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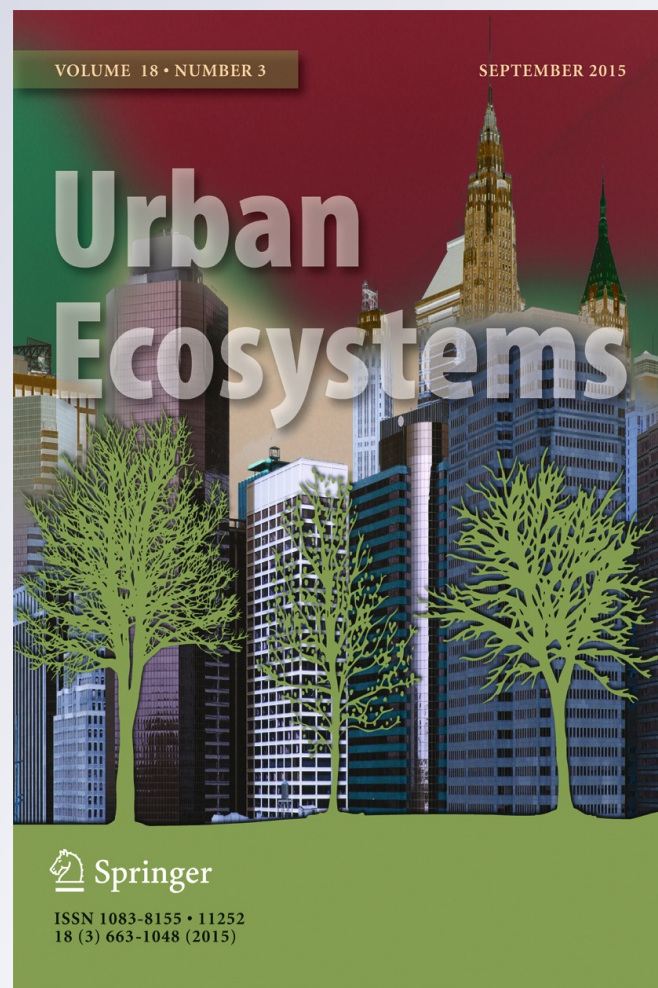
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Urbanization altered latitudinal patterns of bird diversity-environment relationships in the southern Neotropics

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Abstract Given the global expansion of urbanization, it is crucial for planning to understand how that process affects spatial patterns of diversity. At broad geographical scales, climatic conditions such as temperature or rainfall usually explain those patterns. Our objective was to analyze and compare the species richness-environment relationships and the distance decay in similarity of bird communities between urban centres and less intensively modified adjacent rural areas along a latitudinal gradient in the southern Neotropics. We surveyed birds in 15 urban centres and their adjacent rural areas from 26° to 38°S and compiled temperature and rainfall data. We performed regression analyses and Mantel tests to explore latitudinal changes in bird species richness and taxonomic composition as a response to those climatic variables in urban centres and in rural areas. Results showed that species richness decreased with latitude in rural areas, and temperature and rainfall accounted for that decline, but remained relatively constant in urban centres. The difference in species richness between urban centres and rural areas was larger at lower latitudes. Similarity in the composition of bird assemblages declined with distance at a similar rate in both urban centres and rural areas; however, similarity was higher between urban centres than in rural areas at any given distance. Environmental differences due to temperature and rainfall partially accounted for the distance decay in similarity for both urban and rural areas. The impact of urbanization on bird species richness seems to differ according to the climatic context in which urbanization develops, and it is expected to be higher in tropical than in more arid environments. Our study remarks the importance of considering urban systems as components of larger ecological systems.

Keywords Argentina · Biotic homogenization · Community similarity · South America · Species richness · Urban ecology

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Introduction

Urbanization is one of the most severe large-scale disturbances affecting biological communities (Liu et al. 2003, McKinney 2006), and it has become a major conservation issue (McKinney 2002; Miller and Hobbs 2002; Mc Donald et al. 2008). Most studies analyzing the effects of urbanization on biological communities were conducted in Europe and the USA (McKinney 2006; 2008), and some large-scale patterns were described, for example, by Pautasso et al. (2011). In the southern Neotropics, human activities developing in grasslands, savannas, and subtropical forests create a landscape matrix of seminatural-to-rural habitats resulting from the combination of more or less intensive agriculture and livestock; typically, urbanization develops as patches within the rural matrix. Worldwide, the empirical evidence showed that bird communities respond to the urbanization process by changing species richness, composition and relative abundance (Blair 1996; Marzluff 2001; Chase and Wash 2006; McKinney 2008; Garaffa et al. 2009).

