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Bird Diversity Across an Andean City: The Limitation of Species Richness Values and Watershed Scales

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ARTÍCULO DE INVESTIGACIÓN / RESEARCH ARTICLE

BIRD DIVERSITY ACROSS AN ANDEAN CITY: THE LIMITATION OF SPECIES RICHNESS VALUES AND WATERSHED SCALES

Diversidad de aves en una ciudad andina: la limitación de los valores de riqueza de especies
y la escala de cuenca

Running title: Bird diversity across an Andean city

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ABSTRACT

Evaluating several biodiversity descriptors and considering several spatial scales might elucidate conservation issues and improve biodiversity monitoring in urban environments. We estimated species richness (order $q = 0$), Shannon diversity (order $q = 1$), and Simpson diversity (order $q = 2$) based on Hill numbers and performed cluster analysis and non-metric multidimensional scaling (NMDS) to compare seven urban micro-watersheds and a periurban site across a northern Andean city (Medellín, Colombia). We found 113 diurnal resident bird species: 50 (44 %) exclusively within urban sites, 21 (19 %) exclusively in the periurban site, and 42 (37 %) shared species. Some urban watersheds had similar bird species richness to the periurban site, but Shannon and Simpson diversities were always lower, showing decrease in local bird diversity when abundances were considered. Bird species composition differed between urban watersheds and the periurban site, with all urban watersheds grouped altogether by cluster and NMDS analysis, and the periurban site forming its own group. This suggests homogenization of bird species composition due to the species turnover decreasing across urban areas, with endemic, near endemic and rare species restricted to periurban areas where native forest remnants persist. Several scales of biodiversity and analysis at more local scales are needed to better understand biodiversity patterns across Andean cities and to design urban planning strategies that prevent biodiversity loss.

Keywords: Aburrá Valley, Neotropical city, Tropical Andes, urban biodiversity, urbanization.

RESUMEN

La evaluación de varios indicadores de biodiversidad y considerar varias escalas espaciales podría evidenciar problemas de conservación y mejorar el monitoreo de biodiversidad en ambientes urbanos. Estimamos la riqueza de especies (orden $q = 0$) y las diversidades de Shannon (orden $q = 1$) y Simpson (orden $q = 2$) usando números de Hill, y realizamos análisis de agrupamiento y escalamiento no métrico multidimensional (NMDS) para comparar siete microcuencas urbanas y un sitio periurbano en una ciudad del norte de los Andes (Medellín, Colombia). Encontramos 113 especies de aves residentes: 50 (44 %) exclusivamente dentro de la zona urbana, 21 (19 %) exclusivamente en el sitio periurbano y 42 (37 %) compartidas. Algunas cuencas urbanas tuvieron una riqueza de especies similar al sitio periurbano, pero las diversidades de Shannon y Simpson fueron siempre menores, mostrando patrones más claros de reducción de diversidad de aves al considerar las abundancias. La composición de especies de aves se diferenció entre zonas urbanas y el sitio periurbano, con las microcuencas urbanas agrupándose entre sí en los análisis de agrupamiento y NMDS. El sitio periurbano formó su propio grupo, sugiriendo homogenización en la composición de especies de aves debido a la reducción de recambio de especies dentro de la zona urbana, con las especies endémicas, casi endémicas y raras restringiéndose a zonas periurbanas donde persisten remanentes de bosque nativo. Indicadores que evalúen varias escalas de biodiversidad y análisis que consideren escalas espaciales más locales son necesarios para entender mejor los patrones de biodiversidad en ciudades andinas y diseñar estrategias de planificación urbana que prevengan la pérdida de biodiversidad.

Palabras clave: Andes Tropicales, biodiversidad urbana, ciudad Neotropical, urbanización, Valle de Aburrá.

INTRODUCTION

Biodiversity, a commonly used term in urban environmental policies and biological conservation studies (Aronson et al., 2017), represents a multidimensional concept where biological scales are interdependent on each other (Bennie et al., 2011), following the hierarchical nature of biological entities, from genes and species to ecosystems. Furthermore, the term biodiversity is usually simplified to “species loss” (Sol et al., 2014), which under public media and political popularization results in the misunderstanding of the original concept, obnubilating its functional relevance and its direct application on sustainability strategies in highly human-disturbed environments (Puppim de Oliveira et al., 2011).

Conceptual complexity and methodological bias (i.e., lacking of systematic sampling) complicate the measurability and monitoring of urban biodiversity (Ouyang et al., 2018). Therefore, simple descriptors such as species richness are the most used in urban ecosystems (Mckinney, 2008; Nielsen et al., 2013), as they are easier to understand by policymakers. However, focusing on species numbers is not always operational, because biodiversity is inherently dependent on multidimensional ecosystem processes, such as mass and energy flow and several biological interactions that involve individuals, populations or species, and communities, simultaneously (Bennie et al., 2011). Hence, a lack of other spatial and biological dimensions might mask conservation issues driven by urbanization.

Biodiversity monitoring could be an overwhelming task, but it can be optimized by focusing on certain groups that have high taxonomic resolution, predictable ecological responses, and high detectability under the available sampling methods (Anderson, 2018). Birds are a highly conspicuous and well-known bioindicator group in a wide range of environmental conditions (Becker, 2003), and they are useful for monitoring impacts on natural or human-dominated

ecosystems. At the community multispecies level, it is possible to gather huge amounts of bird data with observational methods that are simple to implement and replicate (Sutherland et al., 2004). Unsurprisingly, birds have been one of the most studied groups in urban ecosystems (McKinney, 2008; Nielsen et al., 2013), although some regions of Latin America have knowledge gaps on systematic ecological studies that could be a limitation for biological conservation in cities (Ortega-Álvarez and Macgregor-Fors, 2011).

Fortunately, bird diversity urban studies in Latin America are increasing in numbers (Leveau et al., 2022), and information on bird species abundance and distribution is increasingly gathered by citizen science platforms (e.g., Cornell Lab of Ornithology, 2023). This represents a way forward in understanding bird diversity patterns across cities in the region and a valuable source of information for improving the conservation strategies, social appropriation of knowledge, and environmental politics design (Mazaris, 2017). Nevertheless, the uses and implications of biodiversity concepts, the spatiotemporal scale effects on bird diversity patterns, and methodological issues regarding how bird diversity is measured and monitored will be the keystone for making the most of this new available information.

In some biodiversity hotspots such as the Tropical Andes, urban sprawl is occurring at higher rates than the global average (Cincotta et al., 2000), without detailed background information and analysis to account for the possible consequences for biodiversity (Ortega-Álvarez and Macgregor-Fors, 2011). In addition, the high endemism in northern Andes implies a high risk of homogenization in bird species composition driven by human-induced landscape transformation (McKinney, 2006), as natural habitats across cities and surroundings decrease in extension and increase in fragmentation (Quintero et al., 2017). This highlights the need

of including diversity descriptors beyond species richness and improving methodologies in data taken and analysis to monitoring urban biodiversity.

