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Not too small to matter: park area, bird diversity, and nestedness in pocket parks of an arid city

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Research Article

Keywords: Functional diversity, urban green spaces, vegetation structure

Posted Date: November 13th, 2026

DOI: https://doi.org/10.21203/rs.3.rs-11526355/v1

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Additional Declarations: No competing interests reported.

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Abstract

Urbanization is a major driver of biodiversity loss, making urban green spaces increasingly important for conservation. In arid cities, where water scarcity limits the establishment of large parks, small green areas may be particularly relevant. We examined how park attributes and anthropogenic noise influence taxonomic and functional bird diversity across 15 pocket parks in La Serena, Chile. Additionally, we assessed whether bird assemblages exhibited nestedness along the park-area gradient and whether this pattern could be explained by passive sampling, park characteristics, or species traits. We recorded 29 bird species. Species richness and Shannon diversity increased with park area, whereas noise had weak negative effects. Functional richness increased with park area, functional evenness declined, and functional divergence showed no clear response. Bird assemblages exhibited strong nestedness, with smaller parks supporting subsets of species found in larger ones. However, nestedness exceeded expectations under an equiprobable null model but not under a fixed–fixed model, indicating that the observed structure could be explained by differences in park richness and species occurrence frequencies. Passive sampling did not explain the area–richness relationship. Park area and tree abundance were associated with nestedness rank, whereas species rank varied with habitat preference but not morphological traits. Larger parks supported greater taxonomic diversity, while smaller parks primarily retained widespread, urban-adapted species. These findings highlight the importance of maintaining larger green spaces and tree abundance to conserve urban bird diversity in dryland cities.

Keywords: Functional diversity, urban green spaces, vegetation structure

Introduction

Urbanization is a major driver of habitat loss and biodiversity decline (Angel et al. 2011; Ikin et al. 2013; McKinney 2008; Sol et al. 2020). At a local scale, however, cities can host significant biodiversity through the planning, conservation, and management of urban green spaces (Aronson et al. 2017; Spotswood et al. 2021). Among these, parks are increasingly recognized as key elements for conserving and promoting urban biodiversity (Nielsen et al. 2014). Most research has focused on the role of large parks (>2 ha), typically with dense and tall vegetation, while the potential role of smaller elements, such as plazas or 'pocket parks' (Nielsen et al. 2014) in supporting urban biodiversity is less studied (Carbó-Ramírez and Zuria 2011; Ikin et al. 2013; Jasmani et al. 2017; Strohbach et al. 2013). Although pocket parks may support fewer species than large parks, their cumulative contribution to urban biodiversity can be substantial, particularly when distributed across diverse urban contexts and providing complementary habitat resources (Lepczyk et al. 2017). This contribution may be particularly relevant for birds, an important component of urban biodiversity because of the ecosystem services they provide (Sekercioglu 2006).

Bird diversity in urban parks is influenced by park area (Jokimäki 1999; Fernandez-Juricic and Jokimäki 2001; Schütz and Schulze 2015), habitat diversity (Panda et al. 2021; Yang et al. 2020), and vegetation structure (Morelli et al. 2017). Among these, park area frequently emerges as the primary driver of taxonomic diversity (Chaiyarat et al. 2019; Fernández-Juricic 2000; Jasmani et al. 2017), consistent with island biogeography theory and the species–area relationship

(MacArthur and Wilson 1967). Habitat diversity and vegetation structure, which often co-vary with park size, can also shape bird assemblages (Ricklefs and Lovette 1999; Panda et al. 2021; Carbó-Ramírez and Zuria 2011) by providing a broader range of ecological niches (Kallimanis et al. 2008; Stein et al. 2014; Beninde et al. 2015). Accordingly, parks with tall trees or multi-layered vegetation tend to support greater bird richness than those dominated by uniform vegetation types (Morelli et al. 2017; Yang et al. 2020).

Area gradients can also generate nested species distribution patterns, in which species present in smaller parks form ordered subsets of those in larger, richer ones (Atmar and Patterson 1993). Several processes produce this pattern. First, nestedness can arise from passive sampling because rare species are less likely to occur in smaller areas (Higgins et al. 2006). Second, it can result from selective extinction, when species with large area or resource requirements are lost from smaller parks (Wright et al. 1997; Tan et al. 2021), or from selective colonization, when dispersal limitation prevents species from reaching isolated parks (Cook and Quinn 1995; Lomolino 1996). Finally, nestedness can reflect differences among parks in habitat availability, when smaller parks contain only a subset of the habitats present in larger ones (Wang et al. 2013), or species-specific responses to human disturbance (González-Oreja et al. 2012; Tan et al. 2021). Identifying the processes associated with nestedness can therefore provide insight into the factors structuring urban bird assemblages, yet nestedness remains understudied urban environments (Chen and Cheng 2022; González-Oreja et al. 2012; Tan et al. 2021).

Anthropogenic noise can offset the benefits that park area and habitat diversity confer on urban bird assemblages (Francis et al. 2009). Road traffic is the dominant urban source, and its consequences are well documented: elevated noise reduces bird density, richness, and abundance, shift species composition and lowers community diversity (Halfwerk et al. 2011; McClure et al.

2013; Perillo et al. 2017). Proposed mechanisms include masking acoustic communication, disruption of foraging and reproductive behavior, and chronic physiological stress (Francis et al. 2009; McClure et al. 2013; Perillo et al. 2017; González-Oreja et al. 2012). If noise filters species non-randomly, it may also contribute to nested structure in urban assemblages, though this possibility remains largely untested.

The relationship between park size and functional bird diversity is less consistent than for taxonomic diversity. Schütz & Schulze (2015) reported reduced functional richness in smaller, more isolated parks, whereas Morelli and colleagues (2017) found that land-use composition and vegetation structure, rather than park area, best predicted functional diversity in Beijing's urban parks. Functional traits can help explain these contrasting responses because urban environments filter species according to disturbance tolerance, dietary flexibility, and behavioral adaptability, potentially driving functional homogenization even when taxonomic diversity is maintained (Sol et al. 2020). The same filtering processes may also determine which species persist in small parks. Thus, traits associated with species position in a nested hierarchy can help identify whether nestedness is associated with area sensitivity, habitat filtering, or colonization history (Croci et al. 2008; Neate-Clegg et al. 2023).

In arid and semi-arid cities, the water demand of irrigated vegetation can constrain the establishment and maintenance of large green spaces (Gober et al. 2009; Jenerette et al. 2011), increasing the relevance of smaller parks. These parks can concentrate water, shade, and food resources that are scarce in the surrounding urban matrix (Carbó-Ramírez and Zuria 2011). As dryland cities continue to expand, the design and management of their green spaces may become increasingly important for conserving regional biodiversity (Ren et al. 2022). La Serena, on the southern edge of the Atacama Desert in north-central Chile, provides a relevant example. The city's

population grew by approximately 13% between the 2017 and 2024 censuses (Biblioteca del Congreso Nacional de Chile 2024) accompanied by continued peripheral expansion. Pocket parks have been incorporated into new residential developments, yet little is known about which of their attributes support bird diversity. Addressing this gap is relevant to Target 12 of the Kunming-Montreal Global Biodiversity Framework, which calls for increasing the area, quality, and connectivity of urban green and blue spaces through biodiversity-inclusive planning (GBF 2022) and therefore requires evidence on the attributes that enhance their biodiversity value.

