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EFFECTS OF URBAN SPRAWL ON RIPARIAN VEGETATION: IS COMPACT OR DISPERSED URBANIZATION BETTER FOR BIODIVERSITY?

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ABSTRACT

Compact urbanization is the main strategy for sustainable urban development. However, it is yet unclear whether compact urban forms are ecologically more favourable than dispersed ones. In this paper, we studied the effects of urban sprawl on the riparian vegetation condition in one of the most degraded watersheds in the Buenos Aires metropolitan area, Argentina. We conducted random sampling of the riparian vegetation at sites along streams in the basin and assessed urban indicators at the reach and sub-watershed scales for each of those sites in a geographic information system: urban area, impervious surface, population density and two landscape metrics of dispersion. The indicators assessed explained a high proportion of the variability of the vegetation response variables, thereby confirming the importance of urban sprawl pressure in shaping riparian communities in fluvial ecosystems. Dispersed urbanization had more positive than negative effects on the vegetation in the study area. Riverbanks associated with dispersed urbanization had more plant species, including exotics, when urban sprawl was assessed at the local scale. At the sub-watershed scale, dispersed urbanized areas were richer in native plants and most of the functional groups, and poorer in exotic species. The model of the compact city, including bio-corridors along watercourses, has been proposed for the Buenos Aires conurbation process for the next decades. Our results showed that the quality of existing river corridors across the compact matrix was not desirable and best practices for redesigning a more sustainable landscape structure are necessary, including the restoration of habitats for wetland species. Copyright © 2017 John Wiley & Sons, Ltd.

KEY WORDS: compact city model; riparian restoration; sustainable cities; urban planning

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INTRODUCTION

Cities worldwide are continuously expanding and currently more than half of the world population lives in urban environments (UNPD, 2012). This trend is stronger in Latin America and the Caribbean, where 80% of the people live in cities, and especially in Argentina, where 92% of the population is urban (World Bank, 2016). The growth of cities may assume different spatial patterns, which results from a joint consequence of topography, zoning law, and the geography of highways, railway lines and mass transit (McDonald, 2008). However, in many cases, it has assumed the form of the so-called 'urban sprawl' (European Environment Agency, 2006; Galster *et al.*, 2001; Jaeger *et al.*, 2010; Wilson & Chakraborty, 2013). Although this pattern has been a topic of scientific research for more than 20 years, there is considerable debate about its definition and how it can be measured (Jaeger *et al.*, 2010). Summarizing different characterizations, urban sprawl may be described as a low-density development, both residential and non-residential built-up land, at the boundary of a metropolitan area, which implies segregation of land uses (Johnson, 2001). From a landscape point of view, urban sprawl can be perceived and estimated accordingly, considering the following attributes: either the more the area is built over, or the more dispersed the built-up area, or the higher the land uptake per inhabitant, the greater the urban sprawl (Jaeger & Schwick, 2014; Jaeger *et al.*, 2010).

Urban sprawl is often criticized because of its environmental, social and economic impacts (Johnson, 2001; Wilson & Chakraborty, 2013). The main consequences of urban sprawl include greater air pollution related to the larger numbers of commuters owing to the functional and spatial separation of places for living and working (Travisi *et al.*, 2010); lower water quality resulting from the increase in impervious surface (Tu *et al.*, 2007); reduced diversity of species coupled with loss of habitats and ecosystem fragmentation (Miller, 2012); and loss of different types of land,

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such as arable soil, recreation areas and open spaces (Tan *et al.*, 2005). In reaction to this, the model of the compact city has been seen as being more compatible with the criteria of sustainable development (Ewing, 1997; Holden, 2004). Interest in compact city policies has been almost exclusively limited to the experience of developed countries (USA, Europe, Japan and Australia), with insufficient studies conducted in developing countries (Burgess, 2000; Chen *et al.*, 2008). The case of Chinese cities stands out as they are characterized by a wide range of social and environmental challenges associated with high urban densities (Chen *et al.*, 2008). It has been proved that urban compactness manages to overcome some of the negative consequences of urban sprawl (Chen *et al.*, 2008). However, higher population density may aggravate negative environmental externalities, such as noise and pollution, and it may even surpass the ability of natural ecosystems to successfully absorb pollutants without becoming degraded, which would actually come to jeopardize the sustainability of this urban form (Chen *et al.*, 2008; McDonald, 2008; Tratalos *et al.*, 2007).

