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# Urban Parks and Wetlands Support Complementary Bird Assemblages in Huai'an, Eastern China

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**Abstract**

Urban biodiversity planning depends on whether terrestrial and aquatic habitats retain the same species or make non-redundant contributions to the observed species pool. We surveyed 14 urban parks and four transects nested within two wetlands in Huai'an, eastern China, during two rounds. The audited counts comprised 2085 individual records and 58 species. Observed richness was higher on the sampled wetland transects (median 21.5 versus 16.0 species), but limited independent wetland replication precluded a general habitat-effect claim. Parks and wetlands each contained 43 species, with 28 shared and 15 recorded exclusively in each habitat class. Across 14 park–wetland pairs of pooled landscape units, mean Sørensen dissimilarity was 0.457, partitioned into turnover of 0.353 and nestedness-resultant dissimilarity of 0.104; turnover accounted for 77.3% of the summed dissimilarity. A classification of all 58 species using AVONET showed that 11 of the 15 wetland-only records had a wetland habitat affiliation, whereas eight park-only records were associated with forest or woodland and four with shrubland. These are global species-level traits, not evidence of local habitat specialization. Environmental correlations remained exploratory and did not survive within-family false-discovery-rate adjustment in the primary rank-based analyses. The results support observed compositional complementarity within the sampled landscape, while imperfect detection, two survey rounds, unmatched habitat area and only two wetland systems limit generalization. Conservation should retain both terrestrial green spaces and heterogeneous wetland habitats, with attention to habitat quality and hydrological structure.

**Keywords:** avian assemblage; beta diversity; habitat complementarity; urban biodiversity; urban green space; wetland; water-network city

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## 1. Introduction

Urban development changes the amount, structure, and spatial arrangement of habitat available to birds. Species that depend on particular nesting sites, foods, or hydrological conditions are often lost, while a smaller group of disturbance-tolerant species remains widespread [1,2]. This filtering can make urban bird assemblages increasingly similar even

where local species counts appear moderate [3]. At broader scales, regional history and the geographic species pool also constrain which birds are available to occupy a city [4]. Global comparisons consequently identify both local urban conditions and regional context as drivers of urban bird diversity [5]. A richness value alone therefore says little about whether a city retains ecologically different groups of birds.

Most urban bird studies have concentrated on terrestrial green space. A meta-analysis found that the area of habitat patches is a recurring predictor of biodiversity within cities [6]. Surveys of Manila's remaining green spaces also showed that bird abundance and diversity were unevenly distributed among sites [7]. In small managed green spaces, the surrounding matrix and disturbance can constrain the community [8]. Woody vegetation cover is associated with the richness and abundance of native birds [9], and park assemblages may respond to the coverage of particular microhabitats more strongly than to a general heterogeneity score [10]. These results have practical value for park design, but they do not show whether terrestrial green spaces can represent birds associated with open water and wetland margins.

Studies of aquatic habitats point to a different set of controls. Wetland area and internal habitat heterogeneity affect the richness and abundance of wetland birds [11]. Land-cover composition along an urban–rural gradient also changes wetland bird diversity [12]. Across a large city, small ponds collectively supported high bird diversity because sites differed in habitat structure and management [13]. Wetland bird diversity reflects the range of ecological functions provided by open water, shallow feeding areas, emergent vegetation, and adjacent terrestrial cover [14]. A study in Xiaohongxiang Wetland further showed that taxonomic, functional, and phylogenetic diversity capture different aspects of the bird assemblage [15]. The implication is not simply that wetlands contain more species. Aquatic habitats may add species that are absent from terrestrial parks, while parks may retain woodland and edge species that are uncommon in open wetlands.

That distinction matters in Huai'an. The city contains parks, canals, ponds, lakes, farmland, and managed wetlands within the same lowland network. Studies in the Chengdu Plain found that agricultural composition and heterogeneity affected bird communities [16], and remote-sensing analysis has linked heterogeneous agricultural landscapes to higher bird diversity [17]. These findings are relevant because farmland forms part of the matrix around both parks and wetlands in Huai'an. Regional modeling also predicts uneven avian richness across rapidly urbanizing eastern China [18]. Planning Huai'an's habitats separately could therefore confuse local richness with regional representation. If a wetland and a park each contain 20 species, they are equivalent only if those species largely overlap. If each habitat contains a different set, removing either one reduces the regional pool. Sørensen similarity and species recorded exclusively in one sampled habitat class help evaluate that distinction [19,20].

Local environmental measurements can help explain why assemblages differ, but only when their scale matches the bird data. The classic habitat-complexity model relates bird diversity to vegetation structure because foliage layers provide different feeding and nesting opportunities [21]. More recent work shows that shrub cover can influence functional diversity [22]. Drought can alter waterbird habitat and community assembly [23,24], while urban heat and artificial light affect bird diversity or seasonal occurrence [25,26]. In our archive, however, paired urban parks sometimes shared one environmental coordinate, vegetation was represented by nine landscape units, and nighttime-light data were incomplete. These constraints make the environmental analysis useful for identifying possible mechanisms, but too weak for claiming a causal driver. Correlated predictors and non-independent samples can otherwise produce precise-looking but misleading coefficients [27].

We addressed three questions: whether observed local richness, abundance and diversity differed among the sampled park and wetland units; whether the two habitat classes made non-redundant contributions to the observed city-level species pool; and whether archived environmental measurements were associated with bird diversity. Our principal expectation was compositional complementarity, expressed as species replacement rather than a simple nested subset. We therefore partitioned incidence-based beta diversity and classified species by broad habitat affiliation and feeding guild. The contribution of this study is a quantified comparison within Huai'an's water-network landscape, rather than the general observation that wetlands contain waterbirds. Environmental associations were secondary, exploratory analyses because predictors were incompletely measured and spatially coarser than many bird sampling units.

## 2. Materials and Methods

### *2.1. Study Area and Sampling Hierarchy*

Huai'an is located in northern Jiangsu Province, eastern China (32 degrees 43 min-34 degrees 06 min N, 118 degrees 12 min-119 degrees 36 min E), in a transition between the northern subtropical and warm-temperate climatic zones. The landscape is predominantly low-lying and is crossed by rivers, canals, lakes, ponds, urban districts, parks, and agricultural land. The study included 18 bird sampling units (Figure 1; Table S1). Fourteen units were urban parks, with two parks in each of seven administrative areas: Hongze, Xuyi, Lianshui, Huaiyin, Huai'an, Jinhu, and Qingjiangpu. Four units were wetland transects, two in Baima Lake Wetland and two in Liushuwan Wetland.

