Factors responsible for forest and water bird distributions in rivers and lakes along an urban gradient in Beijing

Shilin Xie [a,b], Xiaoke Wang [a,b,c], Yufen Ren [a,b,c], Zhimin Su [a,b,c], Yuebo Su [a,b], Siqi Wang [a,b,c], Weiqi Zhou [a,b,c], Fei Lu [a,b,c], Yuguo Qian [a,b], Cheng Gong [d,a,b], Binbin Huang [a,b], Zhiyun Ouyang [a,b,*]

[a] State Key Laboratory of Urban and Regional Ecology, Research Center for Eco-environmental Sciences, Chinese Academy of Sciences, Beijing 100085, China
[b] University of Chinese Academy of Sciences, Beijing 100049, China
[c] Beijing Urban Ecosystem Research Station, Research Center for Eco-environmental Sciences, Chinese Academy of Sciences, Beijing 100085, China
[d] School of Life Science, University of Science and Technology of China, Hefei 230026, China

HIGHLIGHTS

- Few bird-environment relationship study simultaneously focused on both forest and water birds.
- Food and available habitat determine avian community structure along an urban intensity continuum.
- 50% of artificial surface within habitats' neighborhood is a threshold for the effects of urbanization on avian diversity.
- Forest and water birds showed similar urbanization gradient distribution pattern but different in significance level.
- Natural shoreline devastation is a key process through which urbanization exerts its detrimental effects on water birds.

GRAPHICAL ABSTRACT
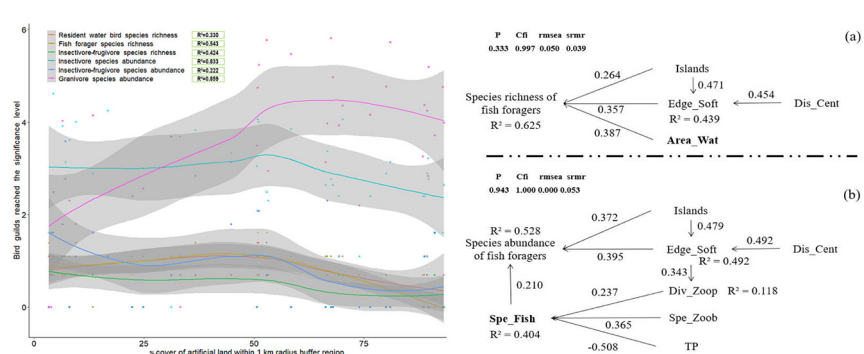
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ABSTRACT

Urban rivers and lakes, in combination with nearby green spaces, provide important habitat for urban birds, but few urban studies have focused on forest and water birds simultaneously along an urban intensity gradient. In this study, we randomly chose 39 rivers and lakes along an urban gradient of Beijing to examine bird community parameters in relation to aquatic and terrestrial habitat conditions, aquatic life data, and water quality data. We selected models with the AICc (corrected Akaike information criterion) method, bivariate linear or generalized linear regressions, and structural equation modeling to determine distribution patterns of avian communities along an urban gradient and bird-environment relationships. We found that both forest and water bird species and individuals peaked at intermediate urbanization intensities, especially for abundance of both forest and water bird and water bird species richness and abundance. We suggest that the differences in the strength of response to urbanization and the similarities in the gradient distribution pattern between forest and water birds should receive more attention in future urbanization gradient studies. Significant correlation ship between species richness of resident water birds, fish foragers, and insectivore-frugivores, abundance of insectivores, insectivore-frugivores (negative), and granivores (positive) and impervious surface proportion within 1-km radius buffer of sampled sites became more evident after coverage of artificial surfaces exceeded a 50% threshold. Regressions showed that distance from the urban center, number of islands in waterbody, and proportion of gross or unarmored shoreline length were significantly and positively related to species richness and abundance

* Corresponding author at: State Key Laboratory of Urban and Regional Ecology, Research Center for Eco-environmental Sciences, Chinese Academy of Sciences, Beijing 100049, China.
E-mail address: zyouyang@rcees.ac.cn (Z. Ouyang).

of both forest and water birds. The availability of unarmored shoreline is a critical pathway through which urbanization detrimentally impacts avian diversity. Our results demonstrate how the urban intensity gradient affects the relative availability of food resources and habitat, which could provide practical applications for urban landscape planning and avian biodiversity conservation in urban areas.