In this study, we evaluated whether bird species richness is a reliable descriptor to detect and monitor biodiversity decline in a northern Andean city, using the watershed-scale urban planning used in Andean Colombian capitals (Andrade et al., 2013; Universidad Nacional de Colombia, 2005). Because bird abundance might increase at higher rates than bird species richness with urbanization sprawl (Sax and Gaines, 2003; Ortega-Álvarez and MacGregor-Fors, 2009), we hypothesized that alpha diversity descriptors that consider the number of species in proportion to their abundances would be better predictors than species richness. Additionally, we hypothesized that urbanization homogenizes bird assemblages across a city of northern Andes, which could represent a conservation concern in this area with high endemism.

MATERIALS AND METHODS

STUDY AREA

The Aburrá Valley of the Colombian Central Andes (06°14'57"N, 75°34'42"W) is located between 1000 and 3000 m.a.s.l.; almost four million people live in urban areas (Departamento Nacional de Estadística [DANE], 2019). The basin has been completely transformed by urbanization and silvicultural management and natural vegetation only remains along the valley's slopes above 1800 m.a.s.l. Mean temperature and precipitation in the study area was 18.29 ± 0.25 °C (prom \pm SD) and 1598.52 ± 289.50 mm/year between 2014 and 2019, respectively (IDEAM, 2023) (the temporal range of our study).

We subdivided the metropolitan area of Aburrá Valley according to micro-watershed delimitation by the Management and Ordering Plan of the Aburrá Valley Watershed

(Universidad Nacional de Colombia [UNAL], 2005), excluding the periurban section of each micro-watershed. We used seven micro-watersheds (with availability of systematic bird count data) to evaluate bird biodiversity patterns and species composition within the city, including a periurban area with native forest remnants in southeastern Medellín as a reference site for comparison.

LAND COVER CLASSIFICATION

We used the Build Virtual Raster tool in QGIS (version 2.18.25) and the SCP tool for atmospheric correction (QGIS Development Team, 2023), based on a semi-automatic land coverage classification, that was summarized into three categories. We defined grass, trees, and built-up areas, based on bands on brightness, greenness, and wetness, using images from Sentinel-2 satellite (10 m of accuracy), taken on 20 December 2017. This classification was performed for the seven urban micro-watersheds to compare bird diversity patterns within the city.

BIRD SURVEYS

We established 77 bird point counts (11 per urban site) within the city and seven more at the periurban site; all of them at least 200 m apart from each other along the altitudinal range 1486-2351 m.a.s.l. (Fig. 1). These points were part of biodiversity studies for independent projects that needed bird inventories for environmental permissions (including the periurban site). The same sampling protocol was used under equivalent sampling design, establishing points across contrasting habitats (e.g., grass, trees, built-up areas with scattered vegetation, etc.).

We recorded every bird seen or heard for four ten-minute visits at 25 m-fixed radius point counts, between 06:00 and 10:00 hours, reaching a total effort of 3640 minutes. We made visits to the same point in different days, but within the same week. Nocturnal, overflying, and Nearctic-Neotropical migratory bird species were excluded. We sampled between September 2014 and June 2019, unifying data under a "space-for-time" substitution approach (Pickett, 1989), where comparisons focused on spatial rather than temporal variation.

The study area changed less than 4 % in land cover percentage by category (grass, trees and built-up areas) during this period (personal unpublished data); also, mean temperature and precipitation changed less than 3 % and 10 %, respectively, during the same period (personal analysis based on data by IDEAM, 2023). Thus, although some temporal bias might exist, we assumed that spatial variation would be more influential on the bird diversity patterns we aimed to evaluate, because we used a wide spatial scale to analyze the system (i.e. micro-watersheds).

DATA ANALYSIS

We considered each point an independent sample (i.e. replicate) and the maximum of individuals per species per point the independent record (to avoid bird recounting), with 50 individuals being the maximum value for mono-specific bird flocks (i.e. we fixed "50" as the abundance when any species obtained more than 50 records at the same point). To compare sites with different sampling effort, we plotted sample-size-based rarefaction/extrapolation curves of order $q = 0$ (Species richness), $q = 1$ (Shannon diversity: number of common species) and $q = 2$ (Simpson diversity: number of very common species), based on bird data

abundance, with 95 % confidence intervals, using the package iNEXT in the program R version 2.5-5 (Hsieh et al., 2016).

We calculated the asymmetrical Hellinger distance between sampling sites using the *decostand* and *vegdist* functions of the Community Ecology Package *vegan* in R (Oksanen et al., 2019). Hellinger distance avoids double zeros issues, and therefore, it is a highly recommended distance for analysis based on abundance data (Legendre and Legendre, 2012). We plotted hierarchical clusters with the *hclust* function of the Stats R Package (R Core Team, 2019), using complete linkage, single linkage, UPGMA, WPGMA, and Ward methods. We selected the most supported cluster (i.e. the highest cophenetic correlation with better goodness of fit, consistent silhouette widths without ambiguous clustering, and the number of supported groups by Fusion levels, see Legendre and Legendre, 2012), and draw a heat map using the *pheatmap* function of the Pheatmap R Package (Kolde, 2019). Additionally, we performed a non-metric multidimensional scaling (NMDS) using the function *metaMDS* of the Community Ecology Package *vegan* (Oksanen et al., 2019); this analysis complemented the cluster analysis by showing which bird species might influence the affinities between sites (similarities/dissimilarities).

Finally, only for urban micro-watersheds, we ran the *glmulti* function of the *glmulti* R Package (Calcagno and de Mazancourt, 2010), based on generalized linear models (GLM) (version 1.0.7.1.), fitting the variables under the formulas “Richness ~.”, “Shannon ~.”, and “Simpson ~.”. Percentage of trees, grass/ or built-up areas, mean and maximum patch size for each land cover type were explanatory variables (all were included simultaneously in each modeling process); Richness, Shannon, and Simpson were the response variables (based

on Hill numbers of order $q = 0$, $q = 1$, and $q = 2$, respectively). We used the Poisson distribution and restricted the model selection to main effects (without variable interactions), due to high autocorrelation between some explanatory variables. We calculated Spearman correlation coefficients between explanatory variables with the function *chart.Correlation* of the package Performance Analytics in R (Peterson et al., 2019), and verified multicollinearity in regression analysis with the variance inflation factor (VIF) with the function *vif* of the package car in R (Fox & Weisberg, 2019). We selected best models based on Akaike's Information Criterion corrected for small sample size ($AICc \leq 2.0$) and estimated individual p-values to evaluate effects of each explanatory variable on bird diversity ($q = 0$, $q = 1$, and $q = 2$), excluding models with high correlated variables ($VIF \geq 5.0$).