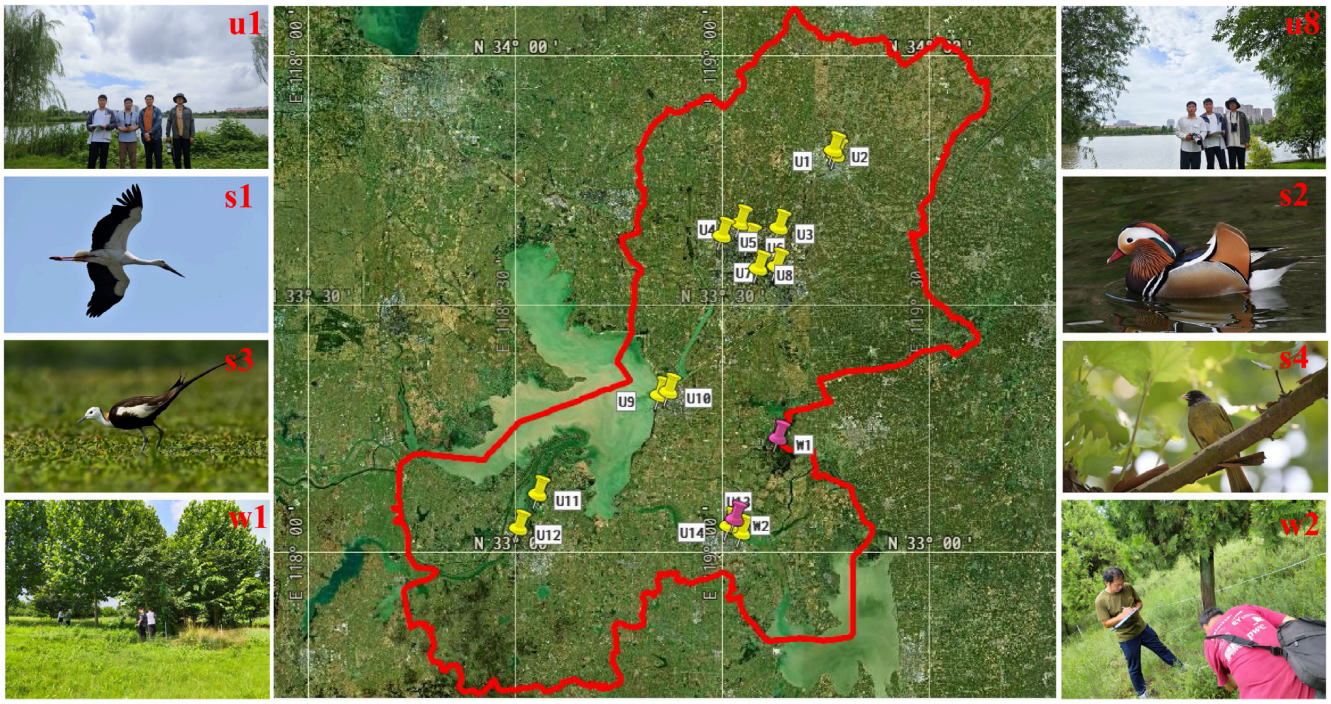


**Figure 1.** Study area, sampling design and site overview. Study area and archived sampling coordinates, retained from the submitted manuscript. The 18 bird sampling units form nine analytical landscape groups and occupy 11 recorded coordinate locations. Seven park pairs share archived coordinates; markers therefore do not establish exact park boundaries, transect geometry or sampled area. Unit IDs and coordinate-verification needs are listed in Table S1. u1–u14 identify park locations and w1 and w2 identify wetland systems according to the supplied figure. Bird panels are labeled s1 Oriental Stork, s2 Mandarin Duck, s3 Pheasant-tailed Jacana and s4 Collared Finchbill. The overview supplies contextual photographs but does not establish exact habitat boundaries, route lengths or record counts. Basemap provenance and the complete label-to-unit mapping remain to be documented.

A bird sampling unit was an individual park or wetland transect. A landscape unit was the union of the two parks in an administrative area or the two transects in a wetland, giving seven park landscape units and two wetland landscape units (Table 1). Park pairs were grouped for consistency with the archived landscape and environmental data; this grouping does not make a pair of parks equivalent in area or habitat structure to a wetland complex. Four wetland transects are subsamples of two systems. The 18 bird units were represented by only 11 distinct archived coordinate locations because seven park pairs shared coordinates. These coordinate groups are extraction locations, not additional biological replicates. Vegetation data came from nine representative survey locations and were matched to the corresponding archived bird unit, rather than to pooled landscape richness.

**Table 1.** Sampling and analysis levels.

| Level | Number | Definition and Use |
|---|---|---|
| Bird units | 18 | 14 parks; four transects within two wetlands. Descriptive alpha diversity and supplemental pairwise beta diversity. |
| Visits | 36 | Two rounds per unit, pooled for primary analysis; separate-round sensitivity analyses. |
| Landscape groups | 9 | Seven paired-park groups and two wetlands. Primary composition analysis and alpha sensitivity. |
| Coordinates | 11 | Seven shared park coordinates and four wetland coordinates. Environmental extraction and aggregation. |
| Vegetation locations | 9 | Representative parks or archived wetland transect-1 responses; three vegetation records averaged at each location. |

Baima Lake is characterized by a predominantly natural wetland setting, with open shallow water, seasonally exposed margins, emergent marsh vegetation and riparian woodland, with relatively limited built development around the surveyed area. Liushuwan is a restored urban wetland with landscaped water, engineered banks, planted woodland and waterside grassland, adjacent to roads and built areas and subject to recreation and maintenance. These are qualitative site descriptions, not measured naturalness, disturbance or habitat-quality scores. Table S1b separates this contextual information from the quantitative data.

### *2.2. Bird Surveys, Archived Counts and Survey Effort*

Bird surveys were conducted during two rounds, on 21–28 June and 22–29 August 2025. Each of the 18 bird sampling units had a 2 km line transect with a prescribed detection strip extending 50 m on either side. Surveys were undertaken between 05:30 and 09:00 or between 16:00 and 19:00, in clear weather with no or light wind. The four-person survey team received common training in bird identification, walking speed and recording procedures. Observer names are listed in Table S1a; the team size does not imply that all four observers attended every visit.

Transects were walked at 1.5–2.0 km $h^{-1}$ without retracing the route. Flocks and flying individuals were recorded at their first observed location and were not followed repeatedly. Calls, flight paths, spatial positions and movements were used to distinguish individuals; repeated detections of the same species in the same area were counted once when there was no evidence of different individuals. Distant birds passing overhead without using the sampled habitat were excluded. Each transect was surveyed at most once per day, with observation times varying among dates. Randomization or systematic rotation of site order

has not been established. These procedures reduce potential duplicate detections but do not identify unique individuals across separate visits.