Regarding urban sustainability, there is evidence that the presence of natural assets in urban contexts contributes significantly to the quality of life of dwellers through the provision of ecosystem services, such as air and water purification, wind and noise reduction, microclimate regulation, as well as social, cultural and psychological values (Gómez-Baggethun *et al.*, 2013). Although some ecosystem services may be generated at a distance from the city, others should be produced locally, within the city limits (Bolund & Hunhammer, 1999). In this context, greening cities, especially compact urban areas, is widely advocated as a key feature of a livable and sustainable city (Lovell & Taylor, 2013). In the urban fabric, streams represent interesting, and sometimes underestimated, opportunities to create 'green infrastructure' (Lovell & Taylor, 2013). Green infrastructure is a multifunctional system with components (trees, soil and constructed infrastructure), organized into a pattern (landscape), that performs functions (e.g. stormwater management, removal of air and water pollutants) and provides benefits. Moreover, green infrastructure is part of a hierarchy; it incorporates multiple subsystems (e.g. hydrology, vegetation, and movement) being a subsystem within a larger system (e.g. region, city, or neighbourhood) where it interacts with other systems, such as transportation, economy and governance (Rouse & Bunster-Ossa, 2013). In the urban matrix, green areas are as important as grey infrastructure and need to be handled and cared for with expertise (Firehock, 2015).

At the catchment scale, natural river channels and their floodplains are hot spots of biodiversity because they provide an array of habitat types (Stella *et al.*, 2013). However, riverscapes within cities are usually severely transformed, with profound changes in their ecosystem functions. Significant loss in habitat heterogeneity takes place when stream channels are engineered, replacing natural features with concrete structures. Also, they may become severely degraded when stream banks are stabilized in order to resist increased flood flows or when extensive piped storm drainage networks are constructed, completely bypassing the riparian zones and channeling large amounts of water from impervious surfaces directly into streams (Groffman *et al.*, 2003; Allan, 2004). Stream incision, combined with reduced infiltration, can lower riparian groundwater levels and have dramatic effects on ecological processes (Groffman *et al.*, 2003). These transformations usually result in impacts on riparian vegetation. Reduced water quality, due to the increased amount and variety of pollutants in the run-off (Sliva & Williams, 2001), may affect plant communities. Altered seed banks result in changes in species composition, diversity and density, including the replacement of native species for exotic ones (Moffatt & McLachlan, 2003).

The ecological impacts of urbanization have been repeatedly addressed in the literature, and, particularly, many studies have assessed biodiversity changes associated with land use gradients (Faggi & Dadon, 2010; Luther *et al.*, 2008; Moore & Palmer, 2005). However, it still remains unknown what development patterns are most effective in supporting ecological functions. In particular, it is as yet unclear whether compact urban forms are ecologically more favourable than dispersed forms (Alberti, 2005; Mohajeri *et al.*, 2015; Tratalos *et al.*, 2007). Few studies have specifically evaluated the impacts of urban sprawl on biodiversity, quantifying the degree of sprawl, and the results are not conclusive. Blair (2004) identified different effects of urban sprawl on birds at multiple levels of organization, detecting that species richness and diversity peaks at intermediate levels of urbanization. Forys and Allen (Forys & Allen, 2005) found that neither native nor rare ant species were significantly affected by urban sprawl, whereas exotic species richness was positively correlated with the amount of development. Concepción *et al.* (2016) found important impacts of urban sprawl on species richness of plants and birds, which varied considerably depending on the species groups, urban sprawl components and spatial scales considered. Regarding vegetation in particular, they detected that non-native and ruderal species benefit from urban sprawl. In this paper, we studied the effects of urban sprawl on the riparian vegetation condition in one of the most degraded watersheds in the Buenos Aires metropolitan area. We wanted to evaluate if different dimensions of urban sprawl affected the structure of riparian vegetation and the composition of native and exotic species in order to make recommendations for urban planning. In particular, we were interested in assessing whether compact or dispersed urban forms favoured riparian native vegetation condition. Knowledge

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of how habitat heterogeneity is linked to biodiversity and, consequently, to ecosystem functions and services is primordial for the management and restoration of riparian ecosystems (Reichert *et al.*, 2007), a practice that has gained increased attention in the revitalization of the urban space.

MATERIALS AND METHODS

Study area

The study area is the Matanza–Riachuelo watershed (CMR, acronym in Spanish for *Cuenca Matanza-Riachuelo*), which is located in the north-east of Buenos Aires province, Argentina. It lies between 34°37′9.31″S and 35°7′25.07″S and 58°21′2.06″W and 59°3′1.21″W (Figure 1), comprising approximately 200 000 ha. According to official estimates (INDEC, 2010), more than eight million people live in the area of influence of the watershed. The CMR is an emblematic case study because it is the most polluted watershed in Argentina and one of the most polluted throughout the world. Its main environmental problems are water, soil and air pollution (ACUMAR, 2010; Ratto *et al.*, 2004); anthropogenic alteration of the drainage system and flooding of urbanizations that occupy floodplains and low river terraces (Pereyra, 2004); open dumps that constitute a risk to human health (ACUMAR, 2010); and the loss of biodiversity associated with the transformation and destruction of habitats, as well as the invasion of exotic species (Zuleta *et al.*, 2012). As a consequence of CMR environmental problems, the authority of the watershed was created, and it is responsible for the enforcement of a comprehensive rehabilitation plan since 2009. It includes the conversion of industries, the expansion of water supply and the sewage system, monitoring of water and sediment quality, re-localization of slums, cleaning up the riverbanks and beds, afforestation of certain reaches of the Riachuelo river and environmental education, among other objectives that are still not completely fulfilled (ACUMAR, 2010; ACUMAR, 2012).