RESULTS

We found 113 diurnal resident bird species (Supplementary Table 1): 50 (44 %) exclusively within urban sites, 21 (19 %) exclusively in the periurban site, and 42 (37 %) shared species. About 70 % of bird records corresponded to 20 species, with greater abundances in the urban micro-watersheds. The most common species along urban micro-watersheds were the exotic *Columba livia* (5.15 % of urban records vs 0 % periurban records) and some Neotropical birds with wide distributional ranges: *Pygochelidon cyanoleuca* (8.24 % urban vs 2.32 % periurban), *Thraupis episcopus* (6.84 % urban vs 1.39 % periurban), *Zenaida auriculata* (6.81 % urban vs 1.85 % periurban), *Columbina talpacoti* (5.34 % urban vs 0 % periurban), among others (Supplementary Table 1).

ALPHA DIVERSITY

Observed and estimated values of alpha diversity of order $q = 0$ (species richness) were usually higher in the periurban site (observed species richness= 63; estimated species

richness = 71.99 ± 5.75), but two urban micro-watersheds showed similar observed species richness and higher estimated species richness in rarefaction curves (Fig. 2) (observed species richness = 62, for both urban sites: MSE and WCI; estimated species richness = 81.57 ± 14.34 and 72.06 ± 7.98 , respectively). Values of alpha diversity of order $q = 1$ (Shannon diversity: number of common species) and order $q = 2$ (Simpson diversity: number of very common species) were always higher in the periurban site (Fig. 2). Hence, estimated values of local diversity showed that the periurban site had more even bird assemblages regarding species numbers in proportion to their abundances, although some urban neighborhoods had similar bird species richness.

BETA DIVERSITY

The Hellinger distance matrix showed that urban sites shared more species composition and abundance patterns than any of them with the periurban site (Fig. 3). The heat map suggested that all urban sites were differentiated from the periurban area, and most of them were indistinguishable from one another, especially those located on highly developed areas (67–100 % of built-up areas), such as all urban sites in the western micro-watersheds of the study area (Fig. 3). Hellinger distances were above 0.95 between any urban site and the periurban site, whereas all urban sites had 0.59 ± 0.10 of distance between them (max = 0.79) ($\sqrt{2}$ is the upper limit of Hellinger distances; approximately 1.41).

NMDS analysis also showed high dissimilarity between the seven urban micro-watersheds and the periurban site, with bird assemblages evidencing a gap across the ordination process regarding their species composition (Fig. 4). This gap differentiated the periurban assemblage based on species that were absent within the city, such as *Hypopyrrhus*

pyrohypogaster (endemic), *Grallaria ruficapilla*, *Vireo leucophrys*, *Myadestes ralloides*, *Tangara gyrola*, *Henicorhina leucophrys*, and *Basileuterus tristriatus*, and species with higher relative abundance out of the city, such as *Ortalis columbiana* (endemic), *Leptotila verreauxi*, *Momotus aequatorialis*, *Colaptes rubiginosus*, *Zimmerius chrysops*, *Arremon brunneinucha* and *Stilpnia heinei*.

LOCAL DIVERSITY ACCORDING TO LAND COVER COMPOSITION AND STRUCTURE

Land cover percentage and mean and maximum patch size of built-up areas, trees, and grass were weak variables for predicting local bird diversity patterns at the watershed-scale across Aburrá Valley's urban areas. In the case of Shannon and Simpson diversities ($q = 1$ and $q = 2$, respectively), the best models ($AICc \leq 2.0$) included the models without any explanatory variable (i.e. Shannon ~ 1 and Simpson ~ 1) (Table 1). Otherwise, although the best model for species richness (we only found one with $AICc \leq 2.0$) included two explanatory variables with significant influence ($p < 0.05$), the estimates for each variable were relatively low (Table 1), with model residuals and null deviance suggesting low predictability. This model suggested that a decrease on built-up areas and an increase on the maximum patch size of trees increased bird species richness (Table 1).

DISCUSSION

The multidimensionality and hierarchical nature of biodiversity constrain the use of a single measurement such as species richness to describe it adequately (Smith et al., 2021). Nevertheless, keeping the lesser and simplest complementary measures to describe it seem a

convenient issue for urban planning and biological conservation in Latin American cities, regarding the difficulties of implementing the scientific available knowledge into public policies (MacGregor-Fors et al., 2020). As we hypothesized, alpha diversity descriptors that consider the number of bird species in proportion to their abundances are better predictors than species richness in the Aburrá Valley; also, we found signals of bird assemblages' homogenization drove by urbanization, as we hypothesized, something that would be undetected without any descriptor of species composition. Indeed, when species richness is the only measurement taking into account, we might underestimate local biodiversity loss, both at alpha and beta-diversity scales, which implies a potential misleading of public policies and conservation efforts to achieve more sustainable cities, an compelling issue in biodiversity hotspots such as Tropical Andes (Cincotta et al., 2000).

We found similar bird species richness in the periurban site and the two urban micro-watersheds with the highest proportion of green cover (grass and trees). Nevertheless, alpha diversity descriptors considering the number of species in proportion to their abundances (i.e. Shannon and Simpson diversities based on Hill numbers) were higher in the periurban site compared to all urban sites. The differences found between these descriptors is challenging because urban planning uses mainly species richness to prioritize conservation efforts and biodiversity monitoring (Nielsen et al., 2013; Aronson et al., 2017).

Other alpha-diversity measurements considering relative abundances that are still easy to interpret by policymakers could be useful to detect biodiversity declines, as abundance might reflect local biodiversity patterns that are masked in species richness (Clergeau et al., 2006). Hence, an approach with complementary alpha-diversity descriptors might detect increasing

abundance of ecologically generalist birds whose presence in cities could be a nuisance for more specialized birds (Sax and Gaines, 2003). In addition, in northern Andes, where species turn over and endemism are high (Rahbek et al., 2019), forest-obligated species might be highly vulnerable to urban sprawl and the loss and fragmentation of native forests (Becker et al., 2008); this could be perceived in urban limits by using Shannon and Simpson diversity values rather than species richness, along with beta-diversity analysis that show differences in species composition. Indeed, the two endemic bird species found in our study were absent (*H. pyrohypogaster*) or less abundant (*O. columbiana*) in urban sites, and similar patterns were showed by other species with restricted distributions (e.g. near-endemics and species restricted to Andean ranges) that were underrepresented in urban sites.

The inclusion of beta-diversity analysis acquires relevance in high biodiverse Andean ranges (Rahbek et al., 2019), especially when biotic homogenization is driven by urbanization sprawl along Neotropical cities (Leveau et al., 2017). Therefore, the prevalence of generalist and widely distributed birds in urban areas could be detected by comparing bird assemblages. In our study, the Aburrá Valley showed highly resembling bird assemblages to each other across urban neighborhoods but all of them differentiated from the periurban site. Hence, including this information in biodiversity monitoring would be essential to develop policies that protect the less abundant and most vulnerable species, beyond enhancing alpha biodiversity in urban areas, recognizing urbanization as “the most homogenizing of all major human activities” (McKinney, 2006).

