sensitive to fragmentation (Gibb and Hochuli 2002). Cities house large numbers of insects, some of them of medical or sanitary relevance, thus predators such as spiders offer ecological services, contributing to their population control (Weterings et al. 2014). The habitat preferences and dispersal abilities of spiders may vary in response to land use, intensity and type of habitat management practices, so they may be considered bio-indicators of habitat quality or anthropogenic disturbance (Maelfait and Hendrickx 1998; Cardoso et al. 2004; Pearce and Venier 2006; Hore and Uniyal 2008).

Recent work on urbanization and spiders show a diverse array of responses, ranging from no significant differences in the richness or abundance of ground-dwelling spiders along rural–urban gradients dominated by coniferous forest in southern Finland (Alaruikka et al. 2002) to increases in spider richness when disturbance or the degree of urbanization increased in forest regions dominated by English oak (Debrecen, Eastern Hungary) (Magura et al. 2008, 2010; Horváth et al. 2012), or beech (island of Zealand, Denmark) (Horváth et al. 2014). On the other hand, in the more arid environment of AZ, USA, spider

abundance increased and richness decreased through a gradient of productivity (Shochat et al. 2004). More recently, Moorhead and Philpott (2013) studied vacant lots, gardens and forests in Ohio City and found divergences in spider family composition but not in richness between habitats. Changes in species composition were also reported by Horváth et al. (2014) in Denmark. Kaltsas et al. (2014) observed in Heraklion, Greece, a decrease in the abundance and richness of Gnaphosidae although not statistically significant, and changes in species composition, such as a higher percentage of generalist species in the urban area.

Studies in urban environments frequently focus on ground-dwelling spiders collected with pitfall traps, leaving aside foliage spiders. This large group of spiders may respond differently to anthropic habitats, either negatively affected by reduced green habitat availability, or alternatively taking advantage of built structures in the absence of plants (Dahirel et al. 2017). In the Sidney area, Australia, translocation experiments showed that the orb-weaving spider *Nephila plumipes* were more successful in terms of establishment and persistence in sites with more urban cover (i.e., with more impervious and less vegetation cover) which were associated with increased prey abundance (Lowe et al. 2016).

In South America there is a growing interest in the urban spider fauna. In Brazil there is a high concentration of spider records around major cities (Oliveira et al. 2017), and although there are studies of urban spiders (for example Brazil et al. 2005; Candiani et al. 2005; Dias et al. 2006), most research in this tropical and subtropical region focus on natural remnants and other human altered habitats, or aim at species of medical relevance (Fischer et al. 2011). The effects of urbanization on spider communities in temperate cities of South America have been less described. In Chile, 31 out of 809 listed species were identified as synanthropic based on sporadic collections in urban areas in different part of the country, published records and museum specimens (Taucare-Ríos et al. 2013). Zapata and Grismado (2015) compiled taxonomic information of spiders collected in Reserva Ecológica Costanera Sur, Buenos Aires city, Argentina, a coastal green land reclaimed from the La Plata river in the 1970s. At least 36% of the morphospecies could not be identified and may potentially be new species for science. We recently described *Neonella acostae* Rubio et al. (2015), a new Salticidae found in green open patches of Córdoba city, Argentina. Also in Córdoba, no clear effects of habitat type (categorized as urban, suburban, and external) were detected on richness, abundance or species composition of Thomisidae, suggesting that local factors may be more relevant for this family (Argañaraz and Gleiser 2017). The present study was carried out in Argentina to address habitat as well as landscape effects on spider diversity patterns in a southern template region. Specifically we assessed whether there are significant effects of habitat, landscape type and structure and local characteristics on the species composition, richness and relative abundance of spiders in green urban patches.

We also sought to assess if Salticidae could be a useful model or bio-indicator group for spiders in the urban environment, especially considering that it is much more labor intensive to do a broad spider sample. The taxonomy of Salticidae in terms of species descriptions is better known compared to other families, especially in South America and more so in Argentina (Rubio 2016; Metzner 2017; Proszynski 2006), which may reduce the errors and problems related with difficulties in the correct taxonomic identification (as discussed by Bortolus 2008). Salticidae, commonly known as "jumping spiders", present three other main characteristics that support to consider this group as an appropriate model for studies on community ecology and biodiversity in urban environments (Coddington and Levi 1991; New 1999; Rubio 2015). This family has the highest number of species within the Order Araneae, a typical mega-diverse group with at least 228 species in Argentina (Catálogo de Arañas de Argentina 2017), and high richness also in tropical or subtropical

regions. Its abundance and species composition are affected by the structural complexity of the vegetation, being selective of the site and microhabitat to catch and consume its prey (Hatley and MacMahom 1980; Cumming and Wesolowska 2004; Tews et al. 2004; Tsai et al. 2006). Most species are hunting-aerial-runners (Höfer and Brescovit 2001) or hunting stalkers (Uetz et al. 1999), easily located and collected due to their conspicuity and high abundance in the ecosystems. If there are direct relationships between the richness and abundance of the spider community and that of Salticidae, focusing on Salticidae as a study model in urban environments should provide information useful for a more general understanding of how cities can affect biodiversity.

Materials and methods

Study area

The study area is Córdoba city (31°25′S; 64°11′W), Argentina, located in the Espinal ecoregion (Brown et al. 2006). The region has been historically subjected to intense anthropogenic disturbance and modifications including deforestation, urbanization and agriculture. Córdoba grows following a "dispersed patches" model (Forman 2014), where the urbanization intensity ranges from patches of natural/natural remnant landscape or agricultural landscape to patches of diffused urbanization (urban sprawl) on to the dense urbanization in the center of the city.

   We defined three concentric areas encompassing habitat types according to urbanization level (Fig. 1a). Within each concentric area, six green patches were randomly selected that were accessible for sampling and had ≥ 70% vegetated surface and ≥ 30% tree cover based on visual interpretation of Google Earth images. Habitat types were: (1) Exurban: landscape comprising a combination of native and non-native vegetation patches, with scarce or no houses (Fig. 1b); (2) Suburban: a mosaic of natural remnant areas within an agricultural matrix and low density urbanization (low built areas, ≤ 10 houses per block) (Fig. 1c); (3) Urban: highly built areas (> 10 houses per block) with scarce natural remnant patches (Fig. 1d). A sampling site within a habitat type was defined as a 2500 m^2 area within a same area or larger green patch. At a local scale, in general exurban sites were more similar, mostly covered with wild vegetation (grasses, herbaceous plants and shrubs), scattered native and naturalized exotic trees, and low human intervention (Fig. 1e); tree species included representatives of the Espinal ecorregion such as *Prosopis nigra*, *Acacia caven* (Molina) Molina and *Geoffroea decorticans* (Gill. Ex Hook. & Arn.) Burkart 1949, while frequent exotic trees and scrubs were *Melia azedarach Crataegus* Tourn. Ex L. and *Ligustrum lucidum*. Urban and suburban sites ranged from vacant-unmanaged lots (Fig. 1f) to parks (Fig. 1g), with a mixture of ornamental and native vegetation, such as *Jacaranda mimosifolia* D. Don 1823, *Acacia*, *Tabebuia* Gomes ex DC. and moderate to high human intervention.