The CMR presents two main types of land use, urban and rural, which are spatially arranged as a gradient (Lafflitto *et al.*, 2011). One of the world's megacities, the city of Buenos Aires, occurs in the lower part of the watershed, representing a compact urbanization. Around it, the Buenos Aires metropolitan area developed in an unplanned and controversial manner, transforming high-quality agricultural land into urban and industrial land uses along the main roads and railways (Baxendale & Buzai, 2011). In fact, a recent study has described its spatial pattern of development as sprawled (Inostroza *et al.*, 2013). So the middle sector of the CMR is occupied predominantly by peri-urban land use, which represents a dispersed urbanization pattern. Rural land use, which includes agriculture and cattle ranching, is located mainly in the upper part of the basin and, to a lesser extent, in the middle sector.

With respect to riparian vegetation, streams in this region are characterized by rich macrophyte communities (Giorgi

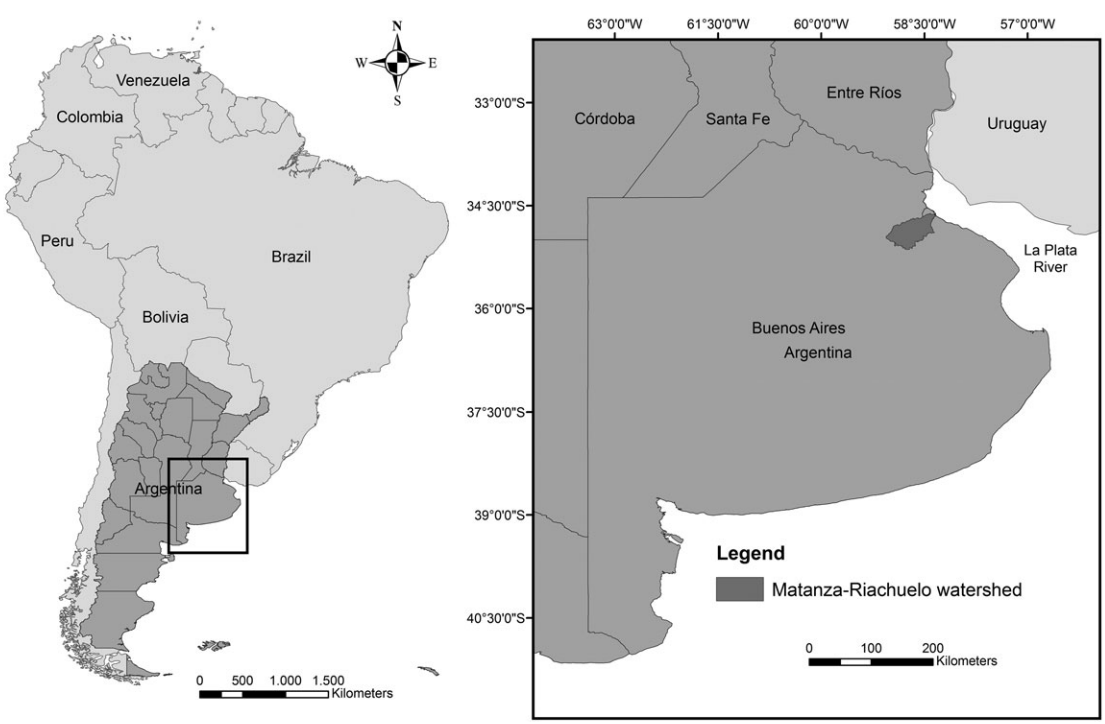

Figure 1. Location of the study area in Argentina

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et al., 2005). The dominant riparian vegetation is herbaceous, and only a few isolated trees occur, such as Humboldt's willow (*Salix humboldtiana*) or tala (*Celtis ehrenbergiana*), which develop naturally in areas with particular soil conditions (Feijoó & Lombardo, 2007). The natural riparian shrubs and herbs growing along riverbanks in the middle and upper parts of the watershed have been replaced by grasslands. Moreover, many riverbanks in these sectors have been invaded by exotic tree species during recent decades, and they are now occupied by riparian forests. The invasive woody species, which have been introduced intentionally or unintentionally from other parts of the world, include *Gleditsia triacanthos* (honey locust), *Morus alba* (white mulberry), *Populus* sp. and *Melia azedarach* (chinaberry tree) (Ghersa *et al.*, 2002). Conversely, the riparian forest characteristic of the lower part of the watershed has almost been eliminated.