Here we examine how the physical attributes of pocket parks shape taxonomic and functional bird diversity in La Serena. We further address whether assemblages exhibit a nested structure, and, if so, which park attributes and species traits are with their positions in the nested hierarchy. We hypothesize that bird diversity is shaped by park area and vertical vegetation structure, habitat diversity, and anthropogenic noise, but that taxonomic and functional diversity respond differently to these factors. We predict that species richness and Shannon diversity increase with park area and vertical structural complexity, reflecting greater resource availability and niche space, and decrease with anthropogenic noise, reflecting acoustic disruption of communication and avoidance by noise-sensitive species. For functional diversity, we expect park area to increase functional richness by allowing the occurrence of species occupying a broader range of trait space, to decrease functional evenness as abundance becomes concentrated among dominant generalist species, and have little influence of functional divergence. Finally, we predict a nested structure beyond random expectation, with smaller parks supporting subsets of the species found in larger parks, and assess whether this pattern can be explained by passive sampling or is associated with park attributes and species traits.

Materials and Methods

Study Area

We conducted this study in La Serena, Chile (29°54'28'' S, 71°15'15'' W), from May through November 2022. La Serena lies at approximately 140 m above sea level and is characterized by native xerophytic scrubland vegetation (Luebert and Pliscoff 2018). The climate is arid, with a mean annual maximum and minimum temperatures of 19.2°C and 10.8°C, respectively. Over the past decades (2004-2024), mean annual precipitation has been 73.4 mm, ranging from 16.9 and 194.5 mm during an El Niño event (CEAZAMET 2025). In 2024, the population of La Serena was approximately 250,000 (Biblioteca del Congreso Nacional de Chile 2024). The commune includes both rural and urban areas and covers a total area of 1,892.8 km^2. The urban area spans for 90.7 km^2 and concentrates 90.8% of the district's population.

Pocket Park Selection and Characterization

We selected 15 pocket parks (Figure 1) based on two criteria: (1) a wide range of sizes and (2) a minimum distance of 500 m between parks to avoid spatial autocorrelation. The Plaza de Armas, , La Serena's central park, (P01) served as the reference point for sampling, and Euclidean distances were calculated from this location to all other selected parks.

The area of each park was determined using Google Earth Pro images. To estimate the number of vegetation strata in each park, we applied the modified point-intercept method described by Threlfall et al. (2016), laying out parallel transects spaced 5 m apart across each park. At 5 m intervals along each transect, we recorded ground cover as either bare ground or vegetation. Where vegetation was present, we measured its height using a PVC pipe-marked at five intervals: 0.0–0.2 m, 0.2–0.5 m, 0.5–1.0 m, 1.0–2.0 m, and >2.0 m. For each park, we calculated the number of height strata with vegetation, and strata diversity using the Shannon diversity index (H'), based on

154 the relative frequency of each stratum. We also counted the number of trees with a diameter at
155 breast height (DBH) greater than 8 cm (Table 1).



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157 **Figure 1.** Spatial distribution of the 15 selected pocket parks in La Serena, Chile. Orange dots
158 indicate park locations, and the corresponding photographs illustrate each park.

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Table 1. Physical and vegetation characteristics of the 15 selected pocket parks in La Serena, Chile.

Park ID	Area (m²)	Total N° of trees	Bare ground Fraction	Number of vegetation strata	Diversity of vegetation strata (H')	Average Noise (db)
P01	14270	120	0.54	4	0.884	57.11
P03	1254	15	0.46	4	0.950	43.12
P04	394	2	0.35	2	0.647	48.84
P07	2433	9	0.26	2	0.577	28.22
P10	1511	10	0.40	3	0.740	29.75
P11	1912	8	0.48	4	0.944	45.00
P13	2516	17	0.62	2	0.662	45.18
P14	318	6	0.45	2	0.688	43.30
P16	1470	19	0.46	3	0.762	30.79
P17	2079	5	0.66	3	0.687	43.88
P24	3839	55	0.65	4	0.767	43.32
P25	979	11	0.21	5	0.837	37.78
P26	1846	3	0.33	6	1.408	49.87
P28	5060	4	0.50	3	1.001	50.46
P29	573	6	0.54	2	0.690	40.00

Ambient noise level estimation

To obtain an average ambient noise level for each pocket park (Table 1), noise was recorded immediately before and after each census. At each point count location, instantaneous noise levels (Li) were measured every 10 s over a one-minute period (González-Oreja et al. 2012) using a Smart Sensor AR854 digital sound level meter positioned at a height of 1.20 m.

Bird Sampling

We conducted bird censuses using the fixed-radius point count method (Geupel et al. 1996). In each pocket park, we established between one and five sampling points, depending on park size. At each point, all birds seen or heard within a 25 m radius were recorded over a 10-minute period. When structures obstructed visibility, limited movement within the radius was allowed to confirm detections. Birds flying above 25 m were excluded. Censuses were conducted on weekdays, twice per day: once in the morning within the first two hours after sunrise (07:00–09:00 h) and once in the afternoon within the last two hours before sunset (17:00–19:00 h). Sampling was carried out in May, July, September, and November 2022, giving a total of 10 censuses per park. All surveys were conducted by a single observer, and audio recordings were made at each point using the RecForge II Pro mobile application. Recordings were later reviewed by an expert to confirm species identifications. We calculated bird richness as the total number of species recorded across all visits and sampling points within each park, and abundance as the cumulative number of individuals recorded per park.

Taxonomic and functional diversity of birds

To assess taxonomic diversity, we calculated species richness, the Shannon diversity index (H'), and Pielou's evenness index (J) for each pocket park using the 'vegan' package (Oksanen et al. 2018) in R (Team 2025). Taxonomic identification was verified using the AVONET database, Birds of the World (Tobias et al. 2022), and Aves de Chile (Couve and Ojeda 2016).

To assess functional diversity, we compiled continuous and categorical traits related to resource acquisition and dispersal capacity for each recorded species. These included body mass, trophic niche, foraging substrate, and primary locomotion mode, among others (Table S1). From this trait

data we constructed a species-by-trait matrix integrating both variable types. Categorical traits were dummy-coded, and Gower's distance was used to compute pairwise species dissimilarities, since it accommodates mixed data types. Functional diversity indices were then calculated using the 'FD' package (Laliberté and Legendre 2010) from the Gower's distance matrix and species abundances per site. We computed three indices following Villéger et al. (2008): functional richness (FRic), functional evenness (FEve), and functional divergence (FDiv). FRic represents the volume of functional space occupied by the community and is independent of species abundances. FEve quantifies the regularity with which species and their abundances are distributed within this space. FDiv reflects the extent to which abundances are concentrated toward the extremes of the functional space relative to its centroid, with higher values indicating dominance by functionally distinct species.