Local habitat and landscape data collection

At a local scale, the vegetation structural complexity of each sampling site (from the ground up to 2 m high, the heights sampled) was quantified by measuring the vertical stratification of vegetation (or vegetation density), following Huang et al. (2011) with minor modifications. Briefly, a 1 × 1 m white board held by a person was used as the
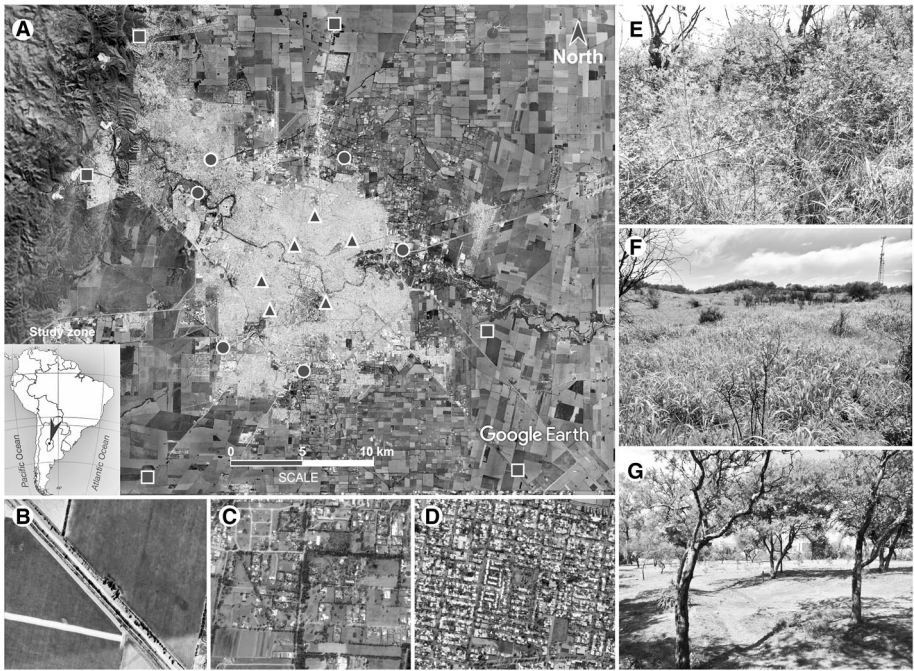
Fig. 1 Location of study area and typology of each sampled site. **a** Triangles represent location of urban, circles location of suburban and squares exurban sites. **b–d** Represents typical landscape urbanization levels: exurban, suburban and urban respectively. **e–g** Represents typical sites at local scale on exurban, suburban and urban sites, respectively

background to estimate vegetation density in front of it, from pictures taken with a Nikon 5300 digital camera by another person standing at 2 m from the board. Pictures were taken facing the four cardinal directions in each of four points within the sampling plot, from ground to 100 cm and from 100 to 200 cm. The photographs were transformed into red (no vegetation) and green (vegetation) images using Adobe Photoshop CS 8.0.1 software, and vegetation cover (expressed as percentage from total surface) was obtained with IDRISI-Selva software. Data from the four points, two heights and sampling seasons were averaged (spring and summer) as the vertical stratification/vegetation density of the site. Land cover type (bare ground, litter cover and plant cover) was measured from pictures taken of five 0.5×0.5 m plots with a Nikon 5300 digital camera, with an AF-P Nikkor 18–55 mm lens and 76° angle vision, facing downward (from 150 cm above the ground); each land cover was expressed as a percentage and averaged over the five plots and sampling dates. Canopy cover was also assessed from five pictures taken with the same digital camera facing upward (from 150 cm above the ground). Thus, in all six variables were recorded at a local scale.

At a landscape scale, land cover/land use characteristics (from now on referred as landscape characteristics) were screen digitized from Google Earth images (September 9, 2016; Google Earth 2016; spatial resolution less than 1 m). Landscape characteristics were assessed for circular buffer areas of increasing radii (100, 500, and 1000 m) around each sampling site, represented as a central 50 m radii area that was discounted to avoid overlap with local factors, and expressed as percentage area covered by each landscape category

within each buffer area (CartaLinx software, Clark Labs 1998–1999; IDRISI Selva, Clark Labs 2015). The landscape categories considered were: 1. Remaining forest (natural remnants, areas with some continuous arboreal vegetation, including shrub-land, with high and dense vegetation); 2. Green areas (public green spaces, parks, and other open areas with low vegetation and low impervious surfaces); 3. Crops (perennial or annual crops); 4. HD buildings (high density of built structures per block ($\geq$ 80% built or impervious surface)); 5. LD building (low density of buildings per block ($<$ 80% built or impervious surface)); 6. Recreation (such as Tennis courts, football and rugby fields); 7. Water bodies; 8. Quarries (sand and gravel quarries, typically open unpaved areas, mostly without vegetation). In addition, category richness (number of different categories) and effective number of categories (exponential Shannon–Wiener diversity index [exp(H)] were estimated as relative landscape heterogeneity indexes.

Spider sampling method and identification

Specimens were collected on each site using a garden-vacuum (G-vac) method to suck spiders from the vegetation. The vegetation in a square meter area was sucked during 1 min using a Sthil® vacuum cleaner with a 110 cm long and 12 cm wide tube. On each site and sampling period we collected ten subsamples, five from vegetation at ground level and 5 up to 200 cm above the ground. Ground level samples were always collected on vegetation patches, which may include relatively small patches of bare ground, but not from bare ground per se. Subsamples were scattered throughout each 2500 m^2 area, with a minimum distance of approximately 10 m between two subsamples. In all, we collected 40 subsamples per site (720 total samples) during two seasons (20 subsamples on springtime—November 2013 and 2014, and 20 on summertime—February 2014 and 2015), diurnally within 9 am–5 pm. The pooled material collected from one site was considered as one sample unit (site) for data analysis. Samples were stored in ethanol 80% and spiders were sorted in the laboratory under stereomicroscope. All adult spiders were identified to family and species or morphospecies level, using original published descriptions and revisions for each family group and consulting taxonomic specialist (see acknowledgment section). Few families were identified only to morphospecies level (based on morphological and reproductive characters) due to lack or scarcity of published taxonomic information about them. The immature were excluded from the analysis to avoid over- or underestimation of richness patterns, because their identification is extremely difficult (Sørensen 2004) or not possible since reproductive characters are considered to separate some species. All material collected were deposited in CREAN—IMBIV (CONICET-UNC). Collecting permits were obtained from Córdoba province (Secretaría de Ambiente de la provincial de Córdoba, Dirección General de Recursos Naturales, Área de Gestión de Recursos Naturales) and from Córdoba city Municipality (Dirección de Espacios Verdes de la Ciudad de Córdoba).

Spider diversity estimates and habitat types

Two approaches were used to assess the global richness for each urbanization level: rarefaction models based on individuals and rarefaction based on samples data. Individual-based rarefaction models explicitly account for the relative abundance of species within the sample pool, while sample-based rarefaction curves account for patchiness in species occurrences (Colwell et al. 2004). Species accumulation curves (and 95% confidence intervals) were estimated for each habitat type, using the multinomial model (Colwell et al. 2012; using EstimateS software, Colwell 2013). The empirical conservative criterion of

non-overlap of the 95% confidence intervals was considered to infer significant differences in species between habitat types. The sample coverage (C) estimated completeness of the sample while the coefficient of variation (CV) characterized the degree of heterogeneity among species discovery probabilities.

To assess whether local (site) abundances and richness differed between urbanization levels (habitat types), landscape or local characteristics, richness, C and CV were also estimated for each site. Three data sets were considered: overall spider community (total), spider community excluding Salticidae (minus Salticidae), and only Salticidae species (Salticidae). Expected richness was assessed with Chao 1-bc and Jackknife order 2.

The hypothesis of equality of means in each habitat type was tested by a general linear model on software R (R Development Core Team 2008). Abundance data were transformed to ln (n + 1) to better fit the normality assumption. If the differences between means were significant, Tukey HSD tests were carried out. Total community or minus Salticidae richness relations with Salticidae richness were assessed using simple lineal regressions. All richness, C and CV estimates were done with SpadeR software (Chao et al. 2015).

Assemblage composition

Differences in species composition between habitats types were assessed with non-metric multidimensional scaling (NMDS), using the Bray–Curtis similarity index, to ordinate spider diversity composition within different classes. Differences were corroborated with ANOSIM, a non-parametric test of significant difference between two or more groups (Anderson 2001), based on the Bray–Curtis similarity index and 9999 permutations on R software. Finally, we used SIMPER (Past software, Hammer et al. 2001) to explore differences in species presence and abundance between habitat categories. To assess if species composition similarities between sites differed within habitat types, the C_{1N} index (q = 1) (equal-weight Horn overlap measure), was estimated for each habitat type using 100 permutations (Chao et al. 2015).