Variables and data sources

In order to assess and compare the effects of compact and dispersed urbanization on the riparian vegetation condition, we conducted a field sampling at sites along streams of the CMR. Sites were located along the urban–peri-urban portion of the urban–rural gradient present in the watershed. In this study, we defined compact and dispersed urbanization as opposite ends of that continuum rather than fixed categories (Johnson, 2001), characterized by different amounts of urban surface, urban dispersion and intensity of use. Therefore, urban surface and intensity of use would increase while urban dispersion would decrease as urbanization assumed a more compact pattern of development (Jaeger & Schwick, 2014; Jaeger *et al.*, 2010).

Sites were randomly selected in a GIS among riversides that presented floodplains occupied by either urban (compact urbanization) or peri-urban (dispersed urbanization) land uses (Figure 2). To this end, we utilized a land use/land cover map generated for the CMR using a Landsat 5 TM image acquired in 2010 (Lafflitto *et al.*, 2011). In order to produce the map, homogeneous patches were visually interpreted and digitalized at a 1:50.000 scale, assigning them to several categories: urban (cities associated to high levels of soil sealing, more than 80%), peri-urban (settlements surrounded by mixed uses including open spaces and farms), urban green space (parks, squares and other public open spaces within city limits), other land uses (e.g. grazing land, farming or mining activities) and natural land covers (e.g. wetland or open water). For the purpose of this study, we simplified the map and only considered four classes: urban, peri-urban, green urban area and non-urban. The location of selected sampling sites was analysed so as to discard sites that overlapped spatially or were inaccessible. This assessment resulted in 32 sampling sites (Figure 3).

At each sampled site, a 10 m × 10 m plot was set up and all species present inside the plot were identified. Cover (%) was estimated visually for each species. The databases of the Institute of Botany Darwinion (www.darwin.edu.ar) and the National Parks Administration (www.sib.gov.ar) were checked in order to determine whether identified species were native or exotic. As compositional indicators of vegetation are linked to vegetation condition (Lawley *et al.*, 2016) and, in the case of the CMR, many riparian species invasions have been recorded, we assessed the following vegetation response variables for each sampled site: total richness, native species richness, exotic species richness, tree richness, herb richness, grass richness, aquatic plant richness, total diversity, native species diversity, exotic species diversity and native species relative cover. Richness was calculated as the number of identified species, whereas diversity was computed using the Shannon diversity Index (Begon *et al.*, 2006).

In order to estimate the degree of urban sprawl associated with each sampled site, we measured urban indicators (Concepción *et al.*, 2016; Jaeger & Schwick, 2014) at the reach and sub-watershed scales for each site in a GIS (Table I), because it has been proved that human pressures operating at different scales may have different effects on riparian communities (Bruno *et al.*, 2014; Wang *et al.*, 2001). The reach scale was defined as a 200-m buffer delimited around each site, whereas sub-watersheds were delineated for each site on the basis of the digital elevation model of the study area. Four urban indicators were estimated at both scales: urban area, mean and maximum proportion of impervious surface and population density; the last three as surrogates of intensity of use (Concepción *et al.*, 2016). We expected that all these indicators would increase as urbanization assumed a more compact pattern of development. The urban area was calculated using the land use/land cover map. In order to compute the extension of urban area, we included urban and peri-urban land, as well as green urban areas. Impervious surface was evaluated using a map of soil-sealing generated by a Tasseled Cap transformation of the same Landsat 5 TM image (Lafflitto *et al.*, 2011). The mean and maximum values of all assessed pixels were recorded in each case, because each pixel holds information about the proportion of impervious surface. Population density data were provided by INDEC (INDEC, 2010) and combined with the limits of the national census units (Dirección General de Estadística y Censos, 2014; Dirección Provincial de Estadística, 2014).

At the sub-watershed scale, another two indicators were included (Table I). Urban spatial dispersion was assessed using two landscape metrics: mean proximity index and clumpiness index (CI) (Gustafson & Parker, 1992). We expected that both indicators would show lower levels of dispersion as urbanization assumed a more compact pattern of development. Mean proximity index equals the sum of

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Figure 2. Sampled riparian sites in the study area: examples for (a) compact and (b) dispersed urbanization. [Colour figure can be viewed at wileyonlinelibrary.com]

the patch area divided by the nearest edge-to-edge distance, squared, to another patch of the same type, which equals zero when the patch has no neighbors, whereas CI ranges from −1 when the patch type is maximally disaggregated (smaller patches and more complex shapes), to 1 when the patch type is maximally clumped (larger patches with compact shapes). The landscape metrics estimation was based on the land use/land cover map using FRAGSTATS v4.2.1.603 (McGarigal *et al.*, 2013). Other environmental explanatory variables (e.g. precipitation or temperature) are invariable across the watershed, and therefore, they were not considered in the modelling.