At broad geographical scales, bird species richness increases with decreasing latitude (Hawkins et al. 2003). There is consensus that climatic conditions given by temperature or rainfall provide a comprehensive framework to understand and predict those patterns (Rabinovich and Rapoport 1975, Hawkins et al. 2003; Belloq and Gómez-Insauti 2005). A family of hypotheses based on energy-water flow was proposed to explain latitudinal patterns of species richness and the relationships between richness and temperature or rainfall (Mittelbach et al. 2001, Hawkins et al. 2003); the evidence indicates positive associations between bird species richness and both rainfall and temperature (Hawkins et al. 2003). However, climatic conditions can be locally obscured in urban centres due to the development of impervious areas and high irrigation inputs. Urban areas often show higher temperature and water inputs throughout the year than the temperature and irrigation levels of rural areas; regardless of latitude, urban areas constitute “pseudo-tropical bubbles” (sensu Shochat et al. 2006). In the context of both the energy-water hypothesis and the pseudo-tropical bubble condition, local climatic conditions will likely obscure species richness-macroclimatic variable associations along broad (i.e., about 1000 km long) scale latitudinal gradients.

The negative relationship between taxonomic similarity and geographical distance typically reflects the spatial turnover in species composition, or beta diversity, as a response of gradual spatial changes in environmental conditions (Nekola and White 1999). Land transformation due to urbanization promotes global homogenization of local environmental conditions such as landcover or perturbation level. Several studies showed that urban assemblages had higher similarities in species compositions than non-urban assemblages, indicating that urbanization is a major cause of biotic homogenization (Clergeau et al. 2006; McKinney 2006; Sorace and Gustin 2008, Luck and Smallbone 2011). Presumably, the rate of distance decay in similarity will be slower between urban than between rural areas. Furthermore, the internal structure of urban ecosystems seems not to affect taxonomic similarity between biotic assemblages; studies conducted worldwide showed no clear increased in similarity between core urban areas compared with other urban sectors (Jokimäki and Kaisanlahti-Jokimäki 2003, White et al. 2005, Ortega-Álvarez and MacGregor-Fors 2009).

Our objective was to analyze and compare the species richness-environment relationships and the distance decay in similarity of bird communities, between highly modified environments such as the urban centre and less intensively modified environments (adjacent rural areas) along a latitudinal gradient from 26° to 38°S in central Argentina. We ask 1) Do urban centres and rural areas show different spatial patterns of bird richness as a response to climate latitudinal patterns?; and 2) Does community composition remain unchanged along latitudinal gradients in urban centres? We hypothesized that a) there will a stronger correlation and a

steeper slope in the species richness-latitude and the species richness-climate relationships in rural areas than in urban centres, due to additional temperature and water inputs in cities; and b) community composition will be more similar between urban than between rural habitats, and the distance decay in similarity will be steeper across rural areas than in urban centres, because of biotic homogenization.

Methods

Study regions and study design

Urbanization in the southern Neotropics provides a useful testing scenario, and birds a good biological model, to identify and understand differences in spatial patterns of community attributes between seminatural and human-modified habitats. We selected 15 towns along a latitudinal gradient in the southern Neotropics, from 26° to 38°S and 59° to 61°W covering approximately 1200 km in the N-S direction in central Argentina (Table 1, Fig. 1), from forests of the Chaco and Espinal regions to Pampean steppes (Cabrera 1971). In the Chaco region, the climate is warm and dry with mean annual temperature ranging 20–23 °C and precipitations occurring primarily in the summer varying between 500 mm in the West to 1200 mm in the East. The original xerophilous forest that characterizes the region is dominated by *Schinopsis* trees, and is generally converted into savannas or steppes due to fires and land use. In the Espinal region, the climate is warm and wet at the North, with temperature and rainfalls decreasing to the West. Annual rainfall varies from 340 to 1170 mm and mean annual temperature between 15 and 20 °C. The savannas and xerophilous forests are dominated by *Prosopis* spp and were also converted to open areas with scarce trees due to human activities. Climate in the Pampean region is temperate-warm; mean annual temperature ranges 13–17 °C and rainfalls occur all year round (from 600 mm in the South to 1100 mm in the North). Grasses of the Poaceae family dominate the native plant community. Natural forests and grasslands from the study regions were intensively degraded due to wood extraction, agriculture and ranching during the 20th century (Soriano 1991); thus all towns were embedded in agricultural-pastoral matrices (e.g., Fig. 2).