Actual count durations, visit-specific observer assignments and dates, mapped route geometry, and separation between wetland transects remain incompletely documented (Table S1a). Equal prescribed lengths and widths do not establish equal effective surveyed area or detection probability in open water and vegetated parks. Walking speed has not been substituted for measured count duration. The two rounds describe early- and late-summer observations; they do not resolve species-specific breeding or migration status.

The re-examined source workbooks contain species-level counts for both rounds at all 18 units: 418 positive unit–round–species records. Counts summed across the two rounds reproduce every species-level pooled count (300 positive unit–species records) and the total of 2085. Pooled richness is the union of detections, and abundance is the sum of individual records across visits, not a count of uniquely marked individuals. A zero in a derived incidence matrix means not recorded during the represented visits, not verified ecological absence. Two rounds are insufficient to establish annual completeness; the current metadata also do not support a defensible detection-corrected model. Species-level counts were available for both rounds at all 18 units, allowing reconstruction of the unit–round–species matrix.

### *2.3. Environmental Measurements and Matching*

Vegetation was surveyed using 100 m × 100 m plots containing nested 20 m × 20 m tree, 5 m × 5 m shrub and 1 m × 1 m herb quadrats. The 10,000 $m^2$ area refers to a vegetation plot, not the entire wetland, park or bird-survey area. The archived analysis contains three vegetation measurement records at each of nine representative locations; those records were averaged within each location. The number and placement of nested quadrats contributing to each record and their spatial independence remain to be verified. Archived wetland responses match transect 1 in each wetland. Cuihu Garden richness was corrected from the vegetation table's 13 to the count-supported 11. Herbaceous measurements were available only for wetlands and were excluded from cross-habitat correlations. The timing falls between the bird visits but does not establish identical phenological conditions across June, July and August.

The environmental archive contains 19 bioclimatic variables, elevation, population density, nighttime lights and road density. Climate variables were obtained from WorldClim v2.1 at 30 arc-seconds (approximately 1 km) [28], representing the 1970–2000 climatic baseline rather than weather during the 2025 visits. Nighttime lights are identified as annual NPP/VIIRS composites supplied by the National Earth System Science Data Center at 500 m, population density as LandScan at approximately 1 km, and roads as the 1:250,000 national basic geographic database accessed through the National Geographic Information Resources Catalogue Service. The reported processing used Arcmap 10.6 kernel density for roads and resampling of raster outputs to a nominal 30 m grid. Product years and exact releases, the elevation source, kernel bandwidth and units, resampling/interpolation choices, and the complete coordinate reference system still require verification (Table S1c). A WGS84 reference alone does not identify a projected coordinate system in meters, and a 30 m output grid does not increase the information content of coarser source layers.

Seven park coordinate pairs duplicate environmental extraction locations. Primary environmental summaries therefore average corrected bird responses within 11 coordinate groups; the original 18-row analysis is retained as a sensitivity check. Nighttime lights are missing from five unit records, leaving eight coordinate groups after aggregation. Observed zeros and missing values are kept distinct. Qualitative wetland descriptions were supplied (Table S1b), but water-depth regime, shoreline naturalness, aquatic vegetation cover,

local noise, visitor activity and habitat heterogeneity were not measured as comparable quantitative predictors.

### 2.4. Alpha Diversity and Dependence-Aware Reporting

For each unit, pooled species counts were used to calculate richness $S$, total individual records N, Shannon diversity $H' = -\Sigma p_i \ln(p_i)$, and Pielou evenness $J' = \frac{H'}{ln(S)}$, where $p_i$ is the fraction of individual records belonging to species i [29,30]. Visits were not treated as independent habitat replicates. Means, medians, quartiles, ranges, and all unit-level values are reported.

Because four wetland transects represent only two wetlands, habitat comparisons are primarily descriptive. As a sensitivity analysis, we averaged unit-level alpha metrics within each of the nine landscape groups and enumerated all 36 allocations of two wetland labels to nine groups for a two-sided Mann–Whitney statistic. This test assumes exchangeability of the observational landscape groups and is not a randomized causal comparison. We report Cliff's delta, defined as the probability that a wetland value exceeds a park value minus the reverse probability, and Holm-adjusted $p$ values across the four alpha metrics. The original unrestricted 18-unit tests are supplied only as legacy benchmarks, not confirmatory habitat tests. With two wetland systems, we do not claim a reliable population-level habitat-effect confidence interval.

### 2.5. Species Overlap, Beta Partitioning and Ordination

Incidence matrices were constructed at three scales: pooled habitat species pools, 18 sampling units and nine landscape groups. For each pair, a is the number of shared species, and b and c are the numbers recorded only in the first or second member. We calculated Sørensen dissimilarity $\beta_{SOR} = \frac{b+c}{2a+b+c}$, turnover $\beta_{SIM} = \frac{min(b,c)}{a+min(b,c)}$, and nestedness-resultant dissimilarity $\beta_{SNE} = \beta_{SOR} - \beta_{SIM}$ [31]. These components describe composition and richness differences; they do not identify the causes of turnover. Pairwise results were summarized separately for park–park, wetland–wetland and park–wetland comparisons. The turnover share for a comparison class is $\Sigma\beta_{SIM}/\Sigma\beta_{SOR}$, not the mean of individual pairwise ratios. All distances, medians, quartiles and ranges are supplied in the data workbook. Repeated use of the same units makes pairwise distances non-independent, so no ordinary test treating the distances as independent observations was performed.

The nine-landscape analysis is primary, with the 18-unit results and separate-round analyses as sensitivity checks. There are 21 park–park, 14 park–wetland and only one wetland–wetland landscape pair. A two-dimensional nonmetric multidimensional scaling (NMDS) solution used Sørensen dissimilarity, 50 starting configurations, a maximum of 3000 iterations and seed 20260926; tied dissimilarities shared fitted disparities. Stress-1 was calculated as the square root of residual squared distances divided by squared ordination distances. The ordination and Shepard plot are descriptive [32–34]. We did not perform habitat-level PERMANOVA or interpret a visual separation as a significance test. Sample-based accumulation curves show exact expected observed richness over subsets of sampled units, not annual completeness or detection-corrected richness.

### 2.6. Ecological Classification and Exploratory Correlations

All 58 species were linked to AVONET version 7, using the AVONET2_eBird taxonomy table, to obtain primary habitat affiliation, trophic niche, trophic level, primary lifestyle, habitat density and global migration category [35]. There were 57 exact scientific-name matches and one documented genus synonym, *Spilopelia chinensis* to *Streptopelia chinensis*. Native database categories were retained. For one summary figure, Wetland and Riverine were grouped as aquatic/wetland, and Forest and Woodland as forest/woodland. Feeding guilds were the published trophic-niche categories rather than assignments inferred from

local detections. Counts and proportions were compared descriptively across 15 park-only records, 15 wetland-only records and 28 shared species. These are coarse global species traits, not measured local diets, demonstrated habitat specialization, disturbance sensitivity or local migratory status.