Local characteristics data analyses

First, differences between the sites located at the three habitat types in their local characteristics were explored with one way ANOVA. Since local variables (except leaf litter) did not differ between habitat types, further analyses were not discriminated by habitat. The relation between observed richness and abundance (total, minus Salticidae and Salticidae) and local variables were assessed using regressions. To avoid multicollinearity of the variables, Variance Inflation Factors (VIF) of the model were analyzed, and variables with a VIF ≤ 2 were selected; litter and 2 m vertical structure were thus excluded. Next, simple linear regressions were assessed between richness or abundance and the local variables, using function "lm" for observed richness (residuals fit normal distribution) and function "glm" for abundance (assuming a Poisson distribution and log link function; software R). Over dispersion of abundance data was compensated by refitting the model using quasi-poisson residual distribution (Crawley 2015). Then, multiple regressions assessed the unique contribution of a variable on the dependent variable (richness or abundance), once the contributions of the other variables are taken into account (Streiner 2013). For the construction of the multiple model, the explanatory variables were considered fixed and the non-significant parameters were removed from the saturated model (observed richness or abundances = herbaceous + bare ground + tree cover + 1 m vertical

structure) when their exclusion did not increase residual deviance values. The model was introduced in an additive way to simplify and avoid decreasing the degrees of freedom (a model with interaction was tested, but by reducing degrees of freedom drastically, the parameters lose significance).

Landscape characteristics data analyses

First, differences between the three habitat types regarding landscape characteristics at different spatial scales (buffer area of radii 100, 500 and 1000 m) were assessed with one way ANOVA or with Kruskal–Wallis non parametric test (based on fulfillment of assumptions). Then, Spearman non parametric correlations were run to assess if richness or abundance were related with landscape characteristics at different scales. Landscape characteristics considered at each of the three buffer areas were percentage covered by HD buildings, green areas, category richness (number of different categories) and category exp (H). HD building and green areas were selected because they are expected to have a higher impact on spider fauna, were not significantly ($p = 0.054$) correlated with each other and were well represented in most sites. All analyses were done using R software. Multiple models were assessed as described for local variables.

Results

Spider diversity and habitat type

A total of 1377 adult and 8913 immature specimens were collected, the latter representing 86.6% of the total sample. Adults were assigned to 20 families, 71 genera, 60 species and 50 morphospecies (see Table 4 in Appendix 1). Of the total 110 species + morphospecies (from now on referred simply as "species"), 24 were singletons and 16 doubletons (both representing together 36.4% of the total sample). The most frequent families in terms of numbers of species were Salticidae (24), Theridiidae (18) and Linyphiidae (14), while the most abundant were Linyphiidae (33.9% of total specimens), Thomisidae (18.0%) and Salticidae (15.5%). The most abundant species were *Misumenops maculissparsus* (Keyserling, 1891) and *Lepthyphantes* sp1.

For the total spider community, for a given habitat type, richness estimates from individual and sample based rarefaction curves were similar, suggesting that individuals were randomly distributed. Both individual based and sample based (see Fig. 5 in Appendix 2) curves showed the same pattern, i.e. the exurban habitat held a significantly higher number of species compared to the urban habitat, while the suburban was intermediate or resembled exurban habitat. Consistently, Salticidae showed significantly lower richness in urban habitat while no differences were detected between suburban and exurban (see Fig. 6 in Appendix 2). All environments had C values close to 100% (94, 94 and 95% for exurban, suburban and urban habitat respectively) indicating a good representation of the species in the samples. The CV values (2.27, 2.48, and 1.88, for exurban, suburban and urban habitat respectively) indicate a high heterogeneity in species discovery probabilities in the samples.

At the site level, C was above 60% for all sites (Table 1), except Salticidae in urban sites, with only 27% average coverage but with a large relative variation (SE = ± 0.4). In fact, adult Salticidae were less common in urban sites, with no Salticidae detected in some

Table 1 Average site diversity values for each habitat type, expressed as the mean value $\pm$ SE

Data set	Habitat	Coverage	Coefficient of variation	Abundance[a]
Total community	U	0.72 $\pm$ 0.23	1.21 $\pm$ 0.27	3.7 $\pm$ 0.4 (59.33)
	Su	0.82 $\pm$ 0.10	1.60 $\pm$ 0.43	4.36 $\pm$ 0.2 (86.66)
	E	0.82 $\pm$ 0.10	1.44 $\pm$ 0.29	4.38 $\pm$ 0.15 (84.17)
Minus Salticidae	U	0.75 $\pm$ 0.25	1.17 $\pm$ 0.28	3.7 $\pm$ 0.36 (53.33)
	Su	0.80 $\pm$ 0.14	1.51 $\pm$ 0.36	4.17 $\pm$ 0.23 (43.4)
	E	0.83 $\pm$ 0.05	1.33 $\pm$ 0.44	4.14 $\pm$ 0.17 (32.64)
Salticidae	U	0.27 $\pm$ 0.41	0.17 $\pm$ 0.26	2.2 $\pm$ 0.5 (5.33)*
	Su	0.63 $\pm$ 0.34	0.63 $\pm$ 0.59	0.9 $\pm$ 0.6 (13.16)
	E	0.72 $\pm$ 0.22	0.85 $\pm$ 0.69	2.7 $\pm$ 0.2 (16.33)*

U urban, *Su* suburban, *E* exurban

*$p < 0.05$

[a] Data transformed to $\ln(n + 1)$ (mean raw values in parenthesis)

sites. The *CV* for the total community and for minus Salticidae were higher than for the Salticidae, which were closer to 0. The low Salticidae *CV*s mean that all species had similar abundances or equal discovery probabilities in each site. The comparison of the *CV*s of the total spider and Salticidae communities indicates that there were at least some dominant species in the green spaces, but they did not belong to the Salticidae family.

Spider average abundances per site were not significantly different ($p > 0.05$) between habitats, except for Salticidae ($p < 0.05$) that were more abundant in exurban compared to urban sites. Also, significant differences between habitat types were detected only in Salticidae richness (Observed, Chao 1-bc or Jackknife 2) (Fig. 2a–c; see Table 5 in Appendix 1). When using the Salticidae as linear regressor of total community richness, there was a 74% explained variation, indicating that the spider species richness may be estimated based on the richness of Salticidae (Fig. 3a). When Salticidae were removed from the response variable, the explained variation percentage drops to 50% (Fig. 3b).

Assemblage composition

In the two dimensional ordination space of an NMDS considering the total spider community (Fig. 4), urban and exurban sites formed two clearly separate groups, while suburban shared species with both groups. This pattern was consistent with richness and abundance assessments, where the suburban spider assemblage showed intermediate values. Of the species pool (110 spp), 23% of the species contributed 70% of the differences between urban and exurban assemblages (see Table 6 in Appendix 1). Dissimilarities were mainly due to abundance rather than occurrence contrasts; the top species in terms of their relative contributions to the dissimilarities were found in both in urban and exurban habitats, being *Lepthyphantes* sp.1, *P. mneon* and *N. montana* more abundant in exurban, while *M. maculissparsus* was more frequent in the urban habitat.