1. Introduction

By 2019, 55% of the world's population was residing in urban areas, with this number being projected to be 68% by 2050 (The United Nations, 2018). Rapid global population growth has long been and will continue to be accompanied by the expansion of urban areas (Seto et al., 2012), much of which will overlap with areas of biodiversity hot spots (Luck, 2010; Wittemyer et al., 2008). Land cover change during urbanization eliminate native habitats (Grimm et al., 2008), making urbanization the main driver of loss in biodiversity (Frishkoff et al., 2016). The effects of urbanization on biodiversity, especially avian biodiversity, have received widespread attention, and it has been demonstrated that urbanization simultaneously drives losses in taxonomic, functional, and phylogenetic diversity of bird communities (Alexander et al., 2019; Evans et al., 2018; Palacio et al., 2018). Gradients in the distribution of human alterations and landscape configurations in urban areas makes them ideal for studying variations in avian adaptation to land cover changes (Marzluff, 2016). However, very little information on avian habitat use is available for Beijing, the capital of China.

Some gradient studies suggested that avian biodiversity peaks at sites of moderate habitat alteration, known as the intermediate disturbance hypothesis (sensu Connell, 1978; Marzluff, 2005), whereas other studies have revealed a monotonic decline in species richness with increasing urbanization and an opposite relationship with abundance (Chace and Walsh, 2006). However, a recent meta-analysis of gradient studies suggested that avian diversity lose by different speed within different urbanization stage, and bird species richness declines as urbanization intensity increases while bird abundances peak in areas of intermediate urbanization (Batary et al., 2018). Other studies implied that the distribution of birds along urban land use gradients vary by guild (Rullman and Marzluff, 2014; Veech, 2006), which is especially evident for water birds (Mckinney et al., 2011; Murray et al., 2013).

A notable number of studies at various temporal and spatial scales have been conducted to determine major factors influencing urban bird distribution and/or community structure. Generally, local (habitat-scale) factors, such as potable water source (Zivanovic and Luck, 2016), shrub cover, and tree species richness, are more important in determining avian community structure than are landscape configurations (Beninde et al., 2015). In contrast, studies all around the world have demonstrated the critical influence of habitat area on avian species richness (Asia: Chang and Lee, 2016; Europe: Dale, 2018; America: Meixler et al., 2019; Oceanica: Watson et al., 2005), while Souza et al. (2019) suggested that the amount of impervious surface could be the key driver of city-scale species distribution. In addition, Robinson et al. (2018) demonstrated that bird occupancy rates are influenced by the size of nearby habitat patches and distances to them. However, urban bird studies have been traditionally focused on vegetated patches scattered throughout urban areas (thus, they mainly focused on forest birds; Pickett et al., 2011), very few studies have examined riparian or lacustrine habitats along an urbanization gradient. One of the exceptional studies conducted in Patagonia suggested that land-use impact and water quality conditions could be well reflected by macroinvertebrates (Miserendino et al., 2011), but the study failed to reveal the species-environment relationship for biotas (fish and birds) of high dispersal capability. In addition, terrestrial forest and aquatic birds have seldom been surveyed simultaneously and independently, even though the two types focus on very different habitat types. An urban forest

study in Sendai (Japan) by Imai and Nakashizuka (2010) showed that water-associated habitats are important for shaping avian community structure of both forest and water birds, although the study did not specifically differentiate between the two types. Therefore, we recommend that forest birds and water birds be separated in analyses of bird-environment relationship to clarify the differences and connections in those two sets of relationships.

To date, a series of studies have been conducted in natural, semi-natural, and constructed wetlands that identify key habitat factors affecting water bird distributions. Suri et al. (2017) suggested that the occurrence of river in the suburban area of Cape Town (South Africa) drives the appearance of a diverse water bird functional guilds cluster. Similarly, a gradient study along Salt River (which passes through the Phoenix Arizona, USA) revealed that land use types adjacent to the river is significantly correlated with the species composition of wintering waterfowl, but habitat factors' characteristics (e.g., water physiognomy, shoreline composition, and terrestrial vegetation cover) are important for influencing waterfowl species richness and abundance (Andrade et al., 2018). Other studies have suggested that the increase of several waterfowl species' density depends on an increase in wetland area (Murray et al., 2013) and length of shoreline (Suter, 1994); while water bird species richness is positively correlated with waterbody size (Hoyer and Canfield Jr, 1994), which was suggested to be of greater importance than habitat diversity (Roach and Griffith, 2015). Additionally, the trophic status (nitrogen and phosphorus concentration) of lakes has a significant positive effect on the species richness and diversity of waterfowl (Cardoni et al., 2011; Hoyer and Canfield Jr, 1994).