Environmental analyses used Spearman correlations, with Pearson results as sensitivity checks. For nine vegetation locations, Spearman $p$ values enumerate all 362,880 permutations; coordinate-level Spearman $p$ values use the usual asymptotic approximation and therefore warrant caution at this sample size. Benjamini–Hochberg q values were calculated separately within dataset, spatial scale and correlation method: 32 vegetation comparisons, 80 climate/elevation comparisons and 16 human-pressure/elevation comparisons. All results, including nonsignificant associations, are supplied. We did not select a multivariable driver model from these small, correlated datasets. The code, count audit, taxonomic match and source-to-unit mapping accompany the revision.

## 3. Results

### *3.1. Observed Species Pool and Local Diversity*

The two rounds contained 1224 and 861 individual records, respectively, totaling 2085 records of 58 species in 33 families and 12 orders. Passeriformes contributed 30 species (51.7%). The two rounds recorded 38 and 54 species; these numbers describe temporal observation differences and are not corrected for detection. Pooled park richness ranged from 8 to 21 species and wetland-transect richness from 17 to 26 (Table S2). Mean richness was 15.29 in parks and 21.50 on wetland transects; medians were 16.0 and 21.5, with interquartile ranges of 13.00–17.75 and 19.25–23.75.

Median abundance was 89.5 individual records in parks (interquartile range 60.25–134.25; range 29–209) and 154.0 on wetland transects (134.00–179.25; 92–237). Median Shannon diversity was 2.136 versus 2.288, and corrected median Pielou evenness was 0.803 versus 0.744 (Figure 2; Table S2). Landscape-mean richness was higher in the two wetlands than in most park groups, with Cliff's delta = 0.857, but the exact landscape-label sensitivity test gave $p = 0.083$ and Holm-adjusted $p = 0.333$. None of the four landscape-level alpha comparisons reached nominal $p < 0.05$. Full estimates and the legacy unit-level benchmarks are reported in Table S4.

### *3.2. Habitat Overlap and Beta-Diversity Partitioning*

Parks and wetlands each contained 43 species, with 28 shared and 15 recorded only in each habitat class. The combined observed pool was 58 species. For the two pooled habitat lists, $\beta_{SOR} = 0.3488$, $\beta_{SIM} = 0.3488$ and $\beta_{SNE} = 0$. This zero nestedness-resultant component follows from equal pooled richness and does not imply zero nestedness at individual sampling units.

Across the 14 park–wetland landscape pairs, mean $\beta_{SOR}$ was 0.4569, comprising mean $\beta_{SIM}$ of 0.3532 and mean $\beta_{SNE}$ of 0.1037. Turnover accounted for 77.3% of the summed dissimilarity. The corresponding shares were 81.3% for the 21 park–park landscape pairs and 92.1% for the single wetland–wetland pair (Table 2; Figure 3). Among 18 units, turnover accounted for 82.9% of park–wetland dissimilarity, 76.0% of park–park dissimilarity and 85.4% of wetland–wetland dissimilarity. The two rounds separately also retained a turnover-dominated contrast between pooled habitat lists: turnover shares were 95.7% and 92.4%, respectively (Table S7).

### *3.3. Ecological Composition of the Recorded Species*

AVONET classified 11 of the 15 wetland-only records (73.3%) as primarily associated with wetlands. The remaining four comprised two forest/woodland species, one shrubland

species and one species associated with human-modified habitats. Among the 15 park-only records, eight (53.3%) were classified as forest/woodland, four (26.7%) as shrubland, one as grassland and two as human-modified; none was assigned a primary wetland or riverine habitat. Seven of the 28 shared species had wetland or riverine affiliations (Figure 4; Table S3).

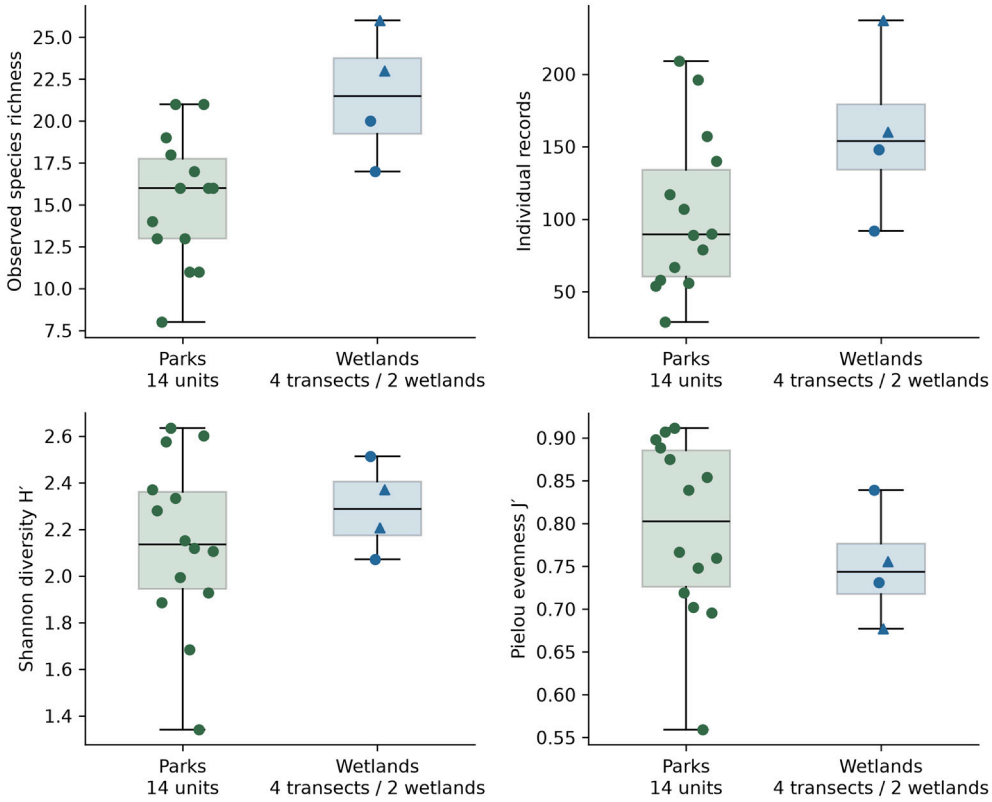


**Figure 2.** Recomputed diversity of the 18 sampling units. Boxes span quartiles, center lines are medians, and whiskers show observed minima and maxima. Each point is a park or transect. Triangles distinguish Baima Lake transects from Liushuwan circles; four wetland points do not represent four independent wetlands. Evenness uses $J' = \frac{H'}{ln(S)}$.