Data analysis

Generalized linear models were used in order to assess the effect of the selected indicators of urban sprawl on the riparian vegetation condition (Bruno *et al.*, 2014; Concepción *et al.*, 2016). We did not intend to select a unique model to explain riparian vegetation in the study area. Rather, we aimed at exploring the amount of variability explained by different models and the positive or negative signs associated with coefficients for individual variables as a means to recognize effects of the urban sprawl indicators on riparian vegetation. Therefore, four types of modelling were performed for each vegetation response variable: (i) using all the explanatory variables; (ii) only variables estimated at the reach scale; (iii) only variables at the sub-watershed scale; and (iv) using each individual variable separately. Firstly, Pearson's correlation coefficients were calculated to discard any highly correlated ($r > 0.7$) explanatory variables. Linear and quadratic terms of urban sprawl indicators were included to account for possible non-linear effects. Generalized linear models were carried out

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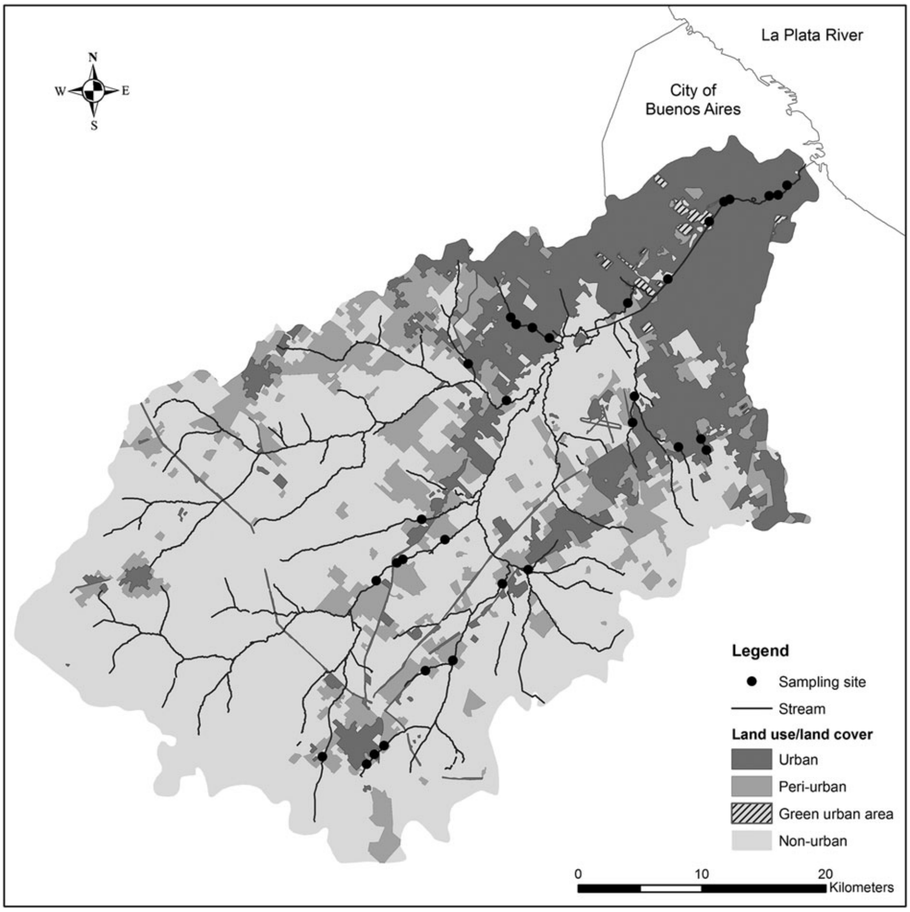

Figure 3. Location of the sampled sites along the urban-peri-urban gradient in the study area. Adapted from Lafflitto *et al.* (2011)

assuming a Poisson distribution when the dependent variable was related to vegetation richness, whereas a Normal distribution was assumed when the dependent variable was either relative cover or one of the diversity variables. The percentage of explained deviance (D^2) was used to estimate the amount of variability of the response variable explained by each model, and thus, it was used to assess its performance. A perfect model has no unexplained deviance after all variables have been included, and its D^2 takes the value 100% (Guisan & Zimmermann, 2000). Therefore, the lower the value of D^2, the lesser the amount of variability of the response variable explained by the model. Statistical analyses were performed using packages 'STATS' and 'MODEVA' of the software R (R Core Team, 2015).