Selected towns ranged between 18000 and 25000 inhabitants, which are common sizes of towns in the study area. Typically, the internal structure of towns showed a 100 × 100 m central park with ornamental (mainly exotic) vegetation (Fig. 2). We selected towns of relative similar size and above the response threshold of 14000 inhabitants (Garaffa et al. 2009), because there is evidence that size influence bird response to urbanization (Jokimäki and Kaisanlahti-Jokimäki 2003; Garaffa et al. 2009; Aronson et al. 2014). To test our predictions, in each of the 15 towns we established six sampling points in the core urban area (impervious area >50 % following Marzluff 2001) and six in the rural zone (<10 % impervious area and >90 % agricultural/pastoral land). The percentage of impervious area was calculated using IKONOS images, in a circle of 100 m radius centered at each sampling point.

Bird surveys and environmental variables

Sampling points were set 200-m apart to avoid double counting of birds (Bibby et al. 1998). At each sampling point, birds were surveyed using a fixed 50-m radius point-count method (Ralph et al. 1996; Melles et al. 2003). All individuals heard or seen were recorded during a 5 min period for the first 3.5 h following sunrise. We surveyed all sites once during the spring. Both single visits and 5 min observation periods are commonly used for bird surveys in urban

Table 1 Study towns and their geographical coordinates, from North to South

Name	Lat (S) / Long (W)
Pirane	25°43'/59°6'
Gral J San Martin	26°32'/59°20'
Quitilipi	26°52'/60°13'
Machagai	26°55'/60°3'
Vera	29°27'/60°13'
San Cristobal	30°18'/61°14'
Sunchales	30°56'/61°34'
Arroyito	31°25'/63°3'
Las Varillas	31°52'/62°43'
San Jorge	31°53'/61°52'
Cañada Gomez	32°49'/61°24'
Firmat	33°27'/61°29'
Lincoln	34°51'/61°32'
Bolivar	36°13'/61°7'
Coronel Suarez	37°27'/61°56'

areas (Jokimäki and Suhonen 1998; Jokimäki and Kaisanlahti-Jokimäki 2003). All observations were conducted by the same two observers.

**Fig. 1** Location of study towns (dots) showing the covered latitudinal gradient



Fig. 2 Aerial image showing the typical structure of Argentinean towns

The climatic variables (mean annual temperature and annual precipitation) were derived from the 10 arc-minutes (~20 km) spatial resolution thematic images, available in the WorldClim database (<http://www.worldclim.org>) (Hijmans et al. 2005). We located all towns in the images using their geographic coordinates and extracted the values for the mentioned variables.

Data analysis

We explored the species richness-latitude and species richness-climate relationships in rural areas and urban centres. To assess whether a latitudinal pattern in species richness actually occurred we first ran simple linear regressions between bird richness and latitude; the distribution of residuals was checked to assess regression assumptions using Infostat software (Di Rienzo et al. 2008). If a latitudinal trend was confirmed for one or both areas, we then ran the correspondent multiple linear regression model with bird richness as response variable and mean annual temperature and annual precipitation as explanatory variables. Using SAM software (Rangel et al. 2010) we also included site geographical coordinates as predictors in the models, and ran partial regressions to assess whether the spatial pattern of bird richness along the latitudinal gradient was efficiently accounted by the environmental variables (i.e., mean annual temperature and annual precipitation) (Borcard et al. 1992). The spatial structure of residuals was analysed through spatial correlograms based on Moran's autocorrelation index (Legendre 1993).