Data analyses

Sampling completeness and spatial independence

All analyses were performed in R version 4.5 (Team 2025). We assessed the completeness of bird sampling using species accumulation curves implemented in the 'BiodiversityR' package, and evaluated spatial autocorrelation among parks using Moran's I index from the 'ape' package.

Relationships between pocket park attributes and bird diversity patterns

To examine the relationship between pocket park characteristics and different components of bird diversity, we used Generalized Linear Models (GLMs). Prior to model fitting we assessed collinearity among predictor variables, park area (log m^2), number of trees, bare ground fraction, number and diversity of vegetation strata, and noise using the Variance Inflation Factor (VIF) from the 'car' package (Fox et al. 2012). The number of vegetation strata and Shannon diversity of strata

were strongly correlated ($r = 0.83$) and had VIF values of 5.12 and 5.90, respectively. We therefore excluded strata diversity from the model, after which no predictor exceeded a VIF of 2.1 (Zuur et al. 2010; Dormann et al. 2013). A sensitivity analysis excluding either strata variable yielded the same top-ranked model for all six response variables.

Separate GLMs were fitted for each response variable: log-transformed species richness, Shannon diversity, functional richness, functional evenness, and functional divergence. All models assumed a Gaussian error distribution with an identity link. Model selection was based on the Akaike Information Criterion corrected for small sample sizes (AICc), using the 'MuMIn' package (Bartoń 2022). Models with $\Delta AICc < 2$ were considered competitive, and where more than one competitive model was found, model averaging was applied to obtain parameter estimates and their relative importance (Burnham and Anderson 2002).

Diversity and patterns of nestedness structure

To assess whether bird assemblages in pocket parks exhibited a nested structure, we applied the NODF metric (nestedness metric based on overlap and decreasing fill) to presence–absence matrices derived from species abundance data. Following the definition of NODF, rows and columns were sorted by decreasing marginal totals before calculating nestedness (Almeida-Neto and Ulrich 2011). We then tested whether the resulting park order reflected the area gradient using a Spearman correlation between each park's position in the packed matrix and its area. NODF values range from 0 (no nestedness) to 100 (perfect nestedness), and quantify nestedness based on species overlap and matrix fill (Almeida-Neto and Ulrich 2011). The significance of the observed nestedness was evaluated against two null models, each based on 999 randomizations. The first was an equiprobable model in which only total matrix fill was constrained, so that every cell had the same probability of occupancy. The second was a fixed–fixed model that preserved both

species frequencies and site richness, implemented with the quasiswap algorithm (Miklós and Podani 2004). The two models constrain different properties of the matrix, and equiprobable models are prone to Type I error when marginal totals are heterogeneous (Ulrich and Gotelli 2013, 2007), so we report both. For each we calculated the standardized effect size as $SES = (NODF_{obs} - \overline{NODF}_{sim})/SD(NODF_{sim})$.

To test whether nestedness could be explained by passive sampling, we applied the random placement model (Higgins et al. 2006), which predicts expected species richness from park area and regional species abundances. The passive sampling hypothesis was rejected when more than one-third of the observed richness values fell outside ±1 standard deviation from model predictions (Coleman 1981; Coleman et al. 1982).

To identify environmental drivers of nestedness, we calculated paired Spearman correlations between species richness ranks and ranked environmental attributes, including area, vegetation structure, noise, and spatial isolation. Spatial isolation was quantified using two metrics: an isolation index, calculated as the mean distance to all other parks, and nearest-neighbor distance, calculated as the distance to the closest park.

To assess which species traits were associated with nestedness, we analyzed the relationship between species positions in the nestedness hierarchy and their morphological and ecological traits. Species ranks were assigned according to their frequency of occurrence across parks, with the most widespread species assigned rank 1. We used paired Spearman rank correlations to test associations between species ranks and continuous morphological traits, including beak length, depth, and width; tarsus length; tail length; wing length; and body mass. For categorical traits (habitat preference, migratory status, trophic niche, and primary lifestyle) we used individual Kruskal–Wallis tests to evaluate variation in species ranks among trait categories.

Results

Sampling completeness and spatial independence

Individual-based rarefaction curves revealed substantial variation among parks in both expected species richness and sampling effort (Figure S1). Parks with more recorded individuals generally exhibited higher expected richness and, in several cases, curves approaching an asymptote (P01, P03, P10, P11, P13, P17, P25, P26), indicating relatively high sampling completeness. Other parks showed lower richness with flatter curves (P14, P29), while a third group did not reach an asymptote (P04, P07, P16, P24, P28), suggesting that additional sampling would likely add new species.

Spatial autocorrelation analyses showed no evidence of geographic structure in bird diversity among parks. Moran's I values were weakly negative and did not differ from zero for either species richness (Moran's I = -0.101, p = 0.649) or Shannon diversity (Moran's I = -0.068, p = 0.957).

Relationships between pocket park attributes and patterns of bird diversity

We recorded a total of 29 bird species across the 15 sampled pocket parks in La Serena (Table S2). The most frequently observed species were the House Sparrow (*P. domesticus*), the Austral Thrush (*T. falcklandii*), Rufous-collared Sparrow (*Z. capensis*), and the West Peruvian Dove (*Z. meloda*), all of which were present in all 15 parks.

Bird communities in pocket parks exhibited high variability in both taxonomic and functional diversity (Table 2). Species richness ranged from 6 to 20 species, and total abundance from 20 to 926 individuals across parks. Shannon diversity was highest in P16 (H'= 2.455) and lowest in P04 (H' = 1.259), while Pielou's evenness was highest in P17 (J'= 0.931) and lowest in P04 (J'= 0.647). Functional diversity also varied among parks. FRic spanned several orders of magnitude, from

281 8.724×10^{-9} in P14 to 2.285×10^{-3} in P01. FEve ranged from 0.495 (P01) to 0.891 (P29), while
282 FDiv ranged from 0.748 in P01 to 0.926 in P03 (Table 2).

283

284 **Table 2.** Taxonomic and functional diversity metrics of bird communities in the 15 selected pocket
285 parks of La Serena, Chile. FRic: Functional richness, FEve: functional evenness, FDiv: functional
286 divergence.