Within a habitat type, species composition was moderately similar between sites as suggested by C_{1N} similarity values. Similarity between urban sites was 0.60 $\pm$ 0.03 (min–max range of pairwise comparisons 0.55–0.95), between suburban sites was 0.59 $\pm$ 0.02 (0.26–0.88) and exurban was 0.59 $\pm$ 0.02 (0.25–0.85).
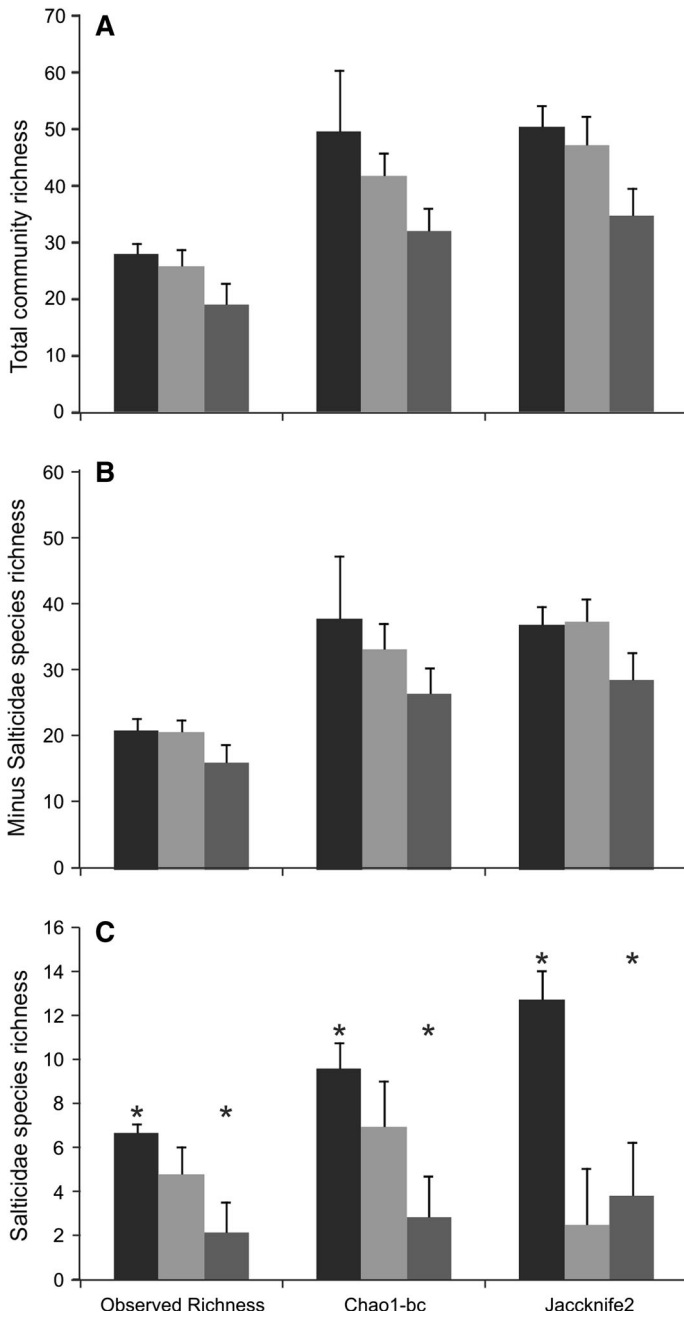
Fig. 2 Average species richness index value + SE of **a** total species, **b** minus Salticidae, **c** Salticidae. Colors indicate habitat type: Exurban (black), suburban (light gray) and urban (dark gray). Within an index, *significant ($p \leq 0.05$) differences between habitat types
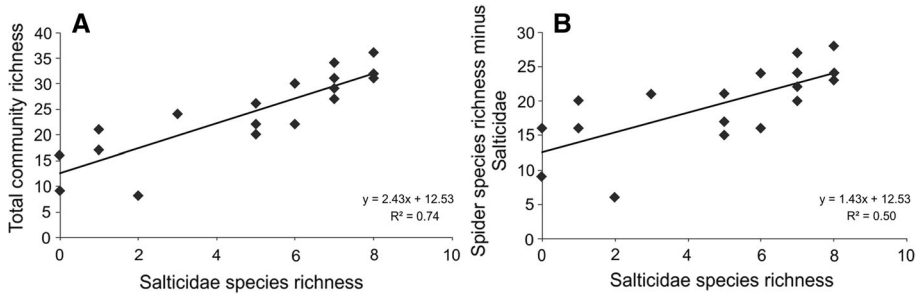
Fig. 3 **a** Total community and **b** minus Salticidae richness in relation with Salticidae species richness
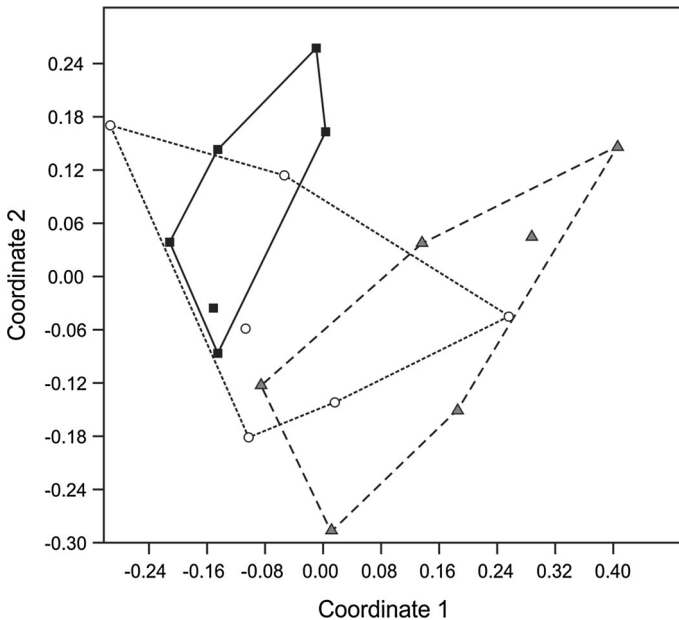
Fig. 4 Non-metric multidimensional scaling of total community spider fauna in Córdoba city. Triangles represent urban, circles suburban and squares exurban sites. Stress is 0.23

Spider diversity and local characteristics

Comparisons of local variables between habitat types indicate that only the leaf litter coverage was significantly different (p = 0.02). Leaf litter coverage was higher in the exurban (28.44 ± 10.89%) than in the urban habitat (8.79 ± 6.19%). The remaining local characteristics did not differ significantly between habitat types.

Spider richness was negatively related to bare ground (see Table 7 in Appendix 1), explaining 43% of total spider community richness and 48% of Salticidae richness found at the study sites. For Salticidae, herbaceous cover and 1 m vertical structure were also significant but only explained approximately 30% richness. When assessing multi-predictor effects, bare ground was retained by all models with the highest weight in all cases,

Table 2 Multiple regressions between spider richness and local variables

Data set	Coefficients	Estimate	SE	t value	p	R^2 adjusted; degrees of freedom
Total species richness	Intercept	35.62	2.97	11.98	< 0.001	0.53;15
	Bare ground	− 0.51	0.11	− 4.56	< 0.001	
	Tree cover	− 0.15	0.06	− 2.39	0.03	
Richness − Salticidae	Intercept	27.25	2.32	11.73	< 0.001	0.44;15
	Bare ground	− 0.33	0.08	− 3.83	0.001	
	Tree cover	− 0.11	0.049	− 2.3	0.03	
Salticidae species richness	Intercept	6.34	1.1	5.82	< 0.001	0.72;14
	Bare ground	− 0.18	0.03	− 5.33	< 0.001	
	Tree cover	− 0.06	0.02	− 2.91	0.01	
	1 m vertical structure	0.1	0.03	3.6	0.002	

Variable %litter was excluded from the independent data set because it was significantly correlated with all remaining variables

followed by tree cover (Table 2). The combined effects of local characteristics explained 72% of the Salticidae richness, but only 44% for the remaining spider species.

When considering abundances (see Table 8 in Appendix 1), Deviance values were higher for Salticidae and the relevant variables were litter cover and bare ground. For the total community abundance, only bare ground was significant. When excluding Salticidae, tree cover was the only significant variable explaining the data variability ($D^2 = 23\%$). When combining variables, bare ground, tree and herbaceous cover were always retained in the models for each of the three spider data sets, with negative relations, while 1 m vertical structure of vegetation had a positive effect on Salticidae abundance (Table 3).