RESULTS

At the sub-watershed scale, the mean percentage of impervious surface was highly correlated to several explanatory variables (urban area $r = 0.83$ and $p < 0.0001$, population density $r = 0.99$ and $p < 0.0001$ and CI $r = -0.82$ and $p < 0.0001$ at the sub-watershed scale, and the mean percentage of impervious surface at the reach scale $r = 0.83$ and $p < 0.0001$). Moreover, population density was highly correlated to urban area at the sub-watershed scale ($r = 0.82$ and $p < 0.0001$) and to the mean percentage of impervious surface at the reach scale ($r = 0.82$ and $p < 0.0001$). Finally, at the reach scale, the mean and maximum percentages of impervious surface were correlated ($r = 0.79$ and $p < 0.0001$). Therefore, in order to avoid redundancy of information, the mean percentage of impervious surface assessed at both scales, and the population density at the sub-watershed scale were discarded for subsequent modelling.

In general, complete models, including all assessed urban sprawl indicators, explained a high proportion of the variability of response variables ($D^2 > 42.50\%$; Table II), showing the importance of urban sprawl pressures in shaping riparian vegetation communities. Partial models, including variables at either the reach scale or the sub-watershed scale, also explained a high proportion of the variability ($D^2 > 15.70\%$, excluding two exceptions). In particular, models only combining indicators evaluated at the reach

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Table I. Response variables used to evaluate the riparian vegetation condition and urban sprawl indicators assessed at the reach and sub-watershed scales

	Explanatory variables	
Response variables	Reach scale	Sub-watershed scale
Richness	Urban area	Urban area
Total	Proportion of impervious surface	Proportion of impervious surface
Native species	Mean	Mean
Exotic species	Maximum	Maximum
Trees	Population density	Population density
Herbs		Mean proximity index (MPI)
Grasses		Clumpiness index (CI)
Aquatic plants		
Diversity		
Total		
Native species		
Exotic species		
Native species relative cover		

scale displayed higher D^2, except for tree and herb richness, exotic species diversity and relative cover of native species. In these latter cases, the sub-watershed scale indicators accounted for a higher percentage of explained deviance instead. This emphasizes the importance of evaluating the vegetation condition at both scales in order to fully recognize the effect of urban sprawl on riparian communities, as these influences would be operating simultaneously at the local and landscape scales.

Models that tested individual urban sprawl indicators assessed at the reach scale showed an increase in richness and diversity of riparian vegetation when urban sprawl increased (Table III). In this way, total richness and total diversity increased with decreasing urban area, and therefore, with dispersed urbanization patterns. Moreover, total richness, native species richness, grass richness, total diversity and native species diversity increased with decreasing impervious surface and therefore, with dispersed urbanization patterns. A negative aspect regarding the vegetation condition is that exotic species richness also increased with decreasing impervious surface. It is worth mentioning that the response of riparian vegetation to local impervious surface was substantial, considering the high percentage of the explained deviance ($D^2 > 13.50\%$). In any case, the effect of population density assessed at the reach scale was not significant, and the proportion of explained variability was, in general, rather low ($D^2 < 1.10\%$, excluding two exceptions).

Considering the models that tested individual urban sprawl indicators assessed at the sub-watershed scale, the effect of sprawl on riparian vegetation condition was also positive (Table III). In this way, native species richness, grass richness and aquatic plants richness increased with decreasing urban area, whereas exotic species richness and diversity decreased with decreasing impervious surface. Moreover, tree richness increased with increasing dispersion. However, herb richness decreased with decreasing impervious surface and relative cover of native species decreased with increasing dispersion. The response of the riparian vegetation to the impervious surface was again substantial, considering the high percentage of explained variability ($D^2 > 15.20\%$).

Table II. Explained deviance D^2 (%) for models tested in order to explain the effect of urban sprawl indicators on riparian vegetation

Dependent variables	Complete models	Reach models	Sub-watershed models
Richness			
Total	56.84	36.51	19.81
Native species	52.13	23.13	20.80
Exotic species	57.69	40.63	35.77
Trees	43.17	9.42	27.07
Herbs	43.46	25.91	30.02
Grasses	63.14	32.76	27.56
Aquatic plants	42.51	23.93	15.73
Diversity			
Total	62.11	46.63	23.86
Native species	54.33	32.29	25.84
Exotic species	64.09	36.88	41.63
Relative cover	53.65	9.03	34.95

DISCUSSION

Riparian corridors are resource-rich habitats within larger landscapes (Stella *et al.*, 2013) and most of them are impacted by urbanization, flow regulation or species invasions, resulting in a substantial loss of the natural heterogeneity and biodiversity (Groffman *et al.*, 2003; Moffatt & McLachlan, 2003). We found that the sprawl indicators assessed explained a high proportion of the variability in the vegetation variables evaluated, thereby confirming the importance of urban sprawl pressure in shaping riparian communities in fluvial ecosystems, especially in the case of impervious surface. Moreover, that pressure would be operating distinctively depending on the scale considered (Bruno *et al.*, 2014; Wang *et al.*, 2001). We found that each urban sprawl indicator estimated, except for population density, was influencing different vegetation response variables when assessed at the reach or sub-watershed scale. Regarding the independent variable population density, it must be noted that the official data available lacked spatial resolution and so it had to be estimated for each sampled site. Results might vary if the exact number for population density for each site was obtained.