Additionally, although built cover and maximum patch size of trees might predict some changes on bird species richness in the Aburrá Valley (i.e. less built-up areas percentage and

bigger tree areas increase species richness), Shannon and Simpson diversities, the descriptors showing the bird diversity patterns more accurately in the study area, are weakly predicted by any landscape variable that we used to evaluate alpha diversity. These weak responses do not mean that landscape composition and structure are irrelevant to understand urban bird diversity, but that the spatial scale might also play an important role in understanding these patterns.

The micro-watershed scale seemed to be unable to detect local differences regarding urban green spaces composition and structure in urban areas of the Aburrá Valley. This limitation could be also a matter of sample size but, in any case (e.g. by sampling design or spatial scale constraints), it seems that more local scales are needed to understand biodiversity patterns across Andean cities, particularly when most urban species have intermediate tolerance to urbanization and might respond to local rather than landscape scales (Chace and Walsh, 2006). In addition, under a scenario where size, shape, and other landscape attributes of green spaces are difficult to manage, identifying local attributes such as vegetation composition and structure could improve green space management to enhance local bird diversity (Shwartz et al., 2008; Threlfall et al., 2017; Garizábal-Carmona and Mancera-Rodríguez, 2021).

When only landscape scales are evaluated and bird species richness is the main alpha-diversity descriptor, other studies have also failed to find differences in bird assemblages across urban ecosystems (Ferenc et al., 2014; Morelli et al., 2016). It has been suggested that urban species richness is a matter of scale, as more local approaches have shown significant differences in species richness (Chong et al., 2014; Shwartz et al., 2013). Furthermore, it

could be helpful that urban planning includes both landscape and local scales simultaneously, as the analysis of only one scale might difficult the understanding of urbanization effects on different components of biodiversity.

Finally, more attention by conservation biologists is needed on Andean urban ecosystems and its surroundings, where urban planning and local development is mostly lead by engineers, architects, politicians, and other people with little knowledge on biodiversity. Interdisciplinary perspectives on urban development and management, and a more precise view on the territory, would help to understand how urbanization affects biodiversity and to improve urban planning strategies that alleviate growing conflicts between biodiversity and urban development across northern Andes, one of the most biodiverse regions of the world.

CONCLUSIONS

Combining spatial scales along with beta-diversity analysis and alpha-diversity descriptors that not only consider species richness could help to detect urbanization effects, especially when losing local identity by local extinction of small range fauna is one of the main conservation issues in human-transformed landscapes (McKinney, 2006). Although species richness could be an indicator of better urban green spaces management (Threlfall et al., 2017), declining of more specialized bird populations and homogenization of species composition are still a conservation issue that needs to be considered, because some bird populations are isolated and fragmented without being noticed, representing a conservation issue for small range forest-dependent birds, which are not found within the most developed areas of Andean cities.

AUTHOR'S PARTICIPATION

JAG: Conceptualization, Sampling design, Data acquisition- Curation- Analysis, Data interpretation, Writing- Original draft preparation, Visualization, Investigation. **JSB:** Data acquisition, Investigation, Writing- Reviewing and Editing. **SMA:** Data acquisition, Writing- Reviewing and Editing. **LFE:** Data acquisition, Writing- Reviewing and Editing. **NRG:** GIS support and analysis. **NJM:** Supervision, Writing- Reviewing and Editing.

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DISCLOSURE OF INTEREST

We declare that there is no conflict of interests regarding this research.

REFERENCES

- Anderson, C. B. (2018). Biodiversity monitoring, earth observations and the ecology of scale. *Ecology Letters*, 21(10), 1572–1585. <https://doi.org/10.1111/ele.13106>
- Andrade, G. I., Remolina, F., & Wiesner, D. (2013). Assembling the pieces: a framework for the integration of multi-functional ecological main structure in the emerging urban region of Bogotá, Colombia. *Urban Ecosystems*, 16, 723–739. <https://doi.org/10.1007/s11252-013-0292-5>
- Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J. L., Macgregor-fors, I., McDonnell, M. J.,

- Mörtberg, U., Pyšek, P., ... Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society*, 281, 1–8. <https://doi.org/10.1098/rspb.2013.3330>
- Aronson, M. F. J., Lepczyk, C. A., Evans, K. L., Goddard, M. A., Lerman, S. B., MacIvor, J. S., Nilon, C. H., & Vargo, T. (2017). Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment*, 15(4), 189–196. <https://doi.org/10.1002/fee.1480>
- Becker, C. D., Loughin, T. M., & Santander, T. (2008). Identifying forest-obligate birds in tropical moist cloud forest of Andean Ecuador. *Journal of Field Ornithology*, 79(3), 229–244. <https://doi.org/10.1111/j.1557-9263.2008.00184.x>
- Becker, P. H. (2003). Biomonitoring with birds. In M. K. Rooks, S. Aramco, K. Snyder, L. Wilson, J. Fleming, & M. Wisnewski (Eds.), *Bioindicators and biomonitors* (1st ed., pp. 677–736). Elsevier Science Ltd. [https://doi.org/10.1016/S0927-5215\(03\)80149-2](https://doi.org/10.1016/S0927-5215(03)80149-2)
- Bennie, J., Anderson, K., & Wetherelt, A. (2011). Measuring biodiversity across spatial scales in a raised bog using a novel paired-sample diversity index. *Journal of Ecology*, 99(2), 482–490. <https://doi.org/10.1111/j.1365-2745.2010.01762.x>
- Calcagno, V., & de Mazancourt, C. (2010). glmulti: An R package for easy automated model selection with (generalized) linear models. *Journal of Statistical Software*, 34(12), 1–29. <https://doi.org/10.18637/jss.v034.i12>
- Chace, J. F., & Walsh, J. J. (2006). Urban effects on native avifauna: A review. *Landscape and Urban Planning*, 74, 46–69. <https://doi.org/10.1016/j.landurbplan.2004.08.007>

Chong, K. Y., Teo, S., Kurukulasuriya, B., Chung, Y. F., Rajathurai, S., Tiang, H., & Tan, W. (2014). Not all green is as good: Different effects of the natural and cultivated components of urban vegetation on bird and butterfly diversity. *Biological Conservation*, 171, 299–309. <https://doi.org/10.1016/j.biocon.2014.01.037>

Cincotta, R. P., Wisnewski, J., & Engelman, R. (2000). Human population in the biodiversity hotspots. *Nature*, 404, 990–992. <https://doi.org/10.1038/35010105>