In addition to habitat structure and nutrient concentrations, available food resource (such as the coarse fish yield for piscivorous species) also accounted for a significant part of variation in waterfowl species richness and abundance (Suter, 1994). Many parts of aquatic plants directly act as vegetative food for waterfowl, or indirectly sustain invertebrates as the animal food (Anderson and Smith, 2000; Ma et al., 2010). In fact, the nutrient concentration (ammonium, phosphate, nitrate, and the percent organic matter) has been demonstrated to be directly connected to the vegetation physiognomy, and indirectly to bird community composition and abundance (Cardoni et al., 2011). Thus, feeding guild composition of waterfowl depends on the composition of their food resources (the relative abundance of available vegetative food and animal food; Zhang et al., 2011). However, the presence of specific feeding guilds, such as the waders and dabbling ducks, depends not only on the availability of aquatic plant food resources, but also on whether suitable foraging habitat is available (Ma et al., 2010; Suter, 1994).

Rivers and lakes provide important ecological (Bertalan et al., 2018), sight value, and societal benefits (Hale et al., 2019; Suri et al., 2017). Because large rivers and lakes are tied to regional groundwater, they are more difficult to be replaced by buildings or other artificial surface than isolated patches of common terrestrial vegetation. As a result, urban rivers and lakes provide hot spots of local biodiversity, especially when combined with island habitat, which could be the refugia of not only local but also migratory species (Ocampo and Londono, 2015; Scheffer et al., 2010). Hassall (2014) suggested that, urban wetlands contribute a significant portion of regional bird biodiversity in urban settings. Thus, data on urban birds in rivers and lakes are important for conserving biodiversity and monitoring ecosystem health (Liu et al., 2006; Ogden et al., 2014), such as the application of the index of water bird community integrity (IWCI) for monitoring estuarine ecosystem integrity (DeLuca et al., 2008) and the utilization of aquatic

bird community composition for the assessment of the nutrient impact on the lagoon food-web (Fernández et al., 2005).

It is well known that urbanization leads to the disruption of ecosystem processes and thus a loss in ecological services (Cabeza, 2010; García and Martínez, 2012; Tagil et al., 2018). However, we have insufficient knowledge about the key linkages between urban aquatic ecosystems and surrounding environments of birds in urban areas and how increased urbanization affects birds. We hypothesize that food and habitat play a predominant role in determining avian community structure, especially for aquatic bird species.

In this study, we focused on both the aquatic and forest bird communities because aquatic habitats and the surrounding terrestrial habitats should be regarded as a whole. Withered or decayed leaves and branches of trees fall into nearby rivers or lakes, while their roots absorb water and nutrients from the same waterbody. Taking bird as example, although *Anatidaes* and *Rallidaes* are typical waterfowl species, without the surrounding terrestrial vegetation, many water bird species (including them) will be deficient from the aquatic habitats; which is especially true in urban area given a high-level nearby disturbance level. However, in our urbanizing world, a significant number of waterbodies have long been existing or were newly created without enough attention paid on their surrounding terrestrial vegetation, which result as a deficient of a diverse bird community within those habitats. Thus, our research on both the aquatic and the nearby forest habitats could provide ecological suggestions for the planning and management of urban habitats with obvious waterbodies aimed at bird biodiversity conservation. In addition, given the consistent focus on the green components, a comprehensive analysis of the green, gray and blue components could be fruitful in urban studies (Moll et al., 2019).

Building on prior studies, our research focused on the following questions: (1) how is the urbanization gradient of Beijing related to avian communities inhabiting urban rivers and lakes, (2) how do environment factors, including water quality, water nutrient concentrations, food resources, hydrologic conditions, adjacent terrestrial habitats, and landscape configurations differentially influence forest and water bird species richness and abundance, and (3) based on a species-environment analysis of species guilds, how does urbanization exert its detrimental effects on avian communities?