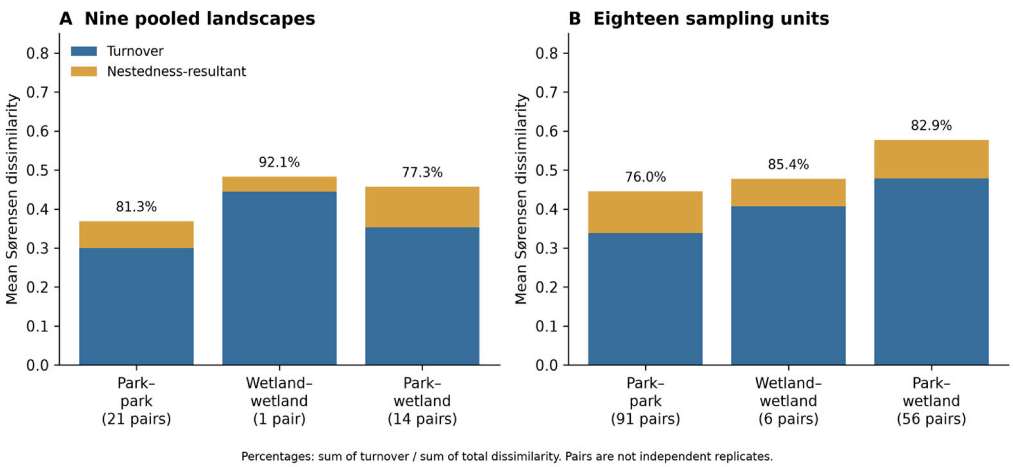


**Figure 3.** Turnover and nestedness-resultant components of mean pairwise Sørensen dissimilarity. Panel (**A**) uses nine landscape groups; Panel (**B**) uses 18 sampling units. Percentages are ratios of summed components. The single wetland–wetland landscape comparison and nested wetland transects cannot support independent between-group inference.

**Table 2.** Mean beta-diversity components among nine landscape groups. Pair counts are descriptive distances, not independent sample sizes.

| Comparison | Pairs | βSOR | βSIM | βSNE | Turnover Share |
|---|---|---|---|---|---|
| Park–park | 21 | 0.3686 | 0.2995 | 0.0690 | 81.3% |
| Park–wetland | 14 | 0.4569 | 0.3532 | 0.1037 | 77.3% |
| Wetland–wetland | 1 | 0.4828 | 0.4444 | 0.0383 | 92.1% |

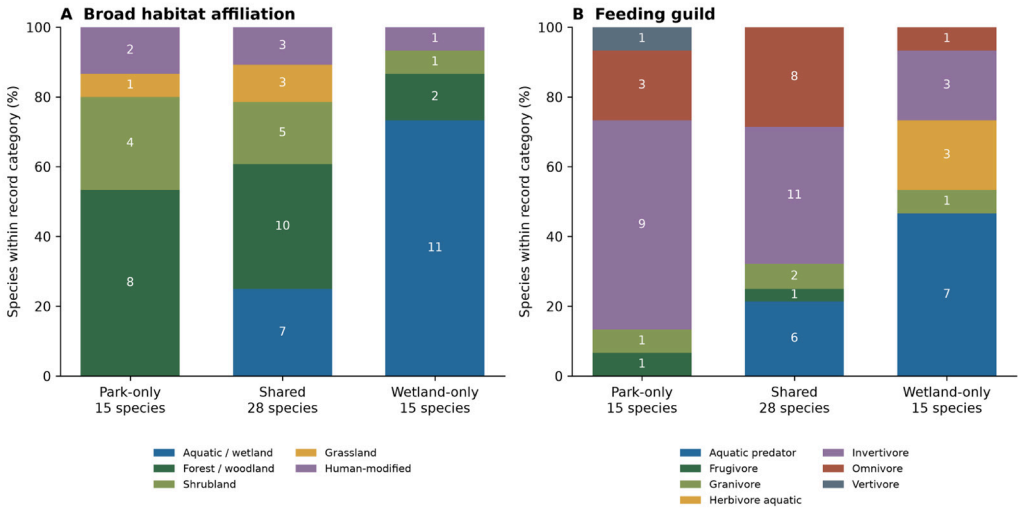


**Figure 4.** Global habitat affiliations and feeding guilds from AVONET for all 58 species, grouped by observed occurrence in this survey. Numbers within bars are species counts; bar heights represent proportions within the 15 park-only, 28 shared and 15 wetland-only record categories. "Only" refers to detections in these two rounds, not ecological exclusivity.

Feeding guilds also differed descriptively. Wetland-only records included seven aquatic predators (46.7%) and three aquatic herbivores (20.0%), together with three invertivores, one granivore and one omnivore. Park-only records included nine invertivores (60.0%), three omnivores, one frugivore, one granivore and one vertebrate predator. The shared pool included six aquatic predators and 11 invertivores. Global migration categories classified five wetland-only records and ten park-only records as migratory; this does not establish their local seasonal status or explain why they were recorded in only one habitat class.

### *3.4. Community Similarity and Ordination*

Pairwise Sørensen similarity among the nine landscape groups ranged from 0.2800 (Baima Lake–Xuyi) to 0.7907 (Huai'an–Jinhu). Jinhu and Liushuwan Wetland shared a similarity of 0.7600. The rebuilt matrix is provided with a conventional diagonal of one (Figure S2). The minimum-stress two-dimensional NMDS solution had stress-1 = 0.0961; 12 of the 50 starts converged within 0.0001 of this minimum. Every landscape group is displayed, together with a Shepard plot (Figure S3). These outputs describe compositional variation without testing a general habitat separation.

### *3.5. Corrected Exploratory Environmental Associations*

At the nine vegetation locations, shrub basal diameter was negatively correlated with Shannon diversity (Spearman $\rho = -0.833$, exact $p = 0.0083$), but this association did not survive adjustment across the 32 vegetation comparisons (q = 0.265). With corrected Pielou evenness, the association was weaker ($\rho = -0.650$, $p = 0.0666$, q = 0.545). No primary

vegetation rank correlation had q < 0.05. These analyses use the representative bird units linked to vegetation records, not pooled responses from two parks or transects.

Across 11 coordinate groups, annual precipitation correlated negatively with corrected evenness (Spearman ρ = −0.688, nominal $p$ = 0.0193, q = 0.386; Pearson r = −0.793, $p$ = 0.0036, q = 0.129). Coldest-quarter precipitation correlated positively with mean richness (Spearman ρ = 0.590, $p$ = 0.0558, q = 0.425; Pearson r = 0.737, $p$ = 0.0097, q = 0.156). Thus, the earlier description of an absent rank association for precipitation and evenness is not retained, but neither relationship survives its within-family adjustment. Primary human-pressure/elevation rank correlations also had q > 0.05. The complete results and missingness are reported in the workbook; selected corrected relationships are illustrated in Figure S4.