Spider diversity and landscape characteristics

As expected, HD buildings cover at any of the three buffer areas was higher in the urban habitat compared to the other two (p ≤ 0.02), while green area was higher on the suburban habitat at 1000 m (p = 0.04). On the other hand, landscape category richness was higher for the suburban compared to urban or exurban at 500 m and 1000 m radii. Besides, the suburban habitat tended (p = 0.06) to a higher heterogeneity [exp(H)] (mean ± SE and range, 1.88 ± 0.77, 1.97) at 1000 m, compared to urban (1.3 ± 0.11, range 0.23) and exurban (1.7 ± 0.37, 1.10).

Positive correlations (p < 0.05) were found at the 1000 m buffer between green area and total spider community or minus Salticidae richness (see Fig. 7 in Appendix 2) and abundances. Salticidae richness was negatively correlated (p = 0.03 in all cases) with HD buildings within 500 m and 1000 m buffer areas, while abundance was only correlated at the 1000 m buffer. For each of the three spider data sets, land cover heterogeneity (exp(H)) was positively correlated with species richness (for 500 and 1000 m radii) (see Fig. 8 in Appendix 2). These results indicate that both a less built surface and higher landscape heterogeneity would increase spider richness. No significant multi-predictor effects were detected.

Table 3 Multiple regressions between spider abundances and local variables

Data set	Coefficients	Estimate	SE	t value	p	Null deviance; residual deviance; degrees of freedom
Total abundances	Intercept	6.85	0.86	7.9	< 0.001	404.97;158.77;14
	Herbaceous	− 0.02	0.01	− 2.13	0.05	
	Bare ground	− 0.04	0.01	− 4.15	< 0.001	
	Tree cover	− 0.02	0.005	− 3.59	0.003	
Abundances minus Salticidae	Intercept	6.43	0.94	6.79	< 0.001	368.49;155.76;14
	Herbaceous	− 0.018	0.01	− 1.65	0.11	
	Bare ground	− 0.035	0.01	− 3.34	0.004	
	Tree cover	− 0.023	0.01	− 3.68	0.002	
Salticidae abundances	Intercept	5.77	1.31	4.4	< 0.001	183.07;48.44;13
	Herbaceous	− 0.03	0.01	− 2.29	0.04	
	Bare ground	− 0.1	0.01	− 5	< 0.001	
	Tree cover	− 0.02	0.01	− 2.51	0.025	
	1 m vertical structure	0.02	0.01	1.96	0.07	

Variable % litter was excluded from the independent data set because it was significantly correlated with all remaining variables

Discussion

Species diversity and habitat type

Cities are complex ecosystems; urban core, suburban and exurban areas farthest from the core have different stressors that generate pressure on the communities that inhabit them (McDonnel and Pickett 1990; Pickett et al. 2011). If we consider urbanization intensity as an indication of habitat disturbance (encompassing built-impervious surface but also more traffic, higher human activity and related consequences, etc., Pickett et al. 1989), according to the increasing disturbance hypothesis (based on Gray 1989) we expected to find lowest species richness in green patches on the city center, intermediate richness on suburban areas and highest richness in exurban habitats. Our results supported this hypothesis on the total community spider richness but are in contrast to other spider studies, such as Alar-uikka et al. (2002) in Finland, who did not find differences in species richness, Magura et al. (2010) and Horváth et al. (2012) on Debrecen, Hungary, who found richness increased from the rural to the urban sites, or Vergnes et al. (2014) who showed abundances and spiders' richness values were highest at intermediate urbanization along an urban–rural gradient in Paris.

At a habitat scale (i.e., pooling data from all sites within a habitat) our results show that species richness is influenced by the habitat type where the green space is located, as a larger number of species were found in the exurban compared to the urban habitat (see Fig. 5 in Appendix 2). However, the mean total community spider richness and abundance per site did not differ between habitats. Although these results may seem contradictory, they may actually reflect a higher local heterogeneity in species assemblages in exurban compared to urban sites. In fact, although the overall community similarity was approximately 60% for each of the three habitat types, similarities between pairs of sites from the

urban habitat ranged from 0.55 to 0.95, while for the exurban habitat ranged from 0.25 to 0.85. These results are consistent with previous reports of richness by itself not being a good indicator of urbanization effects on spider diversity (noted for example by Magura et al. 2010; Varet et al. 2011).

It is interesting that spider species richness of suburban sites did not differ from exurban sites. The similarity between the richness and abundance of spiders in exurban and the suburban patches may be likely due to both habitats granting similar conditions (such as a higher green area cover) and closer distance for dispersal compared to urban core parks. On the other hand, the spider fauna of the exurban habitats may actually be impoverished because these patches are fragmented and mostly surrounded by extensive agricultural crops field, exposed to frequent disturbances such as tillage, chemical treatments, and harvesting (Horváth et al. 2015; Mader et al. 2016).

Varet et al. (2011) studying hedgerows in France, found for the spider guild better represented in their pitfall samples (open-habitat species, ambushers and ground runners) higher activity-density on rural compared to urban or suburban sites. This type of response coincides with that of Salticidae (mainly diurnal hunting species, Cardoso et al. 2011) in Córdoba, which showed to be susceptible to the habitats explored, being richness and abundances higher in exurban compared to urban sites. This can be attributed not only to the relative location of the green area in the city, but also to the local characteristics of each site. Salticidae are active searchers, walking and stopping to look around for preys (Richman and Jackson 1992). Although Salticidae also disperse by ballooning, there is scarce quantitative information for their ballooning activity and this family is assumed to have medium vagility (based on their relative abundance in aerial samples and their ecology) (Szymkowiak et al. 2007; Jiménez-Valverde et al. 2010) compared to pioneer species such as Linyphiidae or Araneidae. These characteristics may explain their reduced abundance and richness in the city core, where larger impervious and built surfaces may hinder dispersal. Vergnes et al. (2014) also hypothesized that higher built-in areas along urban–rural gradient decreased dispersal capabilities of several species.

An alternative explanation of the lower Salticidae abundance and species richness within core city green spaces may be linked to higher habitat disturbance. City parks are covered mostly by planted tree and shrub species, and are regularly maintained by gardening, hence litter cover (removed during maintenance) may be a proxy disturbance variable. In public green spaces such as those studied here in Córdoba city, mowing takes place approximately once a month during the spring and summer. Maintenance activities such as mowing are a disturbance and may directly affect Salticidae, since, for example tillage in agroecosystems has a direct influence on predatory arthropods such as spiders (Hanson et al. 2017). In fact, in our study sites, the percentage of litter cover—a local variable positively related with Salticidae abundance- was lower in urban green spaces.

Spider diversity and local characteristics

Regardless of habitat where the site was located, minor but significant effects were detected of local land cover characteristics. Results suggest that low bare ground cover and open well-structured vegetated spaces have a positive effect on spider communities in terms of abundances and richness in the urban green spaces of Córdoba. Although vertical plant structure is regarded as important to explain spider community structure (Rypstra et al. 1999), most local variables assessed were weakly related with spider richness and abundance, being bare ground the most relevant. This variable has been reported as significant in other studies such as Horváth et al. (2015) on unmanaged grasslands. Bare

ground cover explains over 42% of the richness and 21% of abundance variation within the total spider community sampled, and the combined effects of bare ground, tree cover and 1 m vertical structure explained 72% of Salticidae richness. The higher bare ground cover percentage, the lower the number of species and abundance. The causes of this pattern may be complex and mainly linked to reduction of suitable habitable surface (McKinney 2008). Litter cover only was significant to improve Salticidae abundances and opposed to bare ground, it increases structural complexity of the environment, which has shown to be positively related with spider abundance (Bultman and Uetz 1984). Litter cover is a factor that plays an important role in soil arthropods, for example as nutritional base and habitat space (Bultman and Uetz 1984) and is conditioned by the vegetation present in the sites (Kremen et al. 1993), consistently Salticidae was both related with litter and 1 m vertical plant structure. Salticidae was one of the dominant spider families in leaf litter samples collections from three urban forests of Sao Paulo Brazil (Candiani et al. 2005). We can also speculate that a reduced habitat availability increases intraspecific competition and exposure to predators, and decreases potential prey availability (Polis and Hurd 1995; Gunnarsson et al. 2009).