Clergeau, P., Croci, S., Jokimäki, J., Kaisanlahti-Jokimäki, M. L., & Dinetti, M. (2006). Avifauna homogenisation by urbanisation: Analysis at different European latitudes. *Biological Conservation*, 127(3), 336–344. <https://doi.org/10.1016/j.biocon.2005.06.035>

Cornell Lab of Ornithology. (2023). eBird Basic Dataset. Ithaca, New York.

de Castro Pena, J. C., Martello, F., Ribeiro, M. C., Armitage, R. A., Young, R. J., & Rodrigues, M. (2017). Street trees reduce the negative effects of urbanization on birds. *PLoS ONE*, 12(3), 1–19. <https://doi.org/10.1371/journal.pone.0174484>

Departamento Nacional de Estadística (DANE). (2019). Resultados Censo Nacional de Población y Vivienda 2018 - Valle de Aburrá. <https://www.dane.gov.co/files/censo2018/informacion-tecnica/cnpv-2018-presentacion-3ra-entrega.pdf>

Ferenc, M., Sedláček, O., Fuchs, R., Dinetti, M., Fraissinet, M., & Storch, D. (2014). Are cities different? Patterns of species richness and beta diversity of urban bird communities and regional species assemblages in Europe. *Global Ecology and*

- Biogeography, 23(4), 479–489. <https://doi.org/10.1111/geb.12130>
- Fox, J., & Weisberg, S. (2019). *An R Companion to Applied Regression* (Third edit). Sage Publications. <https://socialsciences.mcmaster.ca/jfox/Books/Companion/>
- Garizábal-Carmona, J. A., & Mancera-Rodríguez, N. J. (2021). Bird species richness across a Northern Andean city: Effects of size, shape, land cover, and vegetation of urban green spaces. *Urban Forestry & Urban Greening*, 64(September), 127243. <https://doi.org/10.1016/j.ufug.2021.127243>
- Hsieh, T. C., Ma, K. H., & Chao, A. (2016). iNEXT: an R package for rarefaction and extrapolation of species diversity (Hill numbers). *Methods in Ecology and Evolution*, 7(12), 1451–1456. <https://doi.org/10.1111/2041-210X.12613>
- IDEAM. (2023). Cambio Climático. Ministerio de Medio Ambiente. <http://www.ideam.gov.co/web/atencion-y-participacion-ciudadana/cambio-climatico>
- Kolde, R. (2019). Package “pheatmap” (1.0.12). CRAN - Rproject. <https://cran.r-project.org/web/packages/pheatmap/index.html>
- Legendre, P., & Legendre, L. (2012). *Numerical Ecology* (Third Ed). Elsevier B. V.
- Leveau, L. M., Leveau, C. M., Villegas, M., Cursach, J. A., & Suazo, C. G. (2017). Bird Communities Along Urbanization Gradients: a Comparative Analysis Among Three Neotropical Cities. *Ornitología Neotropical*, 28(June), 77–87.
- Leveau, L. M., Villaseñor, N. R., & Lambertucci, S. A. (2022). Ornitología urbana en el Neotrópico: estados de situación y desafíos. *El Hornero*, 37(2), 13–22.
- MacGregor-Fors, I., Gómez-Martínez, M. A., García-Arroyo, M., & Chávez-Zichinelli, C.

- A. (2020). A dead letter? Urban conservation, management, and planning strategies from the Mexican urban bird literature. *Urban Ecosystems*, 23(5), 1107–1115.
<https://doi.org/10.1007/s11252-020-00970-y>
- Mazaris, A. D. (2017). Open data and the future of conservation biology. *Ethics in Science and Environmental Politics*, 17, 29–35.
- Mckinney, M. L. (2008). Effects of urbanization on species richness: A review of plants and animals. *Urban Ecosystems*, 11, 161–176. <https://doi.org/10.1007/s11252-007-0045-4>
- McKinney, M. L. (2006). Urbanization as a major cause of biotic homogenization. *Biological Conservation*, 127, 247–260. <https://doi.org/10.1016/j.biocon.2005.09.005>
- Morelli, F., Benedetti, Y., Ibáñez-Álamo, J. D., Jokimäki, J., Mänd, R., Tryjanowski, P., & Møller, A. P. (2016). Evidence of evolutionary homogenization of bird communities in urban environments across Europe. *Global Ecology and Biogeography*, 25, 1284–1293. <https://doi.org/10.1111/geb.12486>
- Nielsen, A. B., van den Bosch, M., Maruthaveeran, S., & van den Bosch, C. K. (2013). Species richness in urban parks and its drivers: A review of empirical evidence. *Urban Ecosystems*, 17, 305–327. <https://doi.org/10.1007/s11252-013-0316-1>
- Oksanen, J., Guillaume, F., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P. R., O'Hara, R. B., Simpson, G. L., Solymos, P., Stevens, M. H., Szoecs, E., & Wagner, H. (2019). *Vegan: Community Ecology Package (2.5-5)*. CRAN - Rproject. <https://cran.r-project.org/web/packages/vegan/vegan.pdf>

- Ortega-Álvarez, R., & Macgregor-Fors, I. (2011). Dusting-off the file: A review of knowledge on urban ornithology in Latin America. *Landscape and Urban Planning*, 101, 1–10. <https://doi.org/10.1016/j.landurbplan.2010.12.020>
- Ortega-Álvarez, R., & MacGregor-Fors, I. (2009). Living in the big city: Effects of urban land-use on bird community structure, diversity, and composition. *Landscape and Urban Planning*, 90(3–4), 189–195. <https://doi.org/10.1016/j.landurbplan.2008.11.003>
- Ouyang, J. Q., Isaksson, C., Schmidt, C., Hutton, P., Bonier, F., & Dominoni, D. (2018). A New Framework for Urban Ecology: An Integration of Proximate and Ultimate Responses to Anthropogenic Change. *Integrative and Comparative Biology*, 58(5), 915–928. <https://doi.org/10.1093/icb/icy110>
- Peterson, B. G., Carl, P., Boudt, K., Bennet, R., Ulrich, J., Zivot, E., Cornilly, D., Hung, E., Lestel, M., Balkissoon, K., Wertz, D., Alexander, A., Martin, R. D., Zhou, Z., & Shea, J. M. (2019). Package “PerformanceAnalytics” (1.5.3). CRAN - Rproject. <https://cran.r-project.org/web/packages/PerformanceAnalytics/index.html>
- Pickett, S. T. A. (1989). Space-for-time substitution as an alternative to long-term studies. In G. E. Likens (Ed.), *Long-Term Studies in Ecology: Approaches and Alternatives* (pp. 110–135). Springer-Verlag New York Inc. <https://doi.org/10.1007/978-1-4615-7358-6>
- Puppim de Oliveira, J. A., Balaban, O., Doll, C. N. H., Moreno-Peñaranda, R., Gasparatos, A., Iossifova, D., & Suwa, A. (2011). Cities and biodiversity: Perspectives and governance challenges for implementing the convention on biological diversity (CBD) at the city level. *Biological Conservation*, 144, 1302–1313.