## 4. Discussion

### *4.1. Evidence and Limits of Compositional Complementarity*

The most direct evidence of complementarity is the addition of species identities, not a general advantage of wetlands in local richness. The two habitat classes each contained 43 species, yet their union contained 58. Relative to that observed union, either list alone omitted 15 species (25.9%). This is a description of the recorded lists, not a forecast of species loss following habitat removal. The beta-diversity partition strengthens the compositional interpretation: turnover accounts for most park–wetland dissimilarity at both landscape and sampling-unit scales. This indicates that the observed park–wetland contrast was dominated by species replacement rather than by simple richness nestedness. The pooled-list result is consistent with this pattern but is mathematically constrained by equal richness. Spatial variation within habitat classes, and the relatively high Jinhu–Liushuwan similarity, caution against treating habitat labels as complete explanations.

Observed wetland-transect richness was higher, but the nesting and small number of systems weaken habitat-wide inference. After aggregation to nine landscape means, the richness contrast did not reach the conventional significance threshold. Averaging avoids treating four transects as four wetlands, but cannot resolve the confounding of habitat type with area, route placement, habitat heterogeneity or particular wetland identity. Counts can also be influenced by flocking and repeat observations of unmarked individuals. The design therefore supports a local compositional comparison and motivates a larger, balanced study; it does not establish that wetlands generally outperform parks.

### *4.2. Ecological Interpretation from Independently Sourced Traits*

The trait table adds structured evidence to the ecological interpretation. Wetland-only detections contain a substantial aquatic component, including predators and herbivores associated with aquatic resources, whereas park-only detections contain many forest, woodland and shrubland species and terrestrial invertivores. This is consistent with complementary resource and vegetation settings [14,36]. However, some wetland-only detections have terrestrial primary affiliations, and several aquatic species were recorded in both habitat classes. Neither the observed lists nor coarse global categories justify calling all habitat-exclusive detections specialists. The AVONET classifications describe typical species ecology across their ranges; local foraging behavior and habitat use were not measured.

We therefore do not equate compositional complementarity with a measured increase in functional diversity. Taxonomic and functional responses can diverge in urban landscapes [37–39]. Broad guilds summarize the kinds of species recorded and provide hypotheses about food and habitat resources, but do not estimate functional richness, ecosystem functioning or dependence on specific local microhabitats. The global migration

assignments also caution against interpreting short-term non-detections as permanent absence: both habitat-exclusive lists include migratory taxa.

### *4.3. Environmental Associations and Scale Mismatch*

Recalculating evenness changes both its absolute values and its environmental correlations, illustrating why the source count audit matters. The remaining negative shrub-diameter–Shannon correlation is exploratory and is not robust to adjustment for the set of examined vegetation relationships. Basal diameter measures stem size, not shrub cover, native plant diversity or foliage-height complexity; it is therefore not directly comparable to studies of shrub cover [22]. No removal or planting recommendation follows from this coefficient. The nine representative vegetation locations also cannot explain the differences between two parks sharing one environmental record.

Likewise, long-term climatic layers are indirect proxies for the water conditions encountered during two bird visits. Water depth, exposure of feeding margins and emergent vegetation would be more direct predictors [23,24,40]. Population, road and night-light layers are not measurements of local disturbance along survey routes, and missing night-light records reduce information further. Studies show context-dependent responses to urbanization, heat, activity and light [25,26,41–43]. Our exploratory results do not demonstrate that those processes are unimportant. Small samples, correlated predictors, temporal mismatch and omitted local variables constrain what can be detected [27,33,34].

### *4.4. Wetland Quality, Habitat Heterogeneity and Management*

The presence of a wetland label is not sufficient to establish conservation value. Natural or semi-natural shorelines, shallow-water gradients, reedbeds and other emergent vegetation, hydrological variability and adjacent terrestrial cover can provide distinct ecological opportunities. Studies of wetland area, heterogeneity and pond management support the relevance of habitat structure [11–14]. Evidence from regulated river meanders also shows that artificial habitat elements may not compensate for losses associated with habitat alteration [44]. These findings motivate closer attention to wetland quality; they do not demonstrate which unmeasured feature caused the Huai'an contrast.

For the sampled landscape, a reasonable conservation response is to retain existing wetland habitat together with mature trees, understory, and other terrestrial elements in parks. Diverse wetlands should not be assumed replaceable by ornamental open water. Where management is undertaken, shoreline hardening, uniform planting, intensive mowing, lighting and recreational disturbance should be evaluated for their effects on the range of habitats available. Connectivity among parks, canals, ponds, riparian vegetation and surrounding land may support movement [45], but connectivity does not justify removing habitat cores. Because hydrology, shore structure, succession, management intensity and disturbance were not quantified consistently here, these are literature-informed priorities and testable hypotheses rather than site-specific prescriptions or demonstrated causal thresholds.

The supplied site descriptions make this distinction relevant locally: Baima Lake includes shallow-water and emergent-marsh elements, whereas Liushuwan includes restored and managed urban wetland elements. These differences motivate documenting water regime, shoreline modification and recreational pressure in subsequent surveys. They cannot establish that one wetland has greater ecological integrity or explain the observed compositional contrast without comparable measurements.

### *4.5. Sampling Limitations and Future Work*

Only two wetland systems were sampled, with unequal replication relative to 14 parks. Wetland transects may not be independent of one another, and park pairs are also grouped

spatially. Two rounds cannot describe annual turnover or establish stable habitat restriction. Detection can differ between open water and densely vegetated parks because visibility, sound transmission, background noise and flocking differ. A species recorded only in one habitat class is therefore a survey-exclusive detection, not proof of true absence from the other. Sample accumulation over the observed units cannot establish detection completeness or remove differences in the effective surveyed area.

The recovered species-level round counts improve reproducibility and permit temporal sensitivity analyses. Available survey metadata include survey-date windows, a common 2 km transect length, prescribed detection width, weather conditions, observer training and procedures to reduce duplicate counting. Actual count durations, per-visit observer/date assignments and route geometry remain incomplete. Seven park pairs share archived coordinates. Vegetation was measured between the two bird rounds, while climate layers represent a historical baseline; matching the grids does not remove temporal or spatial support differences. Qualitative wetland descriptions do not replace systematic quality measurements. Observational comparisons cannot distinguish habitat causation from site selection or landscape context. Future work should add independent wetlands and seasonal repeat visits with complete effort and detection records, plus comparable habitat, hydrological and disturbance measurements.

## 5. Conclusions

Within the sampled Huai'an landscape, urban parks and two wetland systems made non-redundant contributions to the observed bird assemblage. Each habitat class contributed 15 species not detected in the other, and species turnover accounted for 77.3% of park–wetland dissimilarity at the landscape scale. Independent global trait classifications revealed an aquatic component among wetland-only records and a forest, woodland and shrubland component among park-only records. These results support retaining both terrestrial and aquatic habitat diversity. They do not establish a general richness advantage of wetlands or a causal environmental mechanism. More independent wetland systems, verified survey effort, seasonal coverage and direct measurements of habitat quality are needed to test the generality and causes of the observed complementarity.