Park	Richness	Abundance	Shannon (H')	Pielou (J')	FRic	FEve	FDiv
P01	20	926	2.140	0.715	2.285×10^{-3}	0.495	0.748
P03	11	59	2.172	0.906	2.872×10^{-4}	0.793	0.926
P04	7	20	1.259	0.647	8.822×10^{-6}	0.667	0.919
P07	14	77	2.133	0.808	1.111×10^{-3}	0.621	0.824
P10	12	114	2.011	0.809	2.006×10^{-4}	0.666	0.875
P11	11	95	2.050	0.855	1.645×10^{-5}	0.694	0.806
P13	15	142	2.383	0.880	1.250×10^{-3}	0.511	0.856
P14	6	25	1.522	0.850	8.724×10^{-9}	0.891	0.830
P16	15	76	2.455	0.906	9.892×10^{-4}	0.750	0.815
P17	12	102	2.315	0.931	2.351×10^{-4}	0.758	0.801
P24	16	229	2.265	0.817	7.425×10^{-4}	0.564	0.870
P25	13	65	2.312	0.901	2.443×10^{-4}	0.839	0.860
P26	13	108	2.204	0.859	5.659×10^{-4}	0.656	0.778
P28	13	125	2.133	0.832	1.576×10^{-3}	0.555	0.830
P29	9	41	2.035	0.926	5.667×10^{-5}	0.837	0.852

287

288 Model selection revealed that park area was the most consistent predictor of taxonomic bird
289 diversity, with positive effects on both species richness and Shannon diversity (Table 3, Figure 2).
290 Ambient noise had small negative effects on both metrics, whereas the number of vegetation strata
291 showed a moderate positive effect on Shannon diversity (Table 3) but had little influence on species

richness (Figure 2). Pielou's evenness was best explained by the intercept-only model, with a weak negative effect of noise in the second-best model (Table 3).

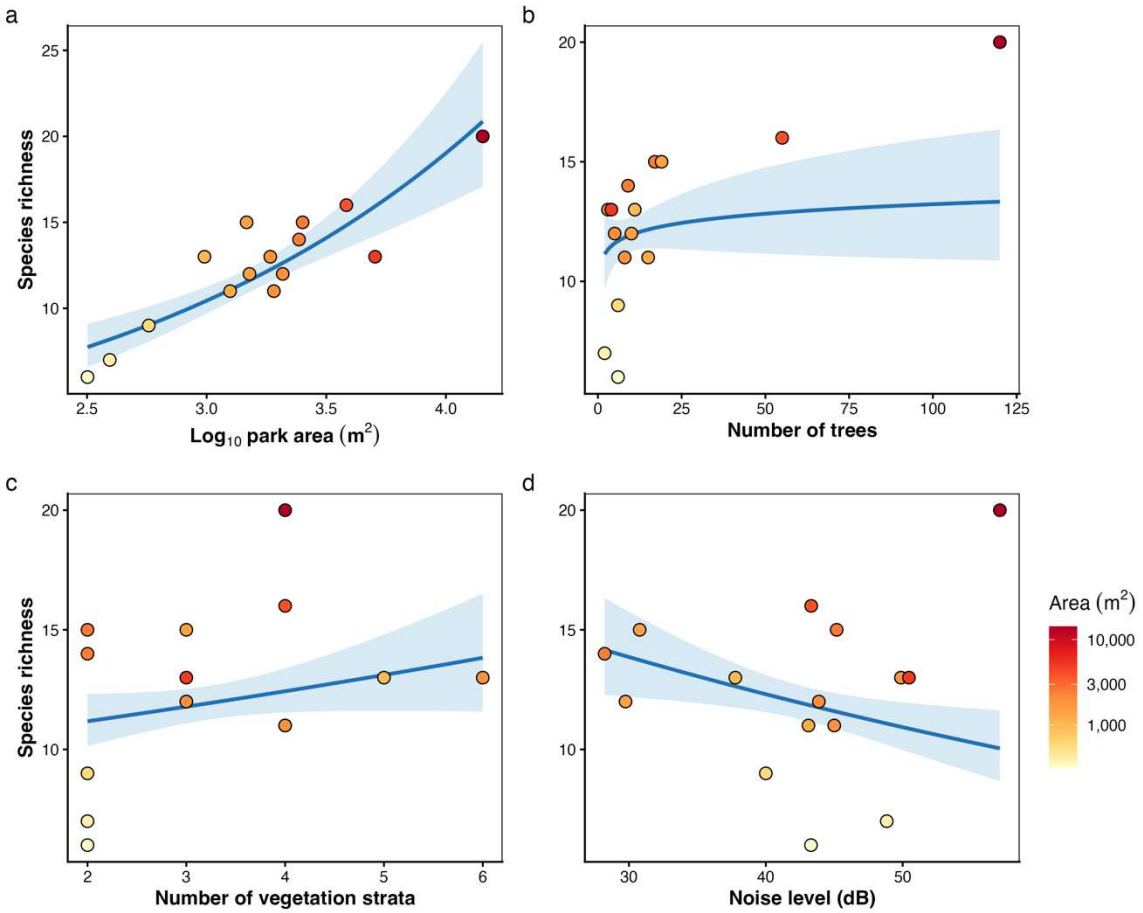


Figure 2. Relationships between bird species richness with: a) Park Area; b) Number of trees; c) Number of vegetation strata and; d) Ambient noise level in 15 pocket parks of La Serena, Chile. Shaded areas represent 95% confidence intervals.

Functional diversity metrics showed weaker and less consistent responses to environmental variables than taxonomic diversity. Park area was positively associated with FRic but negatively associated with FEve (Table 3, Figure 3). For FDiv support was divided between a model including with a weak negative effect of park area and the intercept-only model.

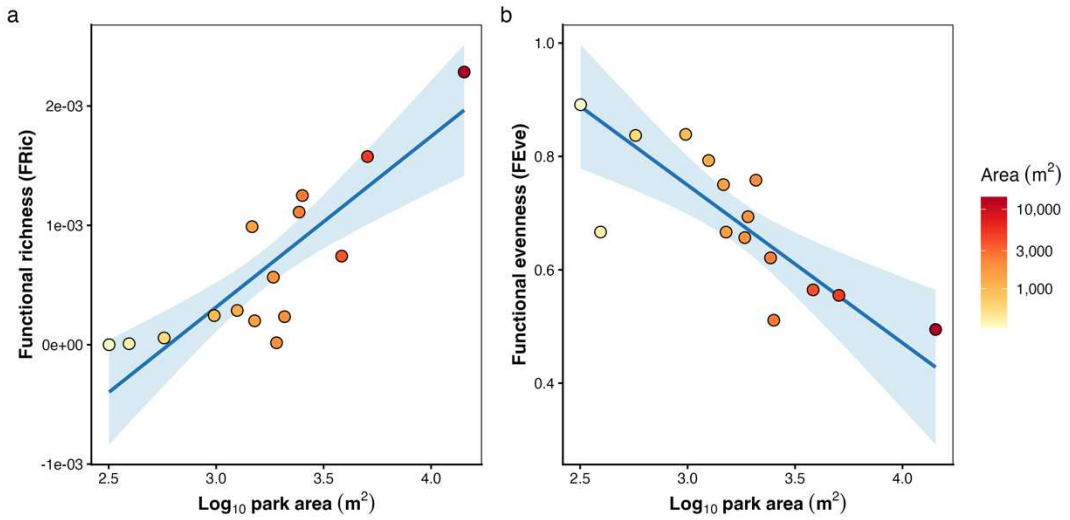


Figure 3. Relationships between functional diversity metrics and park area in 15 urban parks of La Serena, Chile. a) Functional richness (FRic) and b) Functional evenness (FEve). Shaded areas represent 95% confidence intervals of the fitted regression lines.