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Table III. Explained deviance D^2 (%) for models testing the effect of individual urban sprawl indicators on riparian vegetation

Dependent variables	Reach scale			Sub-watershed scale			
	Urban area	Impervious surface	Population density	Urban area	Impervious surface	MPI	CI
Richness							
Total	6.15 (−)**	20.10 (−)***	0.81	8.12	4.12	3.32	5.62
Native	7.86	13.60 (−)***	1.07	12.90 (−)**	0.38	0.05	1.16
Exotic	7.12	14.64 (−)*	0.32	8.18	15.28 (+)*	8.16	9.62
Trees	0.71	2.31	8.30	9.66	6.42	10.71 (+)**	10.95 (−)*
Herbs	8.91	5.32	0.47	5.92	23.00 (+)***	1.05	7.98
Grasses	4.58	24.71 (−)***	0.18	18.43 (−)**	6.40	6.67	3.36
Aquatic	5.40	4.43	5.64	8.78 (−)*	4.00	1.79	1.71
Diversity							
Total	15.05 (−)**	20.11 (−)**	0.86	6.04	8.84	0.30	8.55
Native	16.17	14.72 (−)**	0.95	9.36	2.56	6.44	1.06
Exotic	13.11	8.47	0.56	6.28	21.25 (+)*	10.78	10.86
Rel. cover	6.33	0.63	0.23	8.21	1.39	22.33 (−)**	5.79

CI, clumpiness index; MPI, mean proximity index.
Direction of effects (positive or negative) are indicated (+/−) based on significant estimated coefficients.
***$p < 0.01$.
**$p < 0.05$.
*$p < 0.1$.

In general, we found that dispersed urbanization had more positive than negative effects on the vegetation at both scales in the study area (Figure 4). This result is contrary to a very recent study at the habitat and landscape scales in the Swiss Plateau (Concepción *et al.*, 2016), which could be explained by better environmental quality of the compact city in Switzerland. Instead, our results showed that riverbanks had more plant species, including exotics, in dispersed urbanization when urban indicators were assessed at the local scale. For the urban indicators assessed at the sub-watershed scale, dispersed urbanized areas were again richer than compact ones in most of the functional groups (trees, grasses and aquatic plants) and in native plants, whereas they were poorer in exotic species (richness and diversity). Such results support evidence from many cities around the world showing that urbanization homogenizes biota as communities become more alike each other owing to the introduction and extinction of species (Mc Kinney, 2005; Wittig, 1996).

Our results showed that the habitat quality for spontaneous vegetation decreases in compact areas so that many trees, grasses and aquatic plants disappeared. These transformations were so significant that even exotic plants, for example, invasive trees (*Acer negundo*, *Gledistia triacanthos*) or herbs (*Brassica rapa*, *Centaurea melitensis*, *Centaurium pullchelum*, *Echinochloa crus-galli*, *Galega officinalis*, *Phalaris canariensis*), did not find favourable conditions in riverscapes in compact urbanized areas. Some typical native plant species from wetlands, for example, *Baccharis salicifolia*, *Eleocharis bonariensis*, *Juncus tenuis*, *Luziola peruviana*, *Pluchea sagittalis* and *Mikania cordifolia*, were characteristic of dispersed conditions, but they do not thrive in compact situations, indicating that drying of the habitat occurs in compact urbanization. This coincides with the reduced infiltration that has been documented when stream channels are channelized and are replaced by concrete structures, or when stream banks are stabilized (Groffman *et al.*, 2003; Allan, 2004). It is also in agreement with the general urban ecological patterns shown by Sukopp *et al.* (1973) in the early days of studies on cities. Only a few plant species, for example, a cosmopolitan grass (*Cynodon dactylon*) and four natives (*Echinochloa helodes*, *Manihot grahamii*, *Parkinsonia aculeata* and *Verbena bonariensis*), were indifferent to compact and dispersed urbanization and can be considered as urban adapters.