**Supplementary Materials:** The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/d18100604/s1, Table S1: Sampling units and archived coordinates (including Table S1a, common bird and vegetation survey information; Table S1b, qualitative wetland descriptions; and Table S1c, environmental data sources); Table S2: Pooled observed richness, individual records, Shannon diversity and corrected Pielou evenness for the 18 sampling units; Table S3: AVONET primary habitat and trophic-niche categories of the 58 recorded species; Table S4: Alpha-diversity sensitivity tests based on landscape-group means; Table S5: Source audit and remaining information requirements; Table S6: Guide to the accompanying Supplementary Data workbook; Table S7: Partition of the pooled park and wetland species lists (pooled data, Round 1 and Round 2); Table S8: Mean pairwise beta-diversity components at the sampling-unit scale; Figure S1: Sample-based species accumulation curves (interpolated expected observed richness) for park units and wetland transects; Figure S2: Sørensen similarity among the nine landscape groups; Figure S3: Nonmetric multidimensional scaling (NMDS) ordination and Shepard plot of the nine landscape groups; Figure S4: Selected relationships between corrected bird diversity metrics and environmental variables. The Supplementary Data workbook contains the audited unit–round–species records, incidence matrices for the 18 sampling units and nine landscape groups, vegetation and environmental data, pairwise beta-diversity results, statistical outputs, and documented data limitations; its worksheets are described in Table S6.

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**Data Availability Statement:** The data supporting the findings of this study are available from the corresponding authors upon reasonable request. Relevant species-level survey records, site metadata, vegetation measurements, environmental data, and analysis outputs can be provided where appropriate.

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## References

1. McKinney, M.L. Urbanization as a major cause of biotic homogenization. *Biol. Conserv.* **2006**, *127*, 247–260. [CrossRef]
2. Chace, J.F.; Walsh, J.J. Urban effects on native avifauna: A review. *Landsc. Urban Plan.* **2006**, *74*, 46–69. [CrossRef]
3. 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.; et al. A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proc. R. Soc. B* **2014**, *281*, 20133330. [CrossRef] [PubMed]
4. Jetz, W.; Thomas, G.H.; Joy, J.B.; Hartmann, K.; Mooers, A.O. The global diversity of birds in space and time. *Nature* **2012**, *491*, 444–448. [CrossRef] [PubMed]
5. Hughes, A.C.; Orr, M.C.; Lei, F.; Yang, Q.; Qiao, H. Understanding drivers of global urban bird diversity. *Glob. Environ. Change* **2022**, *76*, 102588. [CrossRef]
6. Beninde, J.; Veith, M.; Hochkirch, A. Biodiversity in cities needs space: A meta-analysis of factors determining intra-urban biodiversity variation. *Ecol. Lett.* **2015**, *18*, 581–592. [CrossRef] [PubMed]
7. Vallejo, B.M., Jr.; Aloy, A.B.; Ong, P.S. The distribution, abundance and diversity of birds in Manila's last greenspaces. *Landsc. Urban Plan.* **2009**, *89*, 75–85. [CrossRef]
8. Liordos, V.; Jokimäki, J.; Kaisanlahti-Jokimäki, M.L.; Valsamidis, E.; Kontsiotis, V.J. Patch, matrix and disturbance variables negatively influence bird community structure in small-sized managed green spaces located in urban core areas. *Sci. Total Environ.* **2021**, *801*, 149617. [CrossRef] [PubMed]
9. Villaseñor, N.R.; Escobar, M.A.H.; Hernández, H.J. Can aggregated patterns of urban woody vegetation cover promote greater species diversity, richness and abundance of native birds? *Urban For. Urban Green.* **2021**, *61*, 127102. [CrossRef]
10. Chan, M.; Tsang, T.P.N.; Dingle, C.; Early, R.; Sorte, C.J.B.; Bonebrake, T.C. Microhabitat coverage influences avian species composition more than habitat heterogeneity in Hong Kong urban parks. *Urban For. Urban Green.* **2024**, *101*, 128519. [CrossRef]
11. Elliott, L.H.; Igl, L.D.; Johnson, D.H. The relative importance of wetland area versus habitat heterogeneity for promoting species richness and abundance of wetland birds in the Prairie Pothole Region, USA. *Condor Ornithol. Appl.* **2020**, *122*, duz060. [CrossRef]
12. Mao, Q.; Liao, C.; Wu, Z.; Guan, W.; Yang, W.; Tang, Y.; Wu, G. Effects of land-cover pattern along urban-rural gradient on bird diversity in wetlands. *Diversity* **2019**, *11*, 86. [CrossRef]
13. Rawal, P.; Kittur, S.; Chatakonda, M.K.; Sundar, K.S.G. Capital ponds: Site-level habitat heterogeneity and management interventions at ponds regulate high landscape-scale bird diversity across a mega-city. *Biol. Conserv.* **2021**, *260*, 109215. [CrossRef]
14. Qiu, J.; Zhang, Y.; Ma, J. Wetland habitats supporting waterbird diversity: Conservation perspective on biodiversity-ecosystem functioning relationship. *J. Environ. Manag.* **2024**, *357*, 120663. [CrossRef] [PubMed]
15. Li, B.; Zhang, S.; Guo, J.; Ma, S.; Zhang, W. The taxonomic, functional and phylogenetic diversity of birds in Xiaohongxiang Wetland, southwest China. *Biodivers. Data J.* **2024**, *12*, e136248. [CrossRef] [PubMed]
16. Liao, J.; Liao, T.; He, X.; Zhang, T.; Li, D.; Luo, X.; Wu, Y.; Ran, J. The effects of agricultural landscape composition and heterogeneity on bird diversity and community structure in the Chengdu Plain, China. *Glob. Ecol. Conserv.* **2020**, *24*, e01191. [CrossRef]