To explore the distance decay in bird assemblage similarity and the relationship between environmental and bird assemblage similarities, we constructed similarity or distance matrices based on geographic, environmental and species data. To estimate species turnover between sites we used the Bray-Curtis quantitative similarity index, based on both species occurrence and relative abundances (Magurran 2004), given that bird abundance is a key component of urban community structure (i.e., few very abundant exotic species, Marzluff 2001). The index ranges from zero to one; an index value of one indicates two urban centres or rural areas sharing all species with the same abundance, and a value of zero indicates two sites with no bird species in common. For the distance decay analysis we ran simple linear regressions with Bray-Curtis similarity index for all pairs of urban and rural areas as the response variable, and

the geographic distance (km) as the explanatory variable. Because similarity values are not independent from each other, we determined the regression probabilities based on 1,000 permutations using the software package Ape in R (Paradis et al. 2012). Using the regressions, we calculated initial similarity (S1) and halving distance (HD) and slopes (Soininen et al. 2007; Thieltges et al. 2009) to compare the similarity–distance relationship between urban and rural zones. Statistical differences between the three parameters of both functions were tested by a permutational procedure following Astorga et al. (2012). We first pooled both urban and rural species matrices and constructed a single one (“big matrix”). Then we randomly permuted the columns (species) of the big matrix and split it in two matrices with the dimensions of the original ones. After that, we proceeded as described above for the original matrices and obtained S1, HD and slope. We then calculated the S1, HD and slope differences between rural and urban areas. This procedure was performed 1000 times to obtain the frequency distribution for the three differences and calculated their significances. The analyses were performed in R (R Core Team 2014). We ran partial Mantel tests for both urban and rural areas to assess the influence of environmental similarity on bird assemblage similarity while controlling for geographic distance (i.e., to distinguish the pure effects of our climate variables on assemblage similarities from other uncontrolled explanatory variables that covary with space) (Legendre and Legendre 1998). Given that Mantel test measures correlation between distance matrices, we converted similarity values to distance measures between sites. We calculated the Euclidean index for geographic distance, Gower index for the environment (as this index is appropriate for variables with different measurement magnitude and unit) and 1-Bray-Curtis similarity index for birds (Legendre and Legendre 1998). The significance of the Mantel tests were conducted using 1000 randomizations through the software package Vegan in R (Oksanen et al. 2007). P-values below 0.05 were considered significant for all tests.

Results

We identified a total of 109 bird species during the surveys. Of that total, four species (3.7 %) were recorded only in urban centres, 70 species (64.2 %) only in rural areas and 35 species (32.1 %) in both land uses (Appendix 1).

Species richness decreased with latitude in rural areas ($R^2=0.656$, $F_{1, 13}=24.779$, $p<0.0001$) but remained relatively constant in urban centres ($F_{1, 13}=2.353$, $p=0.149$) (Fig. 3). The difference in species richness between urban centres and rural areas was larger at lower latitudes. Partial regression analyses showed that latitudinal changes in species richness for the rural areas could be explained by the mean annual temperature and annual precipitation (proportion of explained variation shared by environmental variables and geographical coordinates = 0.653). The correlogram based on Moran’s autocorrelation index showed that residuals did not hold any spatial structure, indicating that mean annual temperature and annual precipitation accounted for the latitudinal variation of bird species richness in rural areas (Fig. 4a). The correlogram for raw bird richness data from urban centres confirmed the lack of a latitudinal trend (Fig. 4b).

Similarity between bird assemblages significantly declined with increasing distance for both urban centres and rural areas (Table 2, Fig. 5). The rate of distance decay was similar for urban centres and rural areas, given that the slopes were not significantly different ($p>0.05$). However, similarity values for urban centres (Su) were higher than for rural areas (Sr) because the initial similarity S1 was significantly higher for urban centres than for rural areas ($p<0.05$). Because resulting lines were parallel (i.e., same slope), the difference between both similarities (Su–Sr) held for every distance between towns. The halving distance did not differ significantly between zones ($p>0.05$).

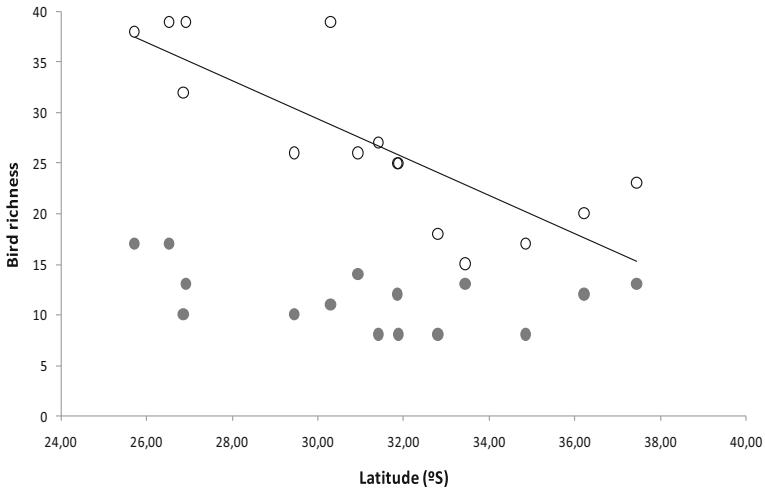


Fig 3 Latitudinal tendencies of bird species richness for urban (*black dots*) and rural (*white dots*) assemblages. Regression details and explanation provided in the text