306 Table 3. Model selection results for predicting taxonomic (richness [S], Shannon diversity [H'], and Pielou's evenness index [J']) and
307 functional diversity (FRic, FEve, FDiv) of birds in 15 pocket parks of La Serena. Model-averaged estimates are shown with their adjusted
308 standard errors (SE). AICc = Akaike Information Criterion corrected for small samples; Weight = Akaike weight. Only models with
309 ΔAICc < 2 are shown. FRic model coefficients and adjusted SE were multiplied by 10^3 for presentation.

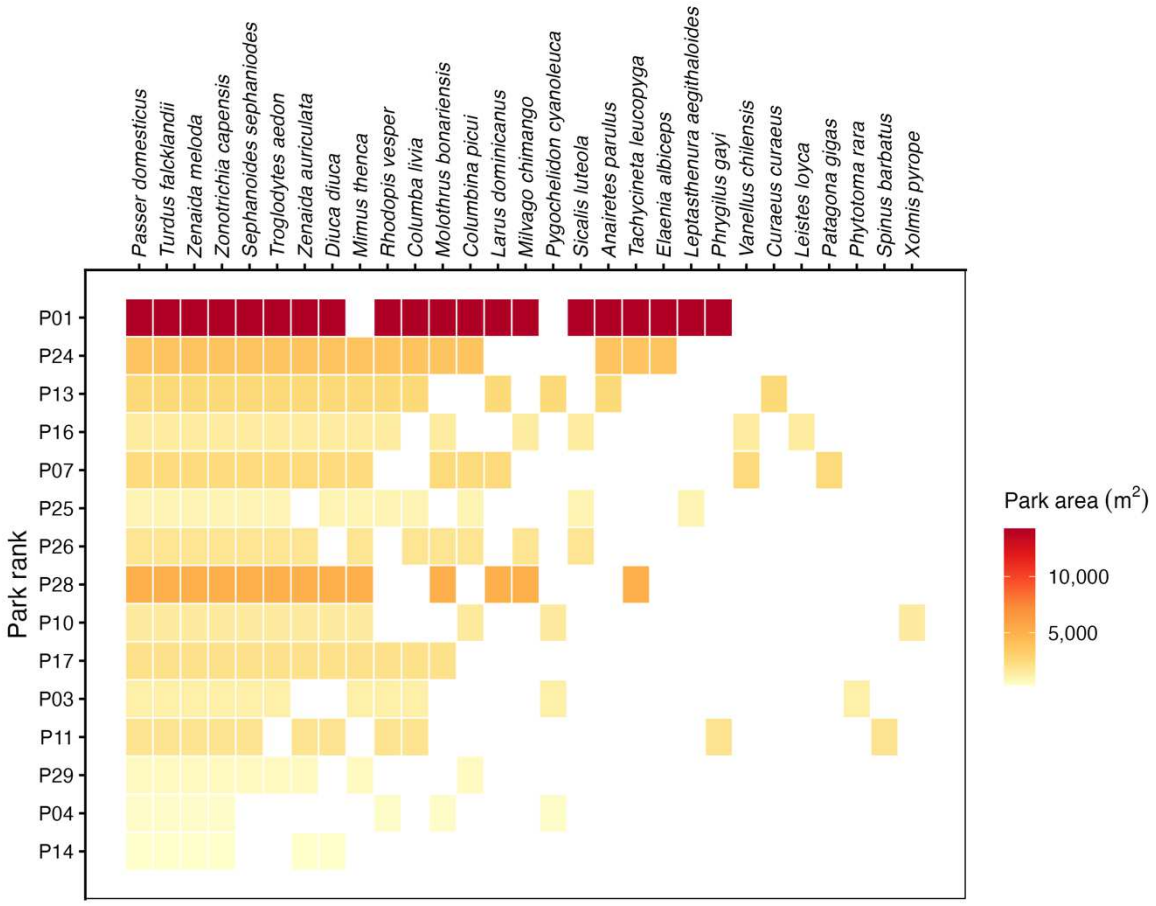
Diversity metric	Intercept	Log_{10} area	Log No. trees	Bare ground fraction	No. veg. strata	Noise	df	logLik	AICc	weight
Richness	0.631	0.720				-0.011	4	11.533	-11.065	0.342
	0.653	0.681			0.051	-0.013	5	13.652	-10.638	0.276
Average	0.61 ± 0.33	0.68 ± 0.11	0.01 ± 0.03	-0.01 ± 0.12	0.02 ± 0.03	-0.01 ± 0.01				
Shannon	1.075	0.540				-0.017	4	1.874	8.252	0.212
	0.687	0.436					3	-0.380	8.942	0.150
	1.111	0.474			0.086	-0.020	5	3.487	9.692	0.103
	1.122	0.348		1.039	0.136	-0.025	6	6.399	9.702	0.103
Average	1.19 ± 0.69	0.34 ± 0.26	0.01 ± 0.05	0.26 ± 0.57	0.05 ± 0.07	-0.01 ± 0.01				
Pielou	0.843						2	17.450	-29.901	0.276
	0.985					-0.003	3	18.421	-28.659	0.148
Average	0.889 ± 0.131	-0.005 ± 0.027	-0.001 ± 0.010	0.052 ± 0.134	0.004 ± 0.012	-0.002 ± 0.003				
FRic × 10^3	-3.727	1.353					3	97.828	-187.474	0.350
	-3.694	1.486			-0.141		4	99.470	-186.941	0.268
Average	-3.688 ± 0.884	1.421 ± 0.308	0.007 ± 0.053	-0.180 ± 0.569	-0.071 ± 0.101	0.001 ± 0.006				
FEve	1.431	-0.231					3	17.914	-27.647	0.427
	1.425	-0.254			0.025		4	19.067	-26.133	0.200
Average	1.441 ± 0.191	-0.240 ± 0.064	0.003 ± 0.013	0.002 ± 0.069	0.008 ± 0.016	-0.000 ± 0.001				
FDiv	1.034	-0.060					3	27.154	-46.126	0.343
	0.839						2	24.695	-44.390	0.144

Diversity metric	Intercept	Log_{10} area	Log No. trees	Bare ground fraction	No. veg. strata	Noise	df	logLik	AICc	weight
Average	0.973 ± 0.120	-0.038 ± 0.039	0.000 ± 0.007	-0.001 ± 0.042	-0.001 ± 0.006	-0.000 ± 0.001				

310

Determinants and structure of nestedness

Bird assemblages were strongly nested along the park area gradient (NODF = 66.82), with smaller pocket parks supporting subsets of the species found in larger parks (Fig. 4). However, statistical support for nestedness depended on the null model. Under the equiprobable model, which constrains only matrix fill, observed nestedness greatly exceeded the null expectation (null mean = 43.77, SES = 10.67, P = 0.001). In contrast, nestedness did not differ from the null expectation under the fixed–fixed model, which preserves species frequencies and site richness (null mean = 67.87, SES = −1.25, P = 0.225). Thus, the observed nested structure can be explained by variation in park richness and species occurrence frequencies, rather than by an additional process structuring species occurrences. The largest park supported the most diverse assemblage, whereas smaller parks contained fewer and more widely distributed species. Urban-adapted species such as *Passer domesticus* and *Zonotrichia capensis* occurred in all 15 parks, together with *Zenaida meloda* and *Turdus falcklandii*, whereas less common species were restricted to larger ones.