17. Anderle, M.; Brambilla, M.; Hilpold, A.; Matabishi, J.G.; Paniccia, C.; Rocchini, D.; Rossin, J.; Tasser, E.; Torresani, M.; Tappeiner, U.; et al. Habitat heterogeneity promotes bird diversity in agricultural landscapes: Insights from remote sensing data. *Basic Appl. Ecol.* **2023**, *70*, 38–49. [CrossRef]
18. Chen, W.; Wang, X.; Cai, Y.; Huang, X.; Li, P.; Liu, W.; Chang, Q.; Hu, C. Potential distribution patterns and species richness of avifauna in rapidly urbanizing East China. *Ecol. Evol.* **2024**, *14*, e11515. [CrossRef] [PubMed]
19. Sørensen, T. A method of establishing groups of equal amplitude in plant sociology based on similarity of species content and its application to analyses of the vegetation on Danish commons. *Biol. Skr.* **1948**, *5*, 1–34.
20. Jaccard, P. Étude comparative de la distribution florale dans une portion des Alpes et du Jura. *Bull. Société Vaudoise Sci. Nat.* **1901**, *37*, 547–579. [CrossRef]
21. MacArthur, R.H.; MacArthur, J.W. On bird species diversity. *Ecology* **1961**, *42*, 594–598. [CrossRef]
22. Melo, M.A.; Piratelli, A.J. Increase in size and shrub cover improves bird functional diversity in Neotropical urban green spaces. *Austral Ecol.* **2023**, *48*, 440–460. [CrossRef]
23. Gao, X.; Liang, J.; Zhu, Z.; Li, W.; Lu, L.; Qiu, X.; Li, S.; Tang, N.; Li, X. Unraveling the impact of drought on waterbird community assembly and conservation strategies. *J. Environ. Manag.* **2025**, *373*, 123685. [CrossRef] [PubMed]
24. Gao, X.; Liang, J.; Zhu, Z.; Li, W.; Lu, L.; Li, X.; Li, S.; Tang, N.; Li, X. Drought-induced changes in hydrological and phenological interactions modulate waterbird habitats dynamics. *J. Hydrol.* **2023**, *626*, 130228. [CrossRef]
25. Cai, Z.; La Sorte, F.A.; Chen, Y.; Wu, J. The surface urban heat island effect decreases bird diversity in Chinese cities. *Sci. Total Environ.* **2023**, *902*, 166200. [CrossRef] [PubMed]
26. La Sorte, F.A.; Horton, K.G. Seasonal variation in the effects of artificial light at night on the occurrence of nocturnally migrating birds in urban areas. *Environ. Pollut.* **2021**, *270*, 116085. [CrossRef] [PubMed]
27. Dormann, C.F.; Elith, J.; Bacher, S.; Buchmann, C.; Carl, G.; Carré, G.; Marquéz, J.R.G.; Gruber, B.; Lafourcade, B.; Leitão, P.J.; et al. Collinearity: A review of methods to deal with it and a simulation study evaluating their performance. *Ecography* **2013**, *36*, 27–46. [CrossRef]
28. Fick, S.E.; Hijmans, R.J. WorldClim 2: New 1-km spatial resolution climate surfaces for global land areas. *Int. J. Climatol.* **2017**, *37*, 4302–4315. [CrossRef]
29. Shannon, C.E.; Weaver, W. *The Mathematical Theory of Communication*; University of Illinois Press: Urbana, IL, USA, 1949.
30. Pielou, E.C. *Ecological Diversity*; Wiley: New York, NY, USA, 1975.
31. Baselga, A. Partitioning the turnover and nestedness components of beta diversity. *Glob. Ecol. Biogeogr.* **2010**, *19*, 134–143. [CrossRef]
32. Clarke, K.R. Non-parametric multivariate analyses of changes in community structure. *Aust. J. Ecol.* **1993**, *18*, 117–143. [CrossRef]
33. Legendre, P.; Legendre, L. *Numerical Ecology*, 3rd ed.; Elsevier: Amsterdam, The Netherlands, 2012.
34. Borcard, D.; Gillet, F.; Legendre, P. *Numerical Ecology with R*, 2nd ed.; Springer: Cham, Switzerland, 2018. [CrossRef]
35. Tobias, J.A.; Sheard, C.; Pigot, A.L.; Devenish, A.J.M.; Yang, J.; Sayol, F.; Neate-Clegg, M.H.C.; Alioravainen, N.; Weeks, T.L.; Barber, R.A.; et al. AVONET: Morphological, ecological and geographical data for all birds. *Ecol. Lett.* **2022**, *25*, 581–597. [CrossRef] [PubMed]
36. Machar, I.; Šimek, P.; Schlossárek, M.; Pechanec, V.; Petrovic, F.; Brus, J.; Špinlerová, Z.; Seják, J. Comparison of bird diversity between temperate floodplain forests and urban parks. *Urban For. Urban Green.* **2022**, *67*, 127427. [CrossRef]
37. Morelli, F.; Benedetti, Y.; Ibáñez-Álamo, J.D.; Tryjanowski, P.; Jokimäki, J.; Kaisanlahti-Jokimäki, M.-L.; Suhonen, J.; Díaz, M.; Møller, A.P.; Moravec, D.; et al. Effects of urbanization on taxonomic, functional and phylogenetic avian diversity in Europe. *Sci. Total Environ.* **2021**, *795*, 148874. [CrossRef] [PubMed]
38. Mbiba, M.; Mazhude, C.; Fabricius, C.; Fritz, H.; Muvengwi, J. Bird species assemblages differ, while functional richness is maintained across an urban landscape. *Landsc. Urban Plan.* **2021**, *212*, 104094. [CrossRef]
39. Petersen, T.K.; Speed, J.D.M.; Grotan, V.; Froyen, Y.K.; Austrheim, G. Urbanisation and land-cover change affect functional, but not compositional turnover of bird communities. *Urban Ecosyst.* **2022**, *25*, 1679–1698. [CrossRef]
40. Li, X.; Anderson, C.J.; Wang, Y.; Lei, G. Waterbird diversity and abundance in response to variations in climate in the Liaohe Estuary, China. *Ecol. Indic.* **2021**, *132*, 108286. [CrossRef]
41. Madhok, R.; Gulati, S. Ruling the roost: Avian species reclaim urban habitat during India's COVID-19 lockdown. *Biol. Conserv.* **2022**, *271*, 109597. [CrossRef] [PubMed]
42. Grunst, A.S.; Grunst, M.L.; Raap, T.; Pinxten, R.; Eens, M. Anthropogenic noise and light pollution additively affect sleep behaviour in free-living birds in sex- and season-dependent fashions. *Environ. Pollut.* **2023**, *316*, 120426. [CrossRef] [PubMed]
43. McNaughton, E.J.; Beggs, J.R.; Gaston, K.J.; Jones, D.N.; Stanley, M.C. Retrofitting streetlights with LEDs has limited impacts on urban wildlife. *Biol. Conserv.* **2021**, *254*, 108944. [CrossRef]

44. Šálek, M.; Svobodová, J.; Zasadil, P. Loss of diversity in bird communities after regulation of riverine meanders: How strong is the compensatory effect of mature growth on fishpond dams? *Acta Ornithol.* **2007**, *42*, 89–97. [CrossRef]
45. Curzel, F.E.; Bellocq, M.I.; Leveau, L.M. Local and landscape features of wooded streets influenced bird taxonomic and functional diversity. *Urban For. Urban Green.* **2021**, *66*, 127369. [CrossRef]

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