<https://doi.org/10.1016/j.biocon.2010.12.007>

QGIS Development Team. (2023). QGIS Geographic Information System (Verona). Open Source Geospatial Foundation Project. <http://qgis.osgeo.org>

Quintero, E., Benavides, A. M., Moreno, N., & González-Caro, S. (2017). Bosques Andinos: Estado actual y retos para su conservación en Antioquia (Andean forests: conservation status and challenges in Antioquia) (E. Quintero Vallejo, A. M. Benavides, N. Moreno, & S. Gonzalez-Caro (eds.); 1ra ed.). Fundación Jardín Botánico de Medellín Joaquín Antonio Uribe- Programa Bosques Andinos (COSUDE). http://www.bosquesandinos.org/wp-content/uploads/2018/01/Libro_Bosques_Andinos_Interactivo.pdf

R Core Team. (2019). The R Stats Package (3.7.0). R Core Team. <https://stat.ethz.ch/R-manual/R-devel/library/stats/html/00Index.html>

Rahbek, C., Borregaard, M. K., Colwell, R. K., Dalsgaard, B., Holt, B. G., Morueta-Holme, N., Nogues-Bravo, D., Whittaker, R. H., & Fjeldsa, J. (2019). Humboldt's enigma: What causes global patterns of mountain biodiversity? *Science*, 365(September), 1108–1113. <https://doi.org/10.1126/science.aax0149>

Sax, D. F., & Gaines, S. D. (2003). Species diversity: From global decreases to local increases. *Trends in Ecology and Evolution*, 18(11), 561–566. [https://doi.org/10.1016/S0169-5347\(03\)00224-6](https://doi.org/10.1016/S0169-5347(03)00224-6)

Shochat, E., Warren, P. S., Faeth, S. H., McIntyre, N. E., & Hope, D. (2006). From patterns to emerging processes in mechanistic urban ecology. *Trends in Ecology and*

Evolution, 21(4), 186–191. <https://doi.org/10.1016/j.tree.2005.11.019>

Shwartz, A., Muratet, A., Simon, L., & Julliard, R. (2013). Local and management variables outweigh landscape effects in enhancing the diversity of different taxa in a big metropolis. *Biological Conservation*, 157, 285–292.

<https://doi.org/10.1016/j.biocon.2012.09.009>

Shwartz, A., Shirley, S., & Kark, S. (2008). How do habitat variability and management regime shape the spatial heterogeneity of birds within a large Mediterranean urban park? *Landscape and Urban Planning*, 84(3–4), 219–229.

<https://doi.org/10.1016/j.landurbplan.2007.08.003>

Smith, M. R., Parker, M. J., Schaefer, J. A., Jokimäki, J., Jukka, S., Marja-Liisa, K. J., Vanegas-Guerrero, J., Fernández, C., Buitrago-González, W., Vargas-Salinas, F., Ryder, O. A., Thomas, S., Judson, J. M., Romanov, M. N., Dandekar, S., Papp, J. C., Sidak-loftis, L. C., Walker, K., Stalis, I. H., ... Banfalvi, G. (2021). Quantifying biodiversity : procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters*, 4(2), 1–6. <https://doi.org/10.1016/j.jnc.2019.125717>

Sol, D., González-Lagos, C., Moreira, D., Maspons, J., & Lapiedra, O. (2014).

Urbanisation tolerance and the loss of avian diversity. *Ecology Letters*, 17(8), 942–950. <https://doi.org/10.1111/ele.12297>

Sutherland, W. J., Newton, I., & Green, R. E. (2004). *Bird Ecology and Conservation: A Handbook of Techniques*. Oxford University Press.

Threlfall, C. G., Mata, L., Mackie, J. A., Hahs, A. K., Stork, N. E., Williams, N. S. G., &

Livesley, S. J. (2017). Increasing biodiversity in urban green spaces through simple vegetation interventions. *Journal of Applied Ecology*, 54(6), 1874–1883.

<https://doi.org/10.1111/1365-2664.12876>

Universidad Nacional de Colombia. (2005). Plan de ordenación y manejo de la cuenca del Río Aburrá (POMCA) (Territorial planning and management of the Aburrá River watershed). http://www.cornare.gov.co/documentos/POMCA_ABURRA.pdf