2. Methods

2.1. Study sites

This study was conducted in Beijing, China, located in the northernmost end of the North China Plain (39°54′ N, 116°23′ E). The average elevation within the city boundary is 43.5 m. The Taihang and Yanshan Mountains border the western and northern parts of the city, whereas the southern and eastern portions of Beijing are bordered by the alluvial plains of the Chaobai and Yongding Rivers. A series of concentric beltways encircle the city, with intensity of urbanization and population density declining from the city center outwards. The climate of Beijing is typical of the North Temperate Zone: semi-humid, continental monsoon, dominated by hot-wet summers and cold-dry winters. There are 1085 distinguishable river basins in Beijing, all of which belong to tributaries in the Haihe River Basin that feed a number of large river systems: the Yongding River, North Canal, Chaobai River, Daqing River, and Ji canal. In addition, there are 88 reservoirs within the city limits and no natural lakes. Rivers and reservoirs throughout Beijing play an important role in supplying water to residents, regulating and storing floodwaters, providing recreational opportunities for citizens, and maintaining aquatic biodiversity (e.g., aquatic bird diversity).

In this study, we randomly selected 39 reservoirs and river reaches in all to evaluate the association between birds residing in aquatic and wetland habitats and environmental characteristics. Our sampled sites were randomly distributed around the entire city area, with the minimum distance between sites exceeding 1 km (Fig. 1).

2.2. Bird survey

We used the belt-transect method (Gregory et al., 2004) to count birds, by species, at the end of the breeding season (August and September 2018). Each site was only surveyed once to give equal weight to each location. We selected three to ten 50-m wide by 200-m-long transect belts that paralleled the shorelines at each sample site, with the number of transects per site and their location being based on the total length of the sampled shoreline at a given site and the configuration of different urban components (including structural components, agents, and abiotic components) nearby the sample sites (Moll et al., 2019; Vaughan et al., 2007), that is, more belt transects were set in positions with a more diverse vegetation community (for example, plots with more plant species and mature vertical structure) and a more complicated topography (more slopes, hollows, and rocks), and the total length of transect belts within a given site account for about half of the gross shoreline length. Sampled belts were arranged so that the 50-m width extended from shore to the land. Each transect belt at a given site was located at least 100 m from other transects. When sampling, the surveyor walked along the shoreline edge of each transect at constant speed of about 2 km h^{-1} counting all bird species and individuals that were in the sampled belt, and water birds within the sample belt and the whole waterbody were censused by the same time. Birds flying over transects, above the tops of trees or hovering in the sky were recorded, but not included in the bird-environment relationship analyses. To ensure consistent conditions among sites, bird surveys were only conducted on clear days without strong wind (i.e., wind >30 km h^{-1}), within 3 h after sunrise and 2 h before sunset. The first author performed all the bird surveys.

2.3. Acquisition of environmental variables and aquatic community indexes

We measured the following environmental variables in four categories: (1) water geomorphic indices [including waterbody area (Area_Wat), waterbody width (Width_Wat), shoreline length (Edge_Wat), water body shoreline/area ratio (Edge/Area), and length of unarmored shoreline (Edge_Soft)], (2) water quality indices [including turbidity (Turb), dissolved oxygen (DO), 5-day biological oxygen demand (BOD$_5$), total phosphorus (TP), total nitrogen (TN), chlorophyll *a* (Chla)], (3) habitat indices [including number of islands (Islands), indices for submerged (Spe_Subm) and emergent plants (Area_Emer, % Cov_Emer), indices for zooplankton (Div_Zoop, Den_Zoop), indices for zoobenthos (Spe_Zoob, Div_Zoob, Den_Zoob), indices for phytoplankton (Div_Phyt, Bio_Phyt), indices for fish communities (Spe_Fish, Ind_Fish, Div_Fish), and indices of terrestrial vegetation (Spe_Tree/ Shrub/Herb, Div_Tree/Shrub/Herb)], and (4) landscape pattern indices [including linear distance to the urban center (Dis_Cent), forest area within the 200-m buffer zone around the waterbody boundary (AreaF_200b), percent forest/artificial surface area relative to the gross area of the 1000-m radius buffer region (%For_1kmb, %Art_1kmb), largest patch index and connectivity of forest patches within a 1000-m buffer (LPIF_1kmb, CONECTF_1kmb)]. Details of our environmental variables are listed and defined in Table 1.