325

326 **Figure 4.** Maximally nested presence–absence matrix of bird assemblages within the 15
327 pocket parks selected in La Serena, Chile. The color inside each cell represents the gradient
328 of area of each pocket park.

329

330 The random placement model showed that nestedness of bird assemblages was not due to
331 passive sampling. Only 13.3% of pocket parks (N=2) fell within one standard deviation of
332 the expected species richness based on area alone, well below the 67.7% threshold required
333 to support the passive sampling hypothesis (Figure 5).

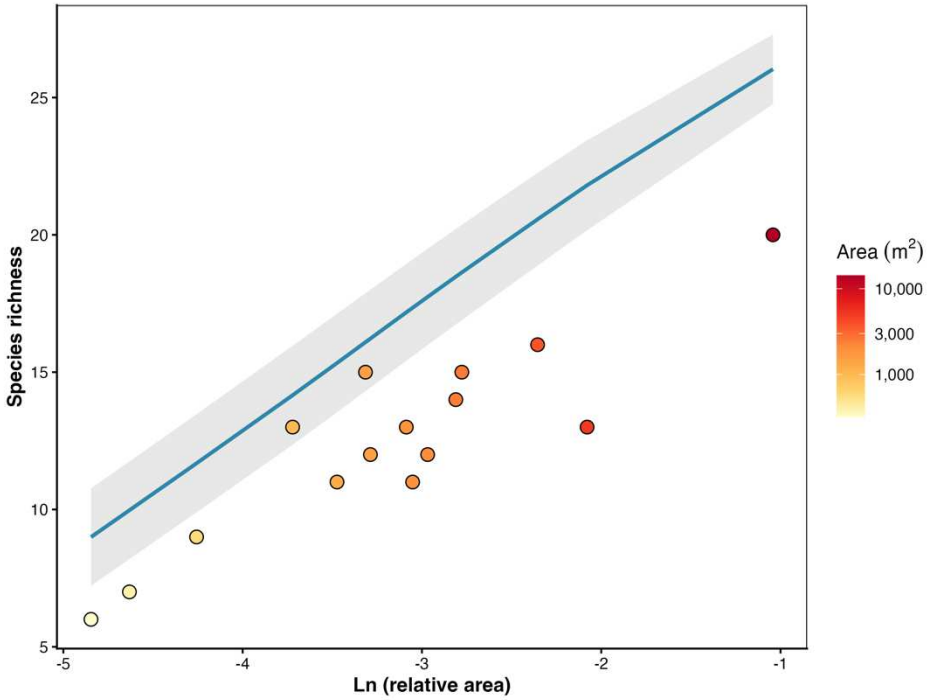


Figure 5. Relationship between observed species richness and log-transformed relative area for 15 pocket parks in La Serena, Chile. The blue line represents species richness predicted by the passive sampling model, with the shaded band indicating ±1 standard deviation. Each point corresponds to a park, colored according to its area (m²).

Spearman correlations revealed that two environmental variables were significantly associated with pocket park ranks in the nestedness hierarchy (Table 4). Park area showed the strongest negative correlation with nestedness rank, indicating that larger parks supported more diverse bird assemblages and occupied higher positions in the nested structure. Tree abundance was also negatively correlated with nestedness rank, indicating that parks with more trees tended to occupy higher positions in the nested hierarchy. None of the remaining variables was significantly associated with nestedness rank (Table 4). None of the morphological species traits was significantly associated with species rank in the nestedness

hierarchy (Table 4). Among categorical traits, species rank varied significantly only with habitat preference (Kruskal–Wallis $\chi^2 = 8.34$, df = 3, $p < 0.05$), with urban-adapted and generalist species ranking higher than habitat specialists.

Table 4. Environmental and species trait correlates of nestedness in the 15 selected pocket parks of La Serena, Chile. Significant correlations (<0.05) are shown in bold.

Environmental correlates			Species trait correlates		
Variable	**Spearman r_s**	**P-value**	**Variable**	**Spearman r_s**	**P-value**
Park area (m²)	**-0.77**	**<0.001**	Body mass (g)	-0.03	0.861
Number of trees	**-0.67**	**<0.001**	Wing length (mm)	0.08	0.697
Number of vegetation strata	-0.32	0.248	Beak length (mm)	0.07	0.726
Vegetation strata diversity	-0.15	0.596	Beak width (mm)	0.25	0.183
Bare ground fraction	-0.27	0.328	Beak depth (mm)	0.16	0.408
Noise level (dB)	-0.11	0.688	Tarsus length (mm)	0.02	0.912
Isolation index (m)	-0.04	0.889	Tail length (mm)	0.11	0.566
Nearest neighbor distance (m)	-0.37	0.172			

Discussion

Pocket parks are increasingly recognized as biodiversity contributors in cities where large green spaces are constrained (Lepczyk et al. 2017; Nielsen et al. 2014). We examined how structural attributes of 15 pocket parks in an arid city shape taxonomic and functional bird diversity, and whether assemblages form nested subsets. Park area was the strongest predictor of taxonomic diversity, increasing species richness and Shannon diversity, while ambient noise had small negative on both metrics. Functional responses were weaker and less consistent: functional richness increased slightly with area, evenness declined with area, and divergence showed only weak support for an area effect. Assemblages were nested along the

363 area gradient, although statistical support depended on the null model, and the pattern was
364 not attributable to passive sampling. Park area and number of trees were significantly
365 associated with with nestedness rank, while habitat preference, but no morphological traits,
366 was associated with position in the hierarchy.

367 *Taxonomic diversity*

368 Park area was the strongest and most consistent predictor of bird species richness and
369 Shannon diversity, a result that is consistent with island biogeography theory and with a broad
370 body of urban ecology research identifying park size as the main predictor of urban bird
371 richness (Fernandez-Juricic and Jokimäki 2001; Jokimäki 1999; MacArthur and Wilson
372 1967; Nielsen et al. 2014), including meta-analytic evidence across 75 cities identifying patch
373 area as the strongest correlate of intra-urban biodiversity (Beninde et al. 2015). Larger parks
374 likely support more species through multiple, non-exclusive mechanisms. Area increases the
375 total resource base and the number of territories a park can accommodate greater resource
376 availability (Ricklefs and Lovette 1999), larger populations face a lower probability of local
377 extinction (Soulé et al. 1988), and larger parks tend to contain greater internal structural
378 heterogeneity, even if not fully captured by our habitat metrics (Tews et al. 2004). In arid
379 cities, these mechanisms should be amplified because green spaces concentrate water, shade,
380 and food resources that are scarce in the surrounding matrix (Carbó-Ramírez and Zuria 2011).