The partial Mantel tests for both urban and rural areas revealed that the environmental distance based on temperature and precipitation accounted, at least partially, for the distance decay of

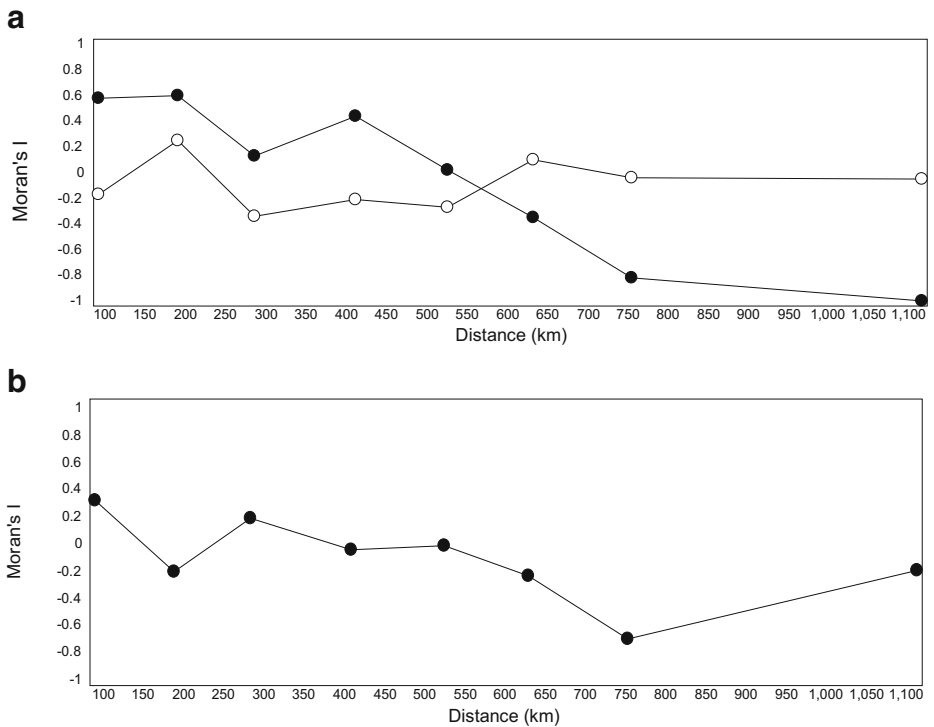


Fig. 4 Spatial correlograms showing the evidence of spatial autocorrelation (i.e., spatial structure) in bird species richness (*black dots*) and residuals after regressing with the environmental variables (*white dots*) for rural **a** and urban **b** assemblages

Table 2 Regression statistics for bird assemblage similarity in species composition as a function of geographic distance between sites. We also provide details on the Halving Distance approach: slope, initial similarity S1, halving distance HD (measure units)

Urban type	Slope	S1	HD (km)	F (1, 103)	p-value
Centre	-0.00024	0.672	1401.3	66.37349	0.001
Rural	-0.00027	0.469	858	105.5396	0.001
p-value for the difference	0.639	0.021	0.263		

similarity. For urban centres, the distance decay of similarity could be accounted by environmental dissimilarity as there was a significant correlation between the species and environmental dissimilarities when controlling for geographic distance, but not vice versa (Table 2). For rural areas, the distance decay of similarity could partially be accounted by environmental dissimilarity as there was a significant correlation between the species and environmental dissimilarities when controlling for geographic distance. There was a significant correlation between species dissimilarity and geographic distance when controlling for environmental dissimilarity indicating that the distance decay in assemblage similarity is also influenced by spatially structured unmeasured factors (i.e., covarying with geographic distance) (Table 3).