Spider diversity and landscape characteristics

Landscape characteristics within 100 m buffer areas did not have an evident effect on Salticidae, while a higher HD buildings cover within 500 and 1000 m buffers had negative effects. These relations may be related with the dispersal capacity of the different species (Gardiner et al. 2010) and suggest that for Salticidae buildings may act as dispersal barriers. Meanwhile, total community spider richness and abundance were not related with landscape characteristics at any of the three buffer areas, which may be due to the inclusion of pioneer species with high dispersal capacity. Therefore, we can hypothesize that the vagility of spiders may be a relevant feature for their establishment in the urban environment. A similar hypothesis was formulated by Varet et al. (2013), who expected younger urban areas to be dominated by generalist species with higher dispersal power. They compared spider collections with pitfall traps from younger (10 year old) and older (30 year old) building age and found that spider assemblages were similar and mainly composed of generalist large individuals (though generally assumed to have low dispersal power), suggesting that colonization is relatively rapid in urban areas.

The lack of effects of landscape variables on total species richness or abundances does not exclude other effects of urban-driven habitat and biotic interaction modifications, such as changes in species traits (Lowe et al. 2014; Alberti et al. 2017), which were not considered in this work. In orb web spiders in Belgium, neither the number of species nor Shannon-based evenness values changed with urbanization level at different scales, but in turn intraspecific variations in web traits such as web vertical inclination or web height were detected in response to urbanization at local and landscape scales (Dahirel et al. 2017).

Conclusions

There are several arguments for conservation and enhancement of biodiversity in urban areas, ranging from ecological services, resources, ecosystem function, to merely aesthetic criteria (Murphy 1988). Urban green spaces should be structured to support a good

diversity of predators such as spiders, as they can mitigate the effect of the reduction of primary production and consequently impoverishment of the trophic network (Shochat et al. 2006). Based on the positive relation between green area and spider richness, and the consistently higher richness of green patches on 1000 m buffer areas with more than 20% green cover, our results suggest that urban patterns including at least 1 ha green areas every 5 ha and heterogeneous landscapes (i.e., with a higher effective number of land cover categories) contribute to more diverse spider assemblages in this context. In addition, our results indicate that on urban green spaces a low impervious surface cover, a good coverage of trees, herbaceous vegetation and a vertical structure of vegetation at least up to 1 m in height contribute to higher richness and abundance of spiders.

Global species richness is one of the key measures of biodiversity. Besides uncertainty in detecting all species (Chao and Colwell 2017), for several taxa and/or regions many species are unknown or processing resources may be limited. Higher taxa surrogates, such as genera for spiders, and the use of indicator (or surrogate) groups of overall richness have been proposed (e.g. Cardoso et al. 2004). This study showed that jumping spider species can predict the richness of the rest of the spider community with 50% accuracy in the current setting, evidencing that Salticidae species richness and abundance are indicators of total community spider richness, and moreover, they are more sensitive to urban landscape effects at local and broader scales.

Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

Appendix 1

See Tables 4, 5, 6, 7 and 8.

Table 4 Total abundances of adult spiders collected within each habitat category in Córdoba city Argentina

Family	Genus/sp/msp	U	Su	E
Anyphaenidae	*Arachosia bergi*	0	2	0
	cf. *Patrera* sp1	0	1	1
	Sanogasta minuta	5	7	1
	Sanogasta maculosa	1	0	0
	Teudis sp1	0	2	2
	Xiruana hirsuta	1	0	0
Araneidae	*Alpaida holmbergi*	3	0	2
	Alpaida versicolor	0	1	0
	Araneus aurantiifemuris	0	0	1
	Araneus omnicolor	0	2	3
	Araneus sp1	0	0	1
	Argiope argentata	0	0	4
	Larinia tucuman	2	11	3
	Metepeira cf. *gressa*	0	2	9
	Metepeira sp1	1	1	0
	Ocrepeira lurida	1	1	3
Coriniidae	*Castianeira coquito*	1	1	5
	Castianeira sp1	0	1	2
Dictynidae	Msp1	0	2	0
	Msp2	0	1	2
	Msp3	1	7	0
	Msp4	3	7	0
	Msp5	5	11	2
Eutichuridae	*Cheiracanthium inclusum*	1	1	5
Gnaphosidae	*Apopyllus silvestrii*	2	1	0
	Micaria sp1	3	0	0
Hahniidae	cf. *Intihuatana* sp2	4	4	9
	cf. *Intihuatana* sp3	0	5	3
Linyphiidae	*Agyneta* sp1	9	20	2
	Dubiaranea difficilis	0	0	1
	Erigone sp1	20	23	14
	Laminacauda montevidensis	6	3	3
	Lepthyphantes sp1	48	124	105
	Mermessus sp1	1	5	1
	Moyosi prativaga	2	9	0
	Notiohyphantes meridionalis	11	6	2
	Ostearius melanopygius	2	3	3
	Pseudotyphistes sp1	17	3	1
	Scolecura propinqua	6	0	1
	Sphecozone rubescens	1	0	0
	Msp1	0	0	1
	Msp2	0	0	1

Table 4 continued

Family	Genus/sp/msp	U	Su	E
Lycosidae	*Navira naguan*	0	0	2
	Pardosa flammula	1	2	0
	Pardosa plumipedata	1	1	4
Mysmenidae	*Mysmena* sp1	0	1	1
Oonopidae	*Neotrops* sp1	0	2	0
	Msp1	0	0	2
Oxyopidae	*Oxyopes* cf. *salticus*	0	0	1
	Peucetia rubrolineata	0	0	2
	Mps1	2	1	1
	Msp2	4	1	1
	Msp3	4	2	8
Philodromidae	*Paracleocnemis* cf. *termalis*	14	4	3
	cf. *Cleocnemus* sp1	6	15	21
Pholsidae	cf. *Metagonia* sp1	0	2	0
Phrurolithidae	*Orthobula* sp1	0	2	4
Salticidae	*Akela* sp1	0	1	0
	Aphirape uncifera	3	10	5
	Atomosphyrus breyeri	0	3	3
	Dendryphantes mordax	7	2	1
	Simonurius gladifer	0	0	1
	Dendryphantes sp1	0	2	3
	Gastromicans sp1	1	0	0
	Habronattus sp1	0	1	1
	Hisukattus transversalis	0	0	3
	Metaphidippus sp1	0	0	1
	Neonella acostae	8	1	5
	Neonella minuta	1	3	6
	Neonella montana	5	13	38
	Phiale roburifoliata	1	1	1
	Phiale sp1	0	1	2
	Psecas sp1	0	1	5
	Saitis catulus	0	0	1
	Sassacus cf. *barbipes*	0	1	1
	Sassacus cf. *helenicus*	0	1	0
	Sassacus sp1	0	5	0
	Semiopyla cataphracta	6	29	16
	Simprulla argentina	1	1	1
	Tullgrenella morenensis	3	2	4
	Tullgrenella serrana	0	1	0