Dispersed urbanization has often been criticized because of the environmental disadvantages (Johnson, 2001; Wilson & Chakraborty, 2013). This has led to the promotion of the compact city model as an alternative in many countries. Such a model is proposed by Buenos Aires council for the conurbation process in the future, until 2060 (Macri *et al.*, 2009), as they see many advantages in terms of the economic, social, mobility and environment dimensions. Although Buenos Aires has over 250 public green spaces, the amount of square meters of green space per inhabitant is still not satisfactory (6.2 m^2/inhab.), as it does not reach the desired value of 10–15 m^2/inhab. As a measure to mitigate this, the master plan for the city growth has proposed the implementation of bio-corridors along watercourses (Macri *et al.*, 2009) to interconnect streams, urban reserves and the Río de la Plata waterfront. This would be in accordance with the importance that has been placed on green

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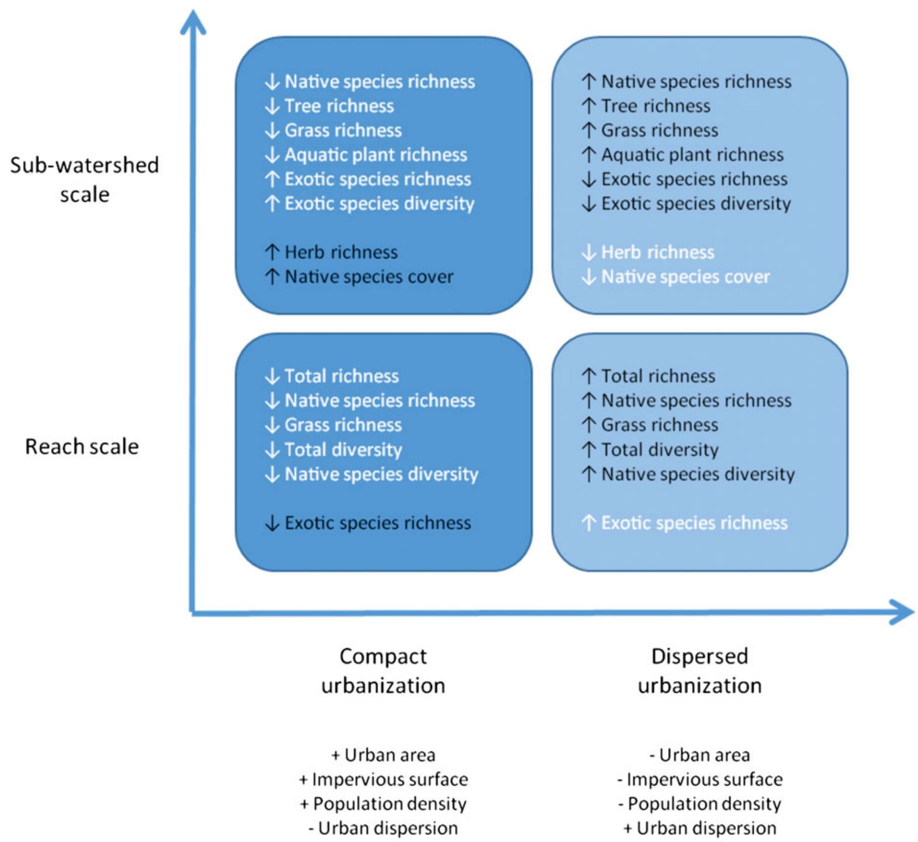

Figure 4. Effects of compact and dispersed urbanization on riparian vegetation when urban sprawl indicators are assessed at the reach or sub-watershed scales. Arrows indicate the direction of change (increase or decrease), positive impacts to riparian biodiversity are in black font and negative impacts in white font. [Colour figure can be viewed at wileyonlinelibrary.com]

infrastructure regarding the provision of ecosystem services and the attainment of urban sustainability (Gómez-Baggethun *et al.*, 2013; Lovell & Taylor, 2013). However, as stated by Hellmund and Smith (Hellmund & Smith, 2006), the design of greenways, which could contribute to wildlife conservation and recreation, is not a simple task and environmental attributes in corridors may need to be managed. In fact, our results showed that the quality of existing river corridors across the compact matrix was not desirable and best practices for redesigning a more sustainable landscape structure are necessary. In order to accomplish this, an ecological approach to landscape design, considering the requirements of plants, should be undertaken. Our findings showed the urgent necessity to decrease impervious surfaces, at least at the reach scale, restoring wet habitats for aquatic plants, sedges and rushes. Some of the native species that were indifferent to compact and dispersed urbanization, and thus, act as urban adapters are recommended to revegetate riparian sites in the early steps of the restoration plans, showing that studies like this are applicable in ecological restoration and in green infrastructure management.

CONCLUSIONS

The dispersed urban form resulted as more favourable to the riparian vegetation promoting native species richness and diversity and could be most effective for supporting ecological functions. Moreover, compact urbanization was associated with lower habitat quality for spontaneous vegetation and drying of habitats. However, considering other dimensions of sustainability, such as infrastructure efficiency and energy or land consumption, urban sprawl is also the most unfavourable urbanization pattern. Development leads inexorably to the increase of urban density, which should be organized around the compact city. What our results showed regarding the dispersed pattern should be applied to the compact city design, in order to implement and manage bio-corridors within the city limits; there is an urgent necessity to decrease impervious surfaces and to restore wet habitats including species identified as urban adapters in this study. These considerations should be taken into account, especially for the planning processes of sustainable growing cities in developing countries.

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