Discussion

As we predicted, results showed a stronger correlation and a steeper slope in the species richness-latitude and the species richness-climate relationships in rural areas than in urban centres. Also, assemblages were more similar between urban than between rural habitats. In contrast to our prediction, however, the distance decay in similarity was not steeper across rural areas than in urban centres. Thus, urban centres held a relatively consistent number of bird species independently of town location, but experienced the same gradual spatial changes in assemblage taxonomic composition as rural areas. Nevertheless, rural areas did reflect a change in both bird richness and taxonomic composition depending on the geographic location. Even when the response to urbanization seems to be species specific (Blair 2004), the ecological niches and sustained resource availability offered by urban centres may allow a

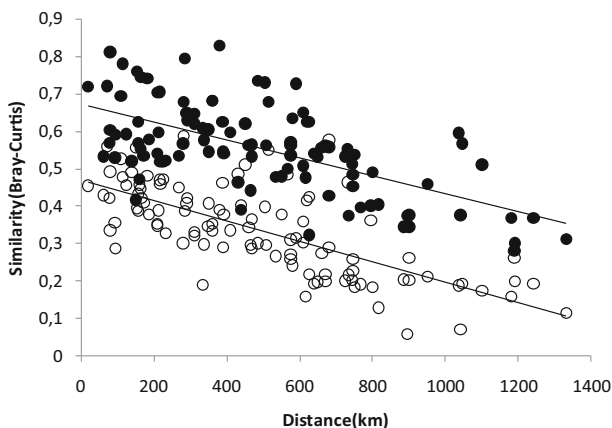


Fig 5 The distance decay in similarity for urban (*black dots*) and rural (*white dots*) assemblages

Table 3 Partial Mantel tests (correlations between distance matrices; climatic distance: Gower index, geographical distance: Euclidean) for urban and rural areas. We provide the coefficient of association between matrices (r) and the level of significance (p -value; $\alpha=0.05$)

Urban type	Mantel tests	
	r	p -value
Centre		
Correlation between assemblage dissimilarity and ... (controlled effect)		
Climatic dis. (Geographical dis.)	0.2429	0.03
Geographical dis. (Climatic dis.)	0.2329	0.056
Rural		
Climatic dis. (Geographical dis.)	0.2777	0.014
Geographical dis. (Climatic dis.)	0.2584	0.02

relatively constant number of species independently of town geographic location; yet, species identities seemed to be determined by climate (i.e., mean annual temperature and annual precipitation). In agreement with McKinney (2006), our results showed that species assemblages in urban environments (“urban-centre adapters”) were mostly composed by ground foraging omnivores (*Mimus saturninus*), frugivores (*Turdus rufiventris*), insectivores (*Furnarius rufus*), granivores (e.g. *Columbina picui*) and also nectarivores that can extract nectar from ornamental plants (e.g., *Heliomaster furcifer*). On the other hand, habitat or food specialists (e.g., grassland granivores *Ammodramus humeralis*, *Nothura maculosa*, *Sporophila minuta*; tree-dependent insectivores *Annumbius annumbi*, *Empidonomus aurantioatrocristatus*, *Synallaxis albescens*) were recorded only in the rural zone (“urban-centre avoiders”). Bird assemblages were more similar between urban centres than between rural areas; however, the taxonomic composition of urban bird assemblages seemed to be influenced by the regional species pool, given that the rate of species replacement was very similar for urban centres and rural areas when moving across regions. Under the paradigm that urbanization is a major cause of biotic homogenization (McKinney 2006, Clergeau et al. 2006, Sorace and Gustin 2008) even retaining endemic native species (Aronson et al. 2014), our results support that idea given that urbanization increased bird assemblage similarities with respect to the rural zone.

Species richness decreased with increasing latitude for the rural areas and those changes could be explained by climate. We provided evidence that urbanization in the southern Neotropics altered the latitudinal patterns of bird species richness observed in less intensively modified areas, presumably through changes in local environmental conditions such as temperature and water inputs. Local productivity is likely the ultimately cause (as proposed by Shochat et al. 2006) that explains why differences in species richness between town centres and rural areas are smaller at higher latitudes. In tropical or subtropical environments (i.e., our northern study sites), primary productivity in towns is reduced by impervious areas compared to the rural zone; in such cases, more species being able to coexist in the rural area than in the town centre could be expected. In more arid zones (i.e., our southern study sites), primary productivity increases in town centres relative to the rural zone due to water irrigation and vegetation management, partially compensating the reduction of vegetation coverage due to paving. Thus, in more arid environments than those studied here, bird species richness may reaching even higher values in town centres than in arid zones could be expected (Mills et al., 1989). Therefore, our study showed that the degree of impact of urbanization on bird species richness will differ according to the climatic context in which urbanization develops. Our results

demonstrated that the impact of urbanization on species richness was higher in the warm and wet northern towns (i.e., subtropical town) than in the temperate and semiarid southern towns.