Table 4 continued

Family	Genus/sp/msp	U	Su	E
Theridiidae	*Anelosimus rupununi*	0	0	1
	Anelosimus sp1	0	0	1
	Chrysso sp1	3	7	13
	Dipoena pumicata	0	1	12
	Dipoena sp1	0	0	1
	Euryopis spinifera	0	2	1
	Euryopis sp1	0	1	1
	Platnickina mneon	9	26	52
	Steatoda iheringi	1	1	0
	Steatoda ancorata	0	1	0
	Steatoda sp1	1	0	0
	Theridion calcynatum	0	0	2
	Theridion chacoense	0	0	2
	Theridion sp1	2	0	5
	Thymoites sp1	2	2	5
	Thymoites sp2	0	0	14
	Thymoites sp3	0	0	1
	Thymoites sp4	0	2	3
Thomisidae	*Misumenops maculissparsus*	73	66	20
	Misumenops pallidus	10	16	7
	Misumenops sp1	10	7	2
	Misumenops sp2	5	3	2
	Misumenops sp3	7	1	0
	Tmarus cf. *digitatus*	2	1	0
	Tmarus elongatus	2	1	5
	Wechselia steinbachi	1	1	6
Trachelidae	*Meriola arcifera*	2	1	3
	Meriola cetiformis	1	0	0
Trechaelidae	*Paradossenus sabana*	0	2	1

U urban habitat, *Su* suburban, *E* Exurban

Table 5 Result of one way ANOVA of spider richness and abundance at three habitat types, followed by Tukey test where differences were significant (p < 0.05)

Data set	Parameters	By habitat		
		Mean ± SE	p	*F*
Total species	Abundances	$84.16 \pm 15_E$	0.48	0.76
		$86.7 \pm 17_{Su}$		
		$59.33 \pm 19_U$		
	Observed richness	$27.83 \pm 2_E$	0.14	2.23
		$25.66 \pm 3_{Su}$		
		$19 \pm 4_U$		
	Chao1-bc	$49.5 \pm 11_E$	0.25	1.5
		$41.76 \pm 4_{Su}$		
		$32.1 \pm 4_U$		
	Jackknife2	$50.31 \pm 4_E$	0.07	3.13
		$47.26 \pm 5_{Su}$		
		$34.61 \pm 5_U$		
Minus Salticidae	Abundances	$67.2 \pm 13_E$	0.41	0.95
		$73.5 \pm 17_{Su}$		
		$53.3 \pm 16_U$		
	Observed richness	$21.2 \pm 1.7_E$	0.25	1.5
		$20.8 \pm 2_{Su}$		
		$16.2 \pm 3_U$		
	Chao1-bc	$38.1 \pm 9.5_E$	0.46	0.81
		$33.4 \pm 4_{Su}$		
		$26.6 \pm 4_U$		
	Jackknife2	$37.15 \pm 3_E$	0.17	1.98
		$37.55 \pm 3_{Su}$		
		$28.69 \pm 4_U$		
Salticidae	Abundances	$16.3 \pm 3_E^a$	0.034	4.26
		$13.2 \pm 5_{Su}^{ab}$		
		$5.3 \pm 3.5_U^b$		
	Observed richness	$6.7 \pm 0.4_E^a$	0.034	4.22
		$4.83 \pm 7_{Su}^{ab}$		
		$2.16 \pm 3_U^b$		
	Chao1-bc	$9.62 \pm 1_E^a$	0.046	3.78
		$7 \pm 2_{Su}^{ab}$		
		$2.8 \pm 2_U^b$		
	Jackknife2	$12.7 \pm 1.3_E^a$	0.034	4.23
		$9.2 \pm 2.5_{Su}^{ab}$		
		$3.84 \pm 2.4_U^b$		

[a,b] For each parameter, different letters indicate statistical differences between habitats (p < 0.05)

U urban habitat, *Su* suburban, *E* exurban

Table 6 Taxa contributing most to the average dissimilarities (SIMPER analysis) between urban and exurban sites are listed in descending order of contribution, with their mean abundance, their percentage contribution and cumulative contribution to the overall dissimilarity

Species	Av. dissim	Contribution (%)	Cumulative (%)	Mean	
				Exurban	Urban
Lepthyphantes sp1	9.08	11.98	11.98	17.5	8
M. maculissparsus	6.17	8.14	20.12	3.33	12.2
P. mneon	4.91	6.48	26.61	8.67	1.5
N. montana	4.64	6.12	32.73	6.33	0.83
Erigone sp1	2.50	3.30	36.04	2.33	3.33
Pseudotyphistes sp1	2.12	2.79	38.84	0.16	2.83
cf. *Cleocnemus* sp1	2.10	2.78	41.62	3.5	1
S. cataphracta	1.90	2.51	44.14	2.67	1
Thymoites sp2	1.78	2.35	46.5	2.33	0
Chrysso sp1	1.75	2.31	48.81	2.17	0.5
D. pumicata	1.49	1.96	50.77	2	0
M. pallidus	1.42	1.87	52.65	1.17	1.67
P. cf. *termalis*	1.40	1.85	54.51	0.5	2.33
Oxyopidae msp1	1.17	1.54	56.05	1.33	0.66
N. meridionalis	1.13	1.50	57.55	0.33	1.83
N. acostae	1.07	1.41	58.97	0.83	1.33
M. cf. *gressa*	1.05	1.39	60.36	1.5	0
cf. *Intihuatana* sp2	1.02	1.35	61.71	1.5	0.66
W. steinbachi	1.01	1.33	63.05	1	0.16
Misumenops sp1	0.98	1.30	64.36	0.33	1.67
Agyneta sp1	0.94	1.24	65.6	0.33	1.5
T. elongatus	0.84	1.11	66.72	0.83	0.33
L. montevidensis	0.80	1.06	67.78	0.5	1
Thymoites sp1	0.80	1.06	68.85	0.83	0.33
N. minuta	0.78	1.04	69.89	1	0.16
Psecas sp1	0.74	0.97	70.87	0.83	0

Table 7 Simple regressions between local variables and spider richness (only significant results are shown)

Regressors	Total community richness				Minus Salticidae richness				Salticidae richness			
	R^2	β_x	p	SE	R^2	β_x	p	SE	R^2	β_x	p	SE
Herbaceous	–	–	–	–	–	–	–	–	0.32	0.13	0.013	0.05
Bare ground	0.43	– 0.41	0.003	0.12	0.33	– 0.26	0.012	0.01	0.48	– 0.17	0.001	0.04
1 m vertical structure	–	–	–	–	–	–	–	–	0.28	0.11	0.023	0.04