TABLES AND FIGURES

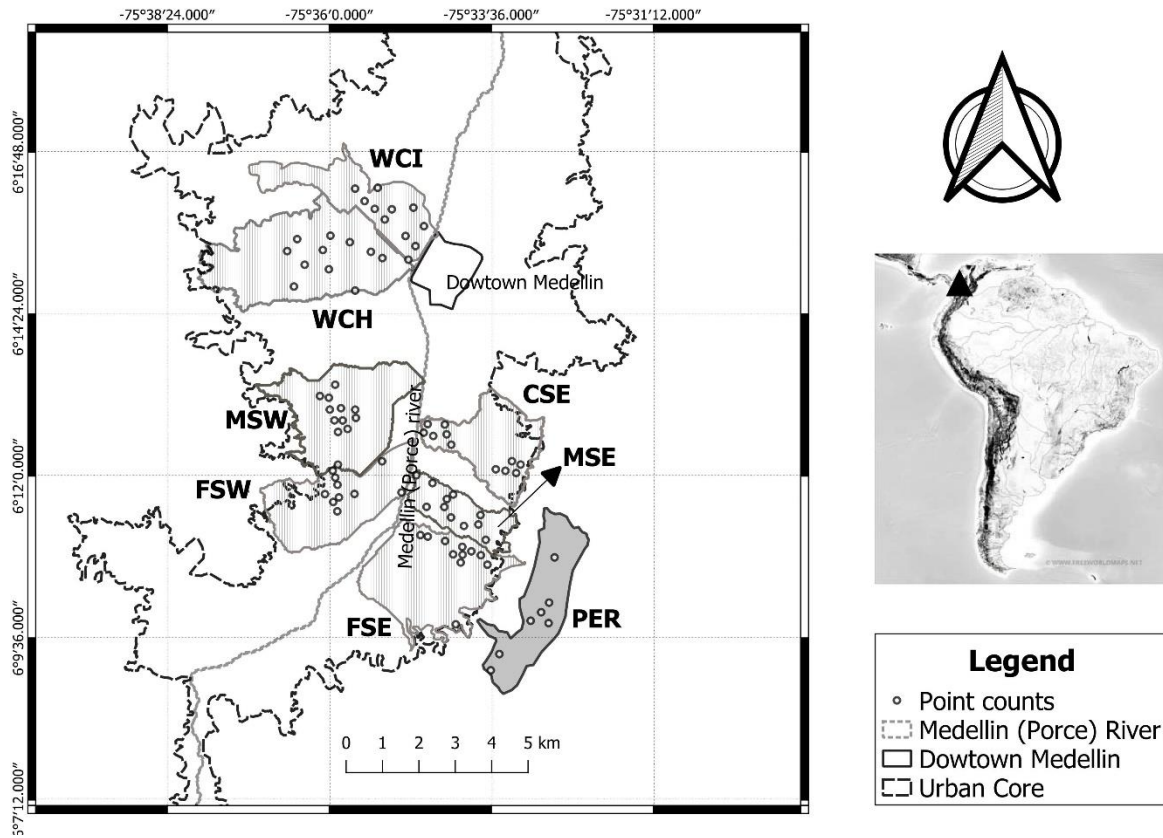


Fig 1. Study area: sites delimited for local bird diversity analysis in the Aburrá Valley (Colombia): periurban reference site (PER), and seven urban micro-watersheds: western center “Iguaná” (WCI), western center “Hueso” (WCH), middle southwest far (MSW), far south west (FSW), close south east (CSE), middle south east (MSE), far south east (FSE).

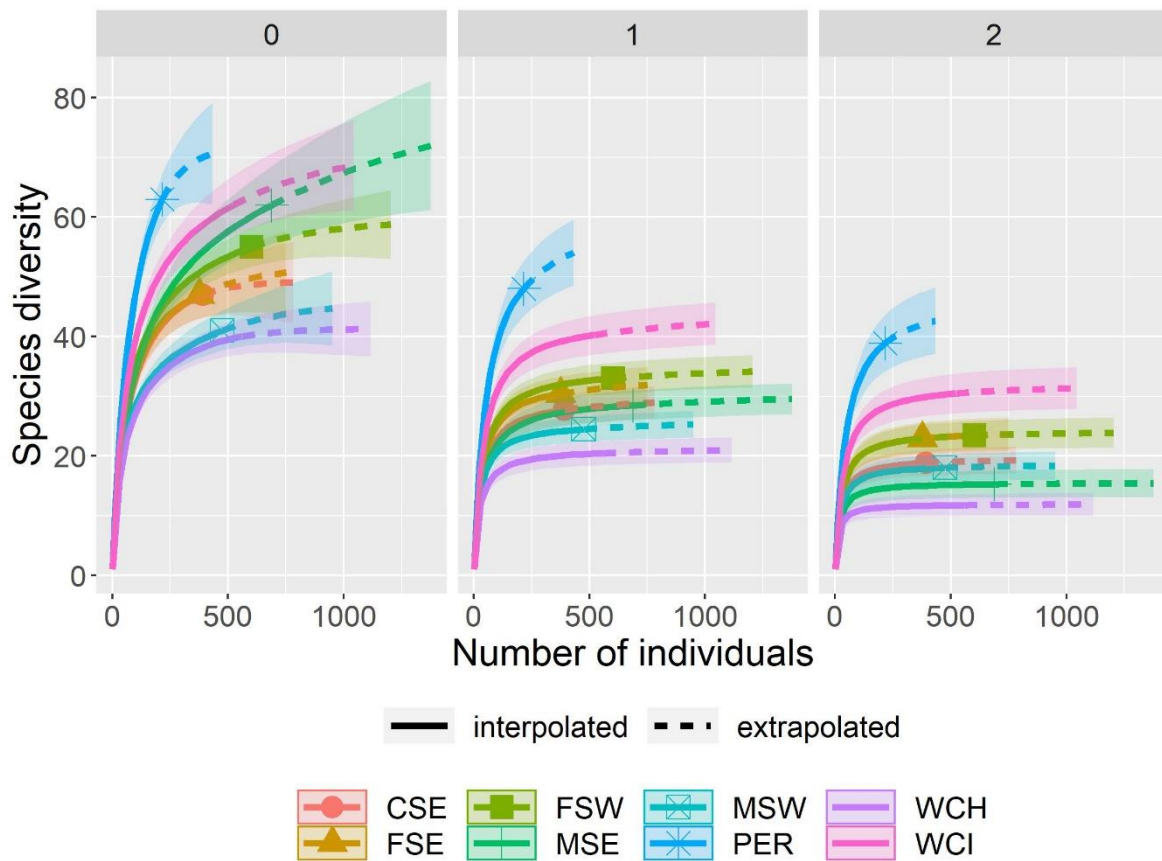


Fig 2. Sample-size-based rarefaction/extrapolation curves of resident bird assemblages across seven neighborhoods (CSE, WCH, FSE, MSE, MSW, FSW, WCI) and a periurban reference site (PER) in the Aburrá Valley (Colombia). Order $q = 0$ (Species richness), $q = 1$ (Shannon diversity: number of common species) and $q = 2$ (Simpson diversity: number of very common species) are plotted based on abundance bird data, with 95% confidence intervals.