381 Ambient noise had small but consistent negative effects on both species richness and
382 Shannon diversity. Three non-exclusive explanations may account for this result. The
383 recorded range was narrow (28–57 dB; Table 1), limiting the detection of strong biological
384 responses. Our measurements also captured short-term conditions around census periods, and
385 may not represent chronic daily to traffic or human activity. Finally, bird assemblages were

386 already dominated by urban-tolerant granivores and omnivores documented as noise-tolerant
387 across urban gradients (Francis et al. 2009; Halfwerk et al. 2011), so noise-sensitive species
388 may have been filtered out the landscape scale before park-level differences became
389 apparent.

390 *Functional diversity*

391 Functional richness increased with park area, whereas functional evenness declined with area
392 and functional divergence showed only weak support for an area effect. Functional trait space
393 varied by several orders of magnitude across parks, with the largest park (P01) occupying the
394 largest volume, although this partly reflected two nectarivores (*Rhodopis vesper*, *Patagona*
395 *gigas*) whose distinctive habit expands functional space in small assemblages. The weak area
396 response indicates that species added in larger parks were often functionally redundant, so
397 gains in richness produced limited expansion of ecological strategies. This decoupling
398 between taxonomic and functional diversity is consistent with urban functional
399 homogenization, in which cities retain species counts while filtering toward generalist,
400 dietary flexible, disturbance-tolerant traits (Sol et al. 2020). Our results contrast with the
401 positive area effect found by Schütz & Schulze (2015) in European parks, but aligns with the
402 null relationship found by Morelli et al. (2017). This may reflect the restricted regional
403 species pool, and the rapid functional niche-filling by a few dominant generalists that impose
404 a functional ceiling irrespective of park area.

405 Functional evenness declined with park area. In the largest parks, particularly P01, abundance
406 was dominated by a small set of species occupying overlapping functional space (*Zenaida*
407 *auriculata, Zenaida meloda, Columba livia, Passer domesticus, Turdus falcklandii*; Table
408 S2), concentrating abundance within a limited region of the functional hypervolume. Smaller

parks, with fewer individuals, showed more even distributions across the trait space they occupied. The same dominant generalists were also the most widespread across parks, linking this result to the nestedness pattern.

The number of vegetation strata was not retained in the best-supported functional diversity models, despite the expectation that vertical complexity expands niche availability (Stein et al. 2014). Assemblages were dominated by ground-foraging granivores and omnivores that use open substrates largely independent of vegetation structure, whereas species likely to respond to vertical complexity (e.g., *Leptasthenura aegithaloides*, *Troglodytes aedon*, *Pygochelidon cyanoleuca*; Table S2) occurred at low abundances. Our trait set also omitted fine-scale foraging microhabitat use; incorporating traits such as foraging height alongside more detailed structural variables would clarify these relationships.

Nestedness structure

Bird assemblages across the 15 pocket parks showed a strongly nested structure. However, statistical support for nestedness depended on the null model: observed nestedness exceeded expectations under the equiprobable model but not under the fixed–fixed model, indicating that the observed structure can largely be explained by variation in park richness and species occurrence frequencies. The strong association between nestedness rank and park area is consistent with predictions that area gradients in fragmented landscapes generate hierarchical biodiversity patterns (MacArthur and Wilson 1967; Matthews et al. 2015). Similar patterns have been reported in urban bird communities across regions, including East Asia and South America (Chen and Cheng 2022; Tan et al. 2021; Godoy et al. 2024; Li et al. 2025). Large parks function as reservoirs of local urban diversity, while smaller parks retain only the most tolerant or widespread species.

The random placement model did not support passive sampling as an explanation for the observed richness pattern, with only two parks falling within the expected range based on area alone. However, the fixed–fixed null model reproduced the observed degree of nestedness, indicating that variation in park richness and species occurrence frequencies is sufficient to account for the nested structure without invoking an additional nestedness-generating process. Nestedness rank was strongly associated with park area and number of trees but was unrelated to spatial isolation metrics. These relationships suggest that local habitat attributes may contribute to differences in richness and species composition among parks, whereas the measured isolation variables showed little evidence of an association with the nested hierarchy. Reductions in area and tree cover may favor tolerant, widespread species over those with greater resource or habitat requirements (Cook and Quinn 1995; González-Oreja et al. 2012).

Species positions within the nested hierarchy were unrelated to the morphological traits examined, but varied significantly with habitat preference. Urban-adapted and generalist species ranked higher than habitat specialists, indicating that tolerance of human-modified conditions is associated with greater occurrence across pocket parks, consistent with trait-based frameworks in which behavioral flexibility and generalism predict urban persistence (Croci et al. 2008; Neate-Clegg et al. 2023; Sol et al. 2020). Smaller parks thus retained primarily widespread generalists, whereas pocket larger parks also supported species with more restricted ecological requirements.

Conservation implications

These findings have three implications for urban planning. First, pocket park area should be treated as a primary design criterion: it was the strongest predictor of richness, diversity and

was strongly associated with nestedness rank, with larger parks supporting greater richness and occupying higher positions in the nested hierarchy. Second, tree number was also strongly associated with nestedness rank, making tree planting a complementary priority. Third, small parks retained mainly urban-adapted generalists, whereas species with narrower habitat requirements tended to be associated with larger parks, which therefore carry disproportionate value for the less tolerant fraction of the urban avifauna. Planning should thus recognize a functional hierarchy, with large pocket parks as biodiversity anchors, and smaller ones as supportive elements.

Some aspects of the study design and data should be kept in mind when interpreting these results. We sampled only 15 parks within a single city, so extrapolating to other urban drylands warrants caution. Some parks did not reach rarefaction asymptotes, so richness may be slightly underestimated there, although this does not alter the overall nested pattern. We also did not include landscape context, such as surrounding land cover, connectivity to other vegetated patches, or management intensity, all of which may influence local bird communities.

Conclusion

Pocket parks in La Serena sustain meaningful bird diversity, but their conservation value is hierarchical rather than uniform. Area influences how many species a pocket park holds, with larger parks supporting greater diversity and occupying higher positions in the nested hierarchy. Biodiversity-oriented planning in arid cities should prioritize larger green spaces, maintain tree cover, and recognizing the complementary role of smaller parks in supporting urban bird communities.

Acknowledgements

We thank Sofía Ochoa, and Felipe Mendez for their assistance during fieldwork. We are also grateful to all members of the Laboratorio de Ecología del Desierto for their valuable insights throughout the course of this study. Finally, we thank Angéline Bertin, Myriam Ramirez and the anonymous reviewers whose comments helped improve the manuscript.

Author contributions

María José Peralta: Writing, review & editing, Fieldwork, Data analysis, Conceptualization. Rodrigo S. Rios: Writing – review & editing, Data analysis, Conceptualization. Marco Garrido Salinas: Review & editing. Andrea P. Loayza: Writing – original draft, review & editing, Visualization, Data analysis, Supervision, Project administration, Conceptualization, Funding acquisition. All authors have read and agreed to the published version of the manuscript.