Degrees of freedom = 16

β_x slope parameter, R^2 multiple R-squared, *SE* standard error

Table 8 Simple regressions between local variables and spider abundance (only significant results are shown)
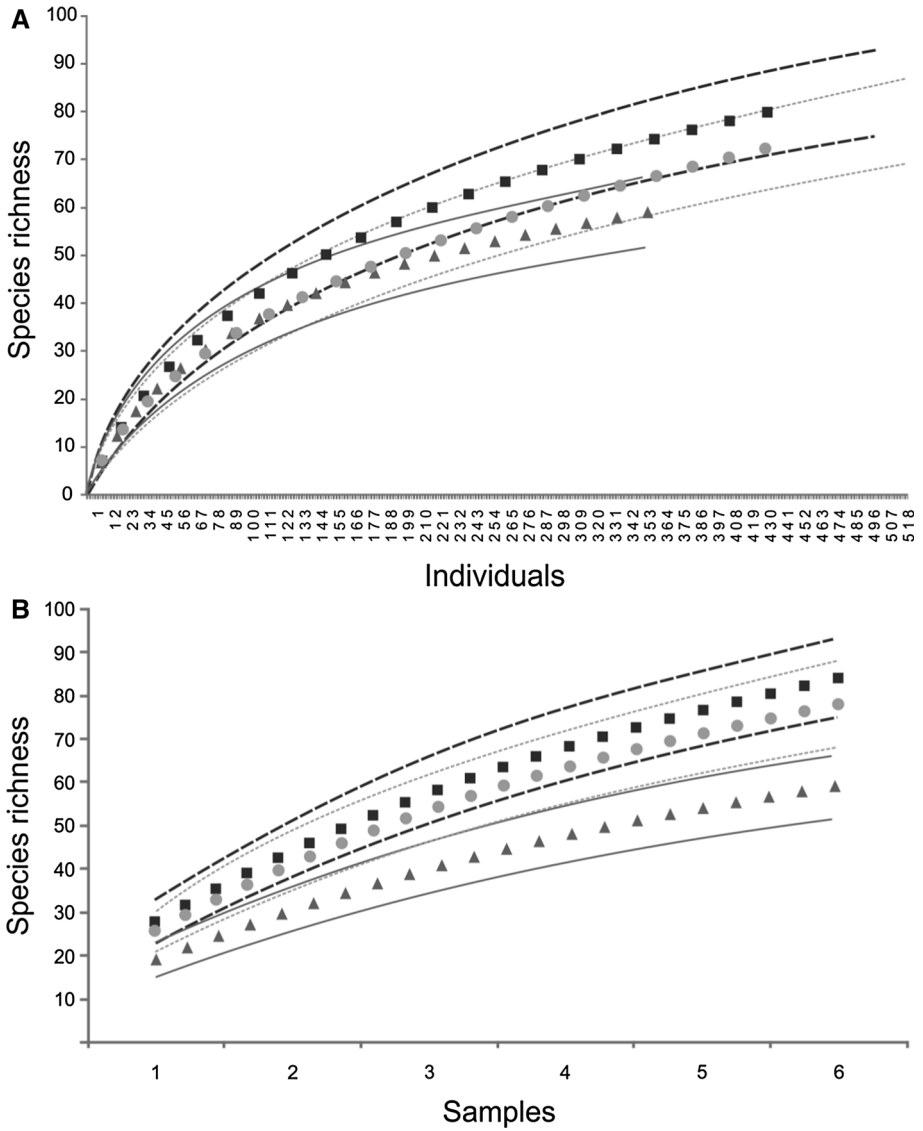
| Regressors	Total abundance				Minus Salticidae abundance				Salticidae abundance			
	D^2	β_x	SE	p	D^2	β_x	SE	p	D^2	β_x	SE	p
Litter	–	–	–	–	–	–	–	–	44.86	0.04	0.01	0.001
Bare ground	21.8	– 0.02	0.01	0.05	–	–	–	–	52.23	– 0.06	0.01	0.001
Tree cover	–	–	–	–	23	– 0.01	0.001	0.05	–	–	–	–

Degrees of freedom = 16

D^2 deviance, β_x slope parameter, *SE* standard error

Appendix 2

See Figs. 5, 6, 7 and 8.

Fig. 5 **a** Individual rarefaction curves. **b** Sample rarefaction curves. Squares line: exurban; circles line: suburban and triangles line: urban. Dashed, pointed and continuous lines represent their respective 95% confidence intervals
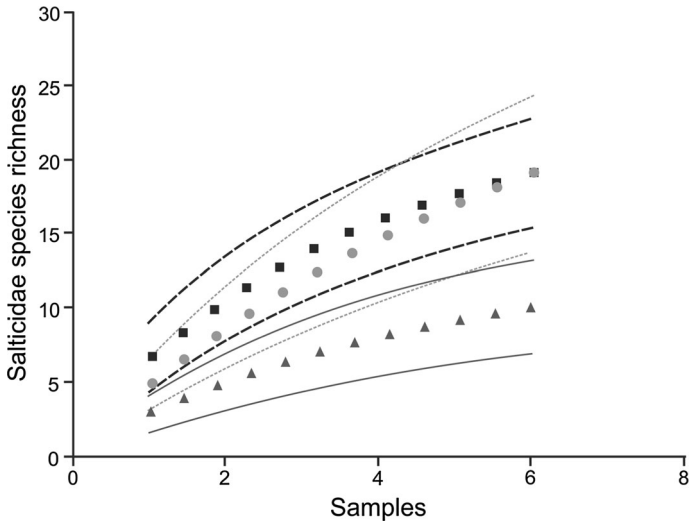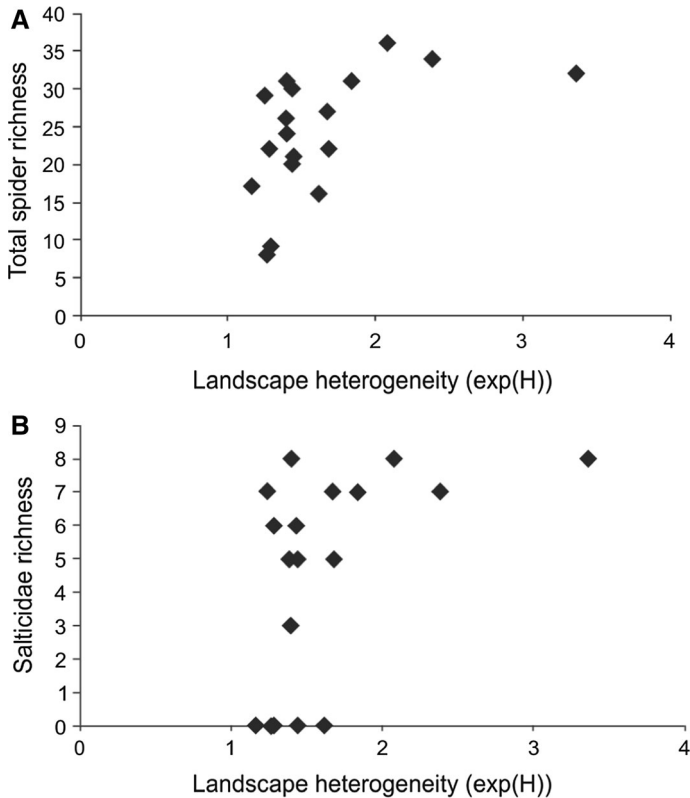
Fig. 6 Sample rarefaction curves using Salticidae richness. Squares line: exurban; circles line: suburban and triangles line: urban. Pointed and dashed lines are respective 95% confidence intervals
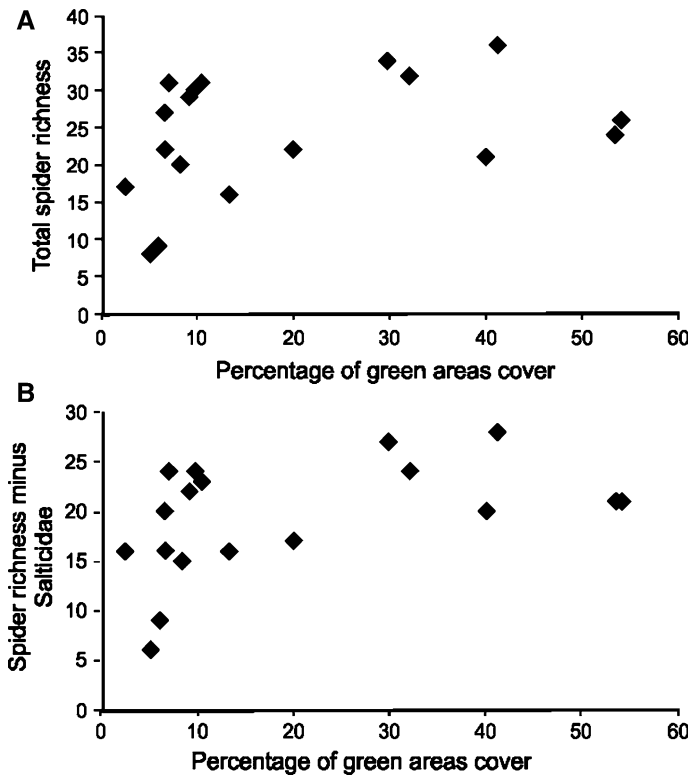
Fig. 7 Relation between green cover within 1000 m buffer area and **a** total spider richness, **b** spider richness minus Salticidae

Fig. 8 Relation between landscape heterogeneity (exp (H)) within 1000 m buffer area and **a** total spider richness **b** Salticidae species richness

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