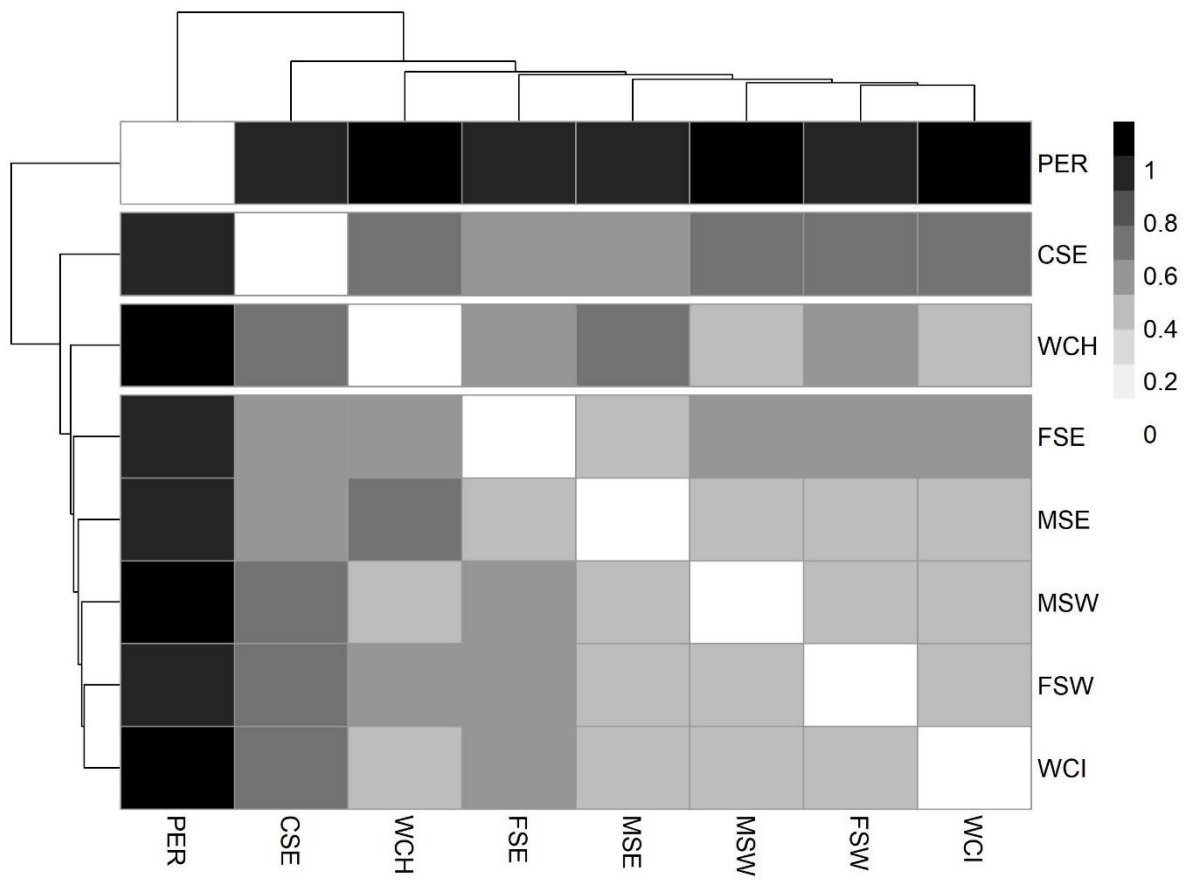


Fig 3. Heat map with Hellinger distances and UPGMA clustering diversity patterns of resident bird assemblages across seven neighborhoods (CSE, WCH, FSE, MSE, MSW, FSW, WCI) and a periurban reference site (PER) in the Aburrá Valley (Colombia).

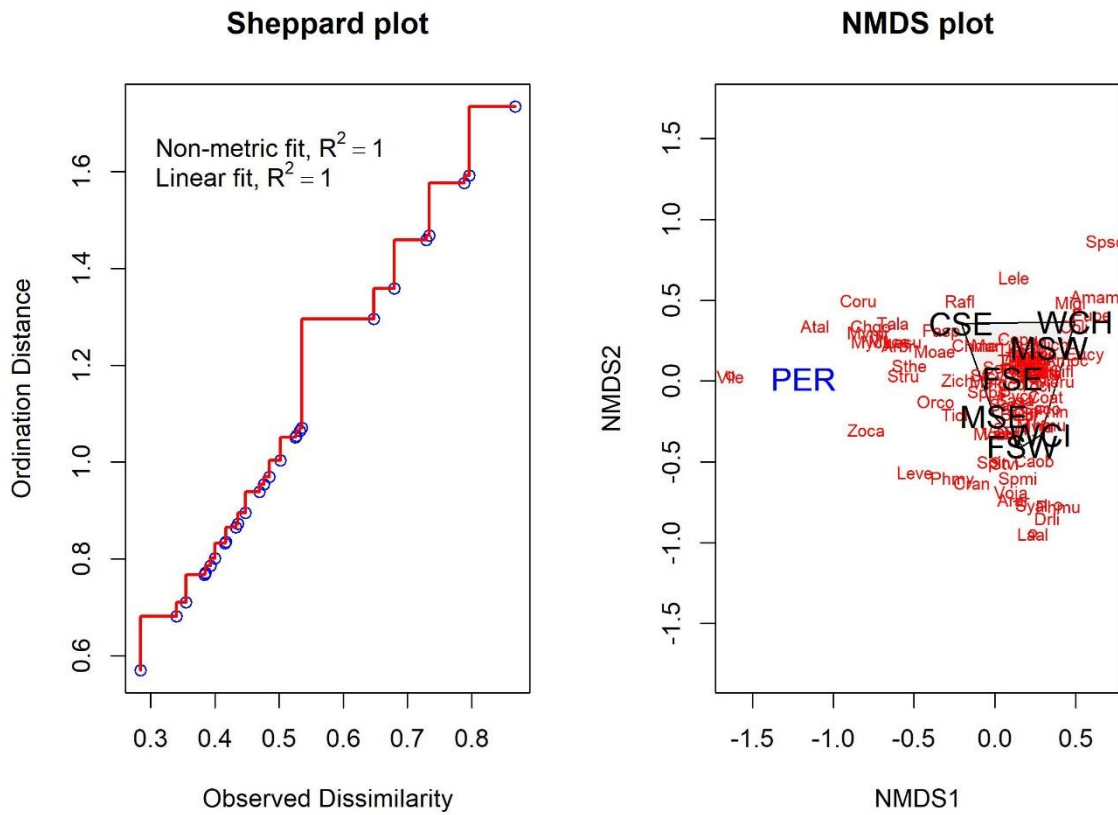


Fig 4. Sheppard and non-metric multidimensional scaling (NMDS) plots from the abundance matrix of resident bird assemblages across seven neighborhoods (CSE, WCH, FSE, MSE, MSW, FSW, WCI) and a periurban reference site (PER) in the Aburrá Valley (Colombia).

Table 1. Best supported models for evaluating resident bird species richness (BR, $q = 0$), Shannon diversity (SH, $q = 1$) and Simpson diversity (SI, $q = 2$), based on Hill numbers, across seven neighborhoods in the Aburrá Valley (Colombia). Models are compared using the Akaike Information Criterion corrected for small sample sizes (AICc) and only models with $\Delta AICc < 2.0$ are shown, with their null (Null dev.) and residual deviances (Res. dev.), and adjusted coefficient of determination (Adj-R²). Significance of variables for each model are marked as ***: $p < 0.001$, **: $p < 0.01$, and *: $p < 0.05$ (variables without marks had $p \geq 0.05$).

Model	Estimate	SE	z-value	AICc	$\Delta AICc$	Null dev.	Res. dev.	Adj-R ²
BR ~ Built + MaxTreesA				54.288	0.000	22.303	7.239	0.676
Built***: Built-up areas	-0.002	<0.001	-3.763					
MaxTreesA*: max Trees patch area	<0.001	<0.001	-2.350					
SH ~ Grass				46.478	0.000	9.319	5.656	0.447
Grass: grass percentage	0.022	0.012	1.868					
SH ~ 1				48.141	1.663	9.319	9.319	-
SH ~ MBuiltP				48.180	1.702	9.319	7.359	0.247
MBuiltP: mean built patch size	<-0.001	<0.001	-1.378					
SI ~ Grass				45.341	0.000	10.554	7.712	0.340
Grass: grass percentage	0.024	0.015	1.641					
SI ~ 1				46.182	0.936	10.554	10.554	-
SI ~ MGrassP				47.118	1.778	10.554	9.489	0.100
MGrassP	<0.001	<0.001	1.05					