We created two sets of boundaries. The first boundary set was created from shorelines depicted in Google Earth, using historical imagery of the same period with our field survey, which was the boundary of waterbody. Another set of boundaries was created in ArcGIS by extending the waterbody boundary outward 50-m from the water body, which is the sample boundary and exactly covered the bird survey area, and it will be used for landscape pattern analysis.

The land-use and land-cover (LULC) data contains two parts, one for the urban area (lands within the 5th beltway of Beijing (Fig. 1) (i.e., LULC_Urban), and another area for the rest of the city (i.e., LULC_nonUrban). LULC_Urban areas were interpreted from a high-resolution satellite image of Beijing acquired in 2015 (Pleiades image, 0.5-m panchromatic band resolution), using the nearest neighborhood
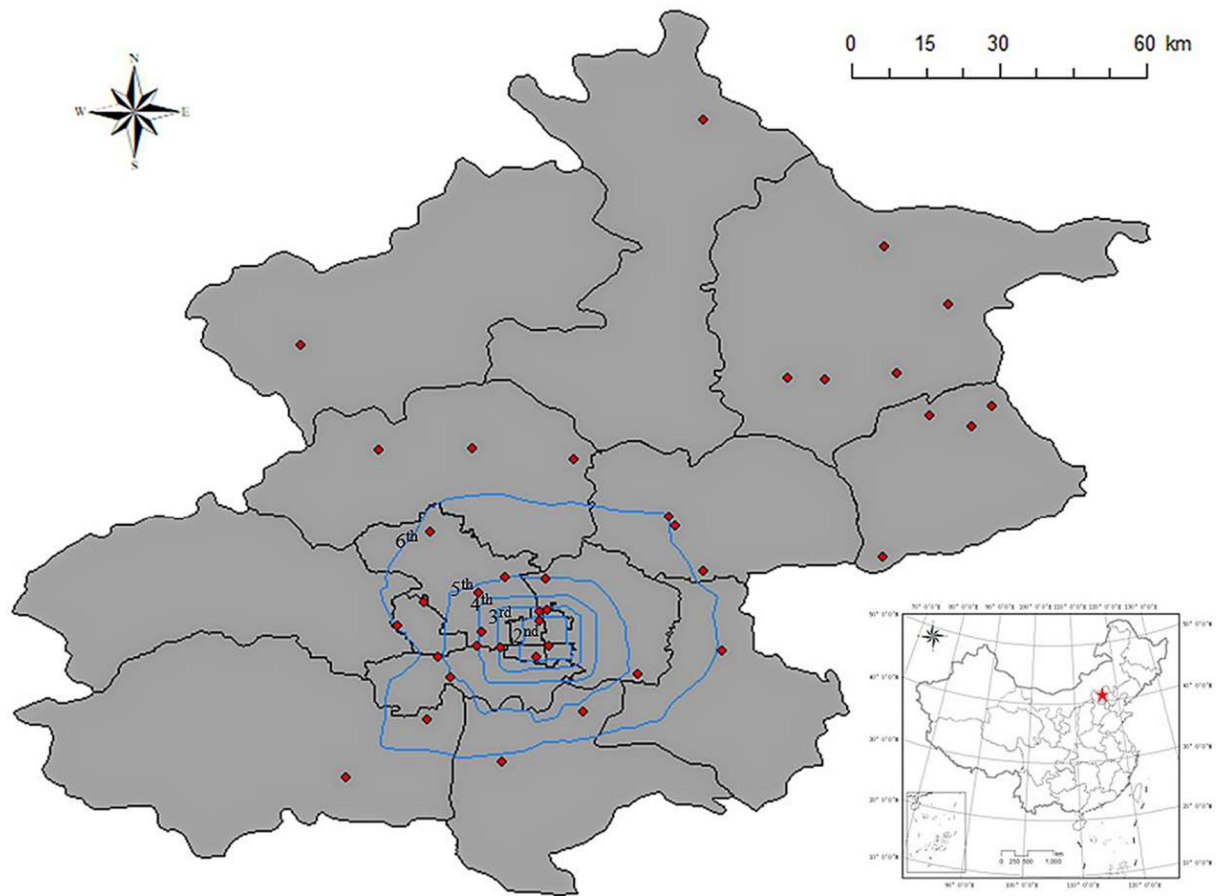
Fig. 1. Geographic distribution of sampled river reaches and reservoirs. The red points designate the geometric centers of the sampled sites, the gray area is the area within Beijing's municipal boundary, the blue concentric lines identify Beijing's major beltways, and the black lines denote county boundaries. The beltways are numbered sequentially (1st–6th) from the inner ring to the outer ring, which are official names identifying them. Referring to criterion of Batary et al. (2018), area within the 5th ring is the urban region, area between the 5th and 6th ring is the suburban area, while the rest part within the city boundary is the rural area dominated by natural or semi-natural mountains and farm lands.