The rate of distance decay in community similarity was relatively similar between urban centres and rural areas, but similarity values were higher in the former. That finding is consistent with the biotic homogenization hypothesis caused by urbanization (McKinney 2006), because similarity between localities was higher for the most urbanized areas at any given distance. However, species composition of bird communities in urban centres was not independent of the environmental context in which urbanization was developed; we observed that the rate of species turnover was virtually the same for both habitat types, likely reflecting the influence of the regional species pool. Furthermore, our results showed that species composition in town centres was not the result of replacing many native species with a small subset of globally widespread non-native species (in fact, we recorded only two non-native bird species in our surveys); but rather urban centres may be composed by extralimital native species (wide spread species from neighbour regions) as previously found for urban floras (La Sorte and Mc Kinney 2006, Aronson et al. 2014). Similar results were observed in a more local study of bird assemblages in suburban areas and remnant forests conducted in Australia (Catterall et al. 2010). On the other hand, the distance decay of similarity that occurred for both urban and rural areas could be explained by the sharing of climate conditions of the ecoregion where the urban centre was developed.

In their seminal article, Nekola and White (1999) proposed that the rate of decay in taxonomic similarity is determined by environmental characteristics and organism life histories influencing species distribution (niche-based processes) and dispersal capability. Dispersal capability is influenced by both the species ability to move long distances across habitat types and the spatial configuration and nature of landscape elements affecting isolation. Given that environmental similarity decreases with increasing geographic distance between sites (Steinitz et al. 2006), the spatial decay in environmental similarity should lead to the spatial decay in the similarity of species composition. Then, it is expected that community similarity will decay more abruptly in more changing environments (rural zones) than in more stable sites (urban centres). However, our results indicate that rather than being mutually exclusive causes, dispersal limitations (neutrality), niche-based processes, and landscape configuration are most likely jointly driving the rate of distance decay in similarity (Cottenie 2005; Soininen et al. 2007).

Finally, our work demonstrates how human activities developed in large extensions of land (i.e., urbanization) may change widely documented large-scale patterns of diversity such as the decline in species richness with increasing latitude and the levels of composition similarity for any geographic distance. We demonstrated that the impact of urbanization on species richness will differ depending on the geographical position (i.e., region) in which urbanization is developed in the southern Neotropics. According to previous studies conducted at smaller geographical scales in Argentine cities and towns, the design of local green areas and vegetation elements (e.g., remnants of native vegetation) within the urban area play a key role in the maintenance of high species numbers (e.g., Leveau and Leveau 2004; Juri and Chani 2005; Faggi and Perpelizin 2006; Germain et al. 2008). Careful planning should be made at the local, regional and interregional scales to develop urbanization. Our findings provide knowledge that increases our understanding of urban ecosystems developing as part of larger ecological systems.

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Appendix

Table 4 Species identified during the bird surveys indicating the habitat (u: urban centres; r: rural areas) where they were recorded

Species	Habitat
<i>Agelaius cyanopus</i>	r
<i>Agelaius ruficapillus</i>	r
<i>Ammodramus humeralis</i>	r
<i>Anthus chii</i>	r
<i>Anthus correndera</i>	r
<i>Anthus furcatus</i>	r
<i>Anumbius annumbi</i>	r
<i>Aramides ypecaha</i>	r
<i>Athene cucularia</i>	r
<i>Buteo magnirostris</i>	r
<i>Camptostoma obsoletum</i>	r
<i>Carduelis magellanica</i>	r
<i>Cathartes burrovianus</i>	r
<i>Certhiaxis cinnamomea</i>	r
<i>Ceryle torquata</i>	r
<i>Circus buffoni</i>	r
<i>Coccyzus melacoryphus</i>	r
<i>Colaptes campestris</i>	r
<i>Coragyps atratus</i>	r
<i>Coryphistera alaudina</i>	r
<i>Coryphospingus cucullatus</i>	r
<i>Crotophaga ani</i>	r
<i>Crypturellus tataupa</i>	r
<i>Cyanocorax cyanomelas</i>	r
<i>Cyclarhis gujanensis</i>	r
<i>Diuca diuca</i>	r
<i>Elaenia parvirostris</i>	r
<i>Elaenia spectabilis</i>	r
<i>Embernagra platensis</i>	r
<i>Empidonomus aurantioatrocristatus</i>	r
<i>Falco sparverius</i>	r
<i>Fluvicola pica</i>	r
<i>Furnarius cristatus</i>	r
<i>Gallinago gallinago</i>	r
<i>Geothlypis aequinoctialis</i>	r
<i>Geranospiza caerulescens</i>	r
<i>Hylocharis chrysura</i>	r
<i>Lepidocolaptes angustirostris</i>	r
<i>Melanerpes candidus</i>	r