Funding

This work was supported by DIDULS PTE2153856 awarded to APL, and by Project ANID/BASAL FB210006 from the Institute of Ecology and Biodiversity (IEB), Chile, also awarded to APL. Financial support for RSR was provided by DIDULS/ULS through funding ADI2653852 and by the Agencia Nacional de Investigación y Desarrollo (ANID) through ANID-FONDECYT Regular Grants 1211181 and 1251961.

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Supplementary Material

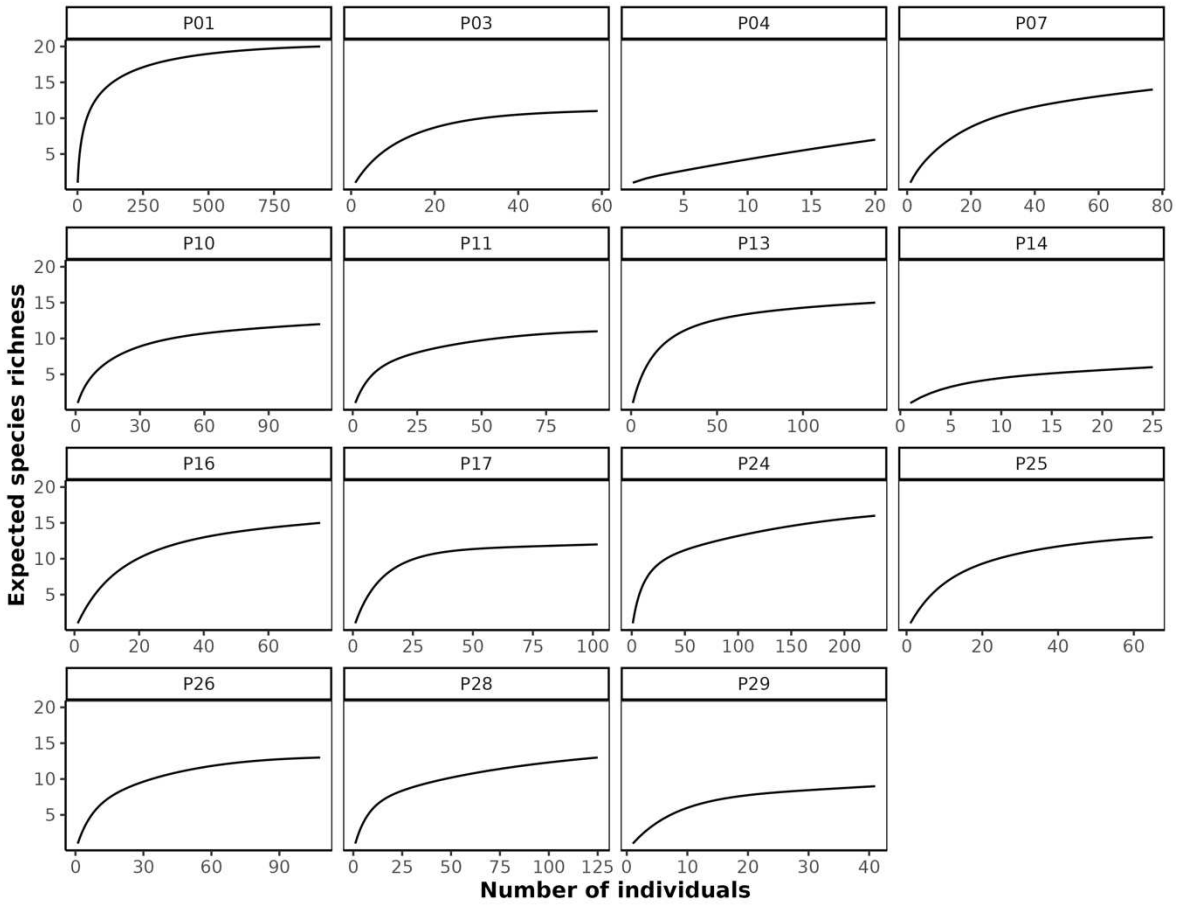


Figure S1. Individual-based rarefaction curves for bird assemblages recorded in each park. Each panel represents one pocket park, and curves show the expected increase in species richness as a function of the number of individuals sampled. The shape of the curves indicates variation among parks in both observed richness and sampling depth.

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708 **Table S1**. Functional traits used for bird species recorded in the study.

Trait Type	Trait	Description	Variable Type	Unit
Morphometric	Total Culmen Length		Continuous	millimeters
	Beak Height			millimeters
	Beak Width			millimeters
	Tarsus Length			millimeters
	Tail Length			millimeters
	Wing Length			millimeters
	Body Mass			grams
Life History	Habitat	Shrubland	Categorical	
		Human-modified		
		Forest		
		Marine		
		Grassland		
		Arboreal		
	Migration	Resident	Categorical	
		Migratory		
		Partially migratory		
	Trophic Niche	Granivore	Categorical	
		Omnivore		
		Invertivore		
		Nectarivore		
		Terrestrial herbivore		
	Primary Lifestyle	Terrestrial	Categorical	
		Sessorial		
		Generalist		
		Aerial		

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711 **Table S2.** Abundance of bird species recorded in the 15 pocket parks of La Serena.

Family	Species	Abundance	Trophic niche	Origin
Trochilidae	*Sephanoides sephaniodes*	191	Nectarivore	Native
	Rhodopis vesper	56	Nectarivore	Native
	Patagona gigas	1	Nectarivore	Native
Charadriidae	*Vanellus chilensis*	7	Omnivore	Native
Laridae	*Larus dominicanus*	48	Omnivore	Native
Columbidae	*Columba livia*	321	Granivore	Introduced
	Zenaida meloda	161	Granivore	Native
	Zenaida auriculata	290	Granivore	Native
	Columbina picui	18	Granivore	Native
Falconidae	*Milvago chimango*	11	Omnivore	Native
Cotingidae	*Phytotoma rara*	2	Terrestrial herbivore	Native
Emberizidae	*Zonotrichia capensis*	208	Omnivore	Native
Fringillidae	*Spinus barbatus*	2	Granívoro	Native
Furnariidae	*Leptasthenura aegithaloides*	5	Invertivore	Native
Hirundinidae	*Tachycineta meyeni*	6	Invertivore	Native
	Pygochelidon cyanoleuca	23	Invertivore	Native
Icteridae	*Curaeus curaeus*	8	Omnivore	Native
	Leistes loyca	3	Omnivore	Native
	Molothrus bonariensis	59	Invertivore	Native
Mimidae	*Mimus thenca*	54	Omnivore	Native
Passeridae	*Passer domesticus*	244	Granivore	Introduced
Thraupidae	*Phrygilus gayi*	10	Granivore	Native
	Diuca diuca	62	Granivore	Native
	Sicalis luteola	12	Granivore	Native
Troglodytidae	*Troglodytes aedon*	45	Invertivore	Native
Turdidae	*Turdus falcklandii*	329	Invertivore	Native
Tyrannidae	*Anairetes parulus*	24	Invertivore	Native

	Xolmis pyrope	1	Invertivore	Native
	Elaenia albiceps	3	Invertivore	Native

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