Table 4 (continued)

Species	Habitat
<i>Molothrus badius</i>	r
<i>Molothrus rufoaxillaris</i>	r
<i>Myiophobus fasciatus</i>	r
<i>Nothura maculosa</i>	r
<i>Phacellodomus sibilatrix</i>	r
<i>Phacellodomus striaticollis</i>	r
<i>Phytotoma rutila</i>	r
<i>Picoides mixtus</i>	r
<i>Polioptila dumicola</i>	r
<i>Polyborus plancus</i>	r
<i>Poospiza melanoleuca</i>	r
<i>Pseudoseisura lophotes</i>	r
<i>Pyrocephalus rubinus</i>	r
<i>Rhynchotus rufescens</i>	r
<i>Rostrhamus sociabilis</i>	r
<i>Saltator aurantiirostris</i>	r
<i>Saltatricula multicolor</i>	r
<i>Schoeniophylax phryganophila</i>	r
<i>Sicalis luteola</i>	r
<i>Sporophila minuta</i>	r
<i>Sturnella superciliaris</i>	r
<i>Suiriri suiriri</i>	r
<i>Synallaxis albescens</i>	r
<i>Tapera naevia</i>	r
<i>Taraba major</i>	r
<i>Thamnophilus doliatus</i>	r
<i>Thraupis bonariensis</i>	r
<i>Tringa solitaria</i>	r
<i>Volatinia jacarina</i>	r
<i>Xenopsaris albinucha</i>	r
<i>Xolmis irupero</i>	r
<i>Chlorostilbon aureoventris</i>	r, u
<i>Columba livia</i>	r, u
<i>Columba maculosa</i>	r, u
<i>Columba picazuro</i>	r, u
<i>Columbina picui</i>	r, u
<i>Drymornis bridgesii</i>	r, u
<i>Furnarius rufus</i>	r, u
<i>Guira guira</i>	r, u
<i>Heliomaster furcifer</i>	r, u
<i>Hirundo rustica</i>	r, u
<i>Machetornis rixosus</i>	r, u
<i>Milvago chimango</i>	r, u
<i>Mimus saturninus</i>	r, u

Table 4 (continued)

Species	Habitat
<i>Molothrus bonariensis</i>	r, u
<i>Myopsitta monachus</i>	r, u
<i>Paroaria coronata</i>	r, u
<i>Passer domesticus</i>	r, u
<i>Phacellodomus ruber</i>	r, u
<i>Phaeoprogne tapera</i>	r, u
<i>Pitangus sulphuratus</i>	r, u
<i>Saltator coerulescens</i>	r, u
<i>Sicalis flaveola</i>	r, u
<i>Sporophila caerulescens</i>	r, u
<i>Syrigma sibilatrix</i>	r, u
<i>Tachycineta leucorrhoa</i>	r, u
<i>Thraupis sayaca</i>	r, u
<i>Troglodytes aedon</i>	r, u
<i>Turdus amaurochalinus</i>	r, u
<i>Turdus rufiventris</i>	r, u
<i>Tyrannus melancholicus</i>	r, u
<i>Tyrannus savana</i>	r, u
<i>Vanellus chilensis</i>	r, u
<i>Vireo olivaceus</i>	r, u
<i>Zenaidura macroura</i>	r, u
<i>Zonotrichia capensis</i>	r, u
<i>Amazona aestiva</i>	u
<i>Progne chalybea</i>	u
<i>Riparia riparia</i>	u
<i>Serpophaga subcristata</i>	u

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