method in eCognition 8.7.1. The LULC_nonUrban areas were interpreted from a GF-1 image acquired in 2019 (2-m panchromatic band resolution, 8-m multispectral band resolution), using the random forest method in ArcGIS 10.3. Both LULC images were sub-classified into six categories: forest, grassland, waterbody, aquatic plants, artificial surfaces, and bare ground. Because the non-urban area was dominated by farmland and mountains, which are more homogeneous than urban landscape, the accuracies of our LULC interpretations for the two areas (urban and nonurban) were both relatively accurate (>90%).

Based on the LULC data, we used Fragstats 4.1 to calculate landscape indices of different land cover types (Table 1), indices measured at three spatial scales: at focal waterbody patches, within 200-m radius buffer zones, and within 1000-m radius buffer zones.

At each site, we measured the length of natural or semi-natural shoreline (soft edge), which included mud, grass, or scattered pebbles. Several studies have demonstrated the importance of natural shoreline for water bird reproduction (Chabot et al., 2014; Henningsen and Best, 2005). In contrast, artificial or hardened shorelines (hard edge) consisting of flat, steep concrete, or stone walls that would largely obstruct the exchange of nutrients and energy between terrestrial and aquatic environments, thus eroding ecosystem integrity. We used canopy cover within a 200-m radius outside from the waterbody boundary as potential terrestrial habitat for forest birds, because in most of our sample sites the width of riparian forest is within that distance and the possible effects of waterbody will be diluted as the distance from it increases, especially in fragmented urban landscapes, in which circumstances ecosystem processes may suddenly become disrupted (With and Crist, 1995). We used 1 km as the maximum radius we examined because Pautasso

(2007) identified 1 km as the critical threshold distance needed for evaluating environment-diversity relationships. Descriptions of field survey methods for other explanatory variables, such as aquatic organism and terrestrial vegetation data, are provided in Appendix A.

2.4. Data analysis

In order to examine the bird-environment relationship, bird species were classified as being either water birds (Grallatore, Natatores, and Alcedinidae) or forest birds (all species other than water birds) (sensu Zheng, 2012). Based on foraging preferences, water birds were subdivided into three sub-groups [aquatic omnivores (Wat_Omn), aquatic carnivores (Wat_Car), and piscivorous (Wat_Pis)], whereas forest birds were subdivided into five sub-groups [granivores (For_G), insectivores (For_I), insectivore/frugivores (For_IF), omnivores (For_O), and carnivores (For_C, they mostly hovered above tree line and were only recorded in non-urban sites; thus, they were excluded from habitat preference analyses)]. Bird species were also classified as Passenger (P), Resident (R), Summer breeders (S), or Winter visitors (W) (sensu Zheng, 2017). Bird species diversity was calculated with the Shannon-Wiener diversity index ($H' = \sum P_i \log P_i$, P_i), which is the ratio of individual number of species i and total individual number.

To explore the effects of urbanization on avian communities and the distribution of various guilds along an urbanization gradient, we first calculated a bivariate regression between bird guilds and the proportion of urban matrix in the landscape, using the generalized linear model (GLM). Then we drew scatter plots for species richness and abundances of forest and water birds separately with %Art_1kmb as the x-axis.

Table 1
Definitions of selected explanatory variables and descriptive statistics.

Types	Variable abbreviation	Description	Data source	Mean (range)
Water geomorphic indices	Area_Wat (ha)	Total area of water body at each site, including portion under emergent plants	Calculated from the water border shape files	59.01 (0.29–238.00)
	Width_Wat (m)	Average width of a sampled river or lake at each site	Area_Wat/length of longest axis of water body	179.31 (4.16–958.50)
	Edge_Soft (km)	Total length of unarmored shoreline	Measured in Google earth and checked during the field survey	7.78 (0.00–55.44)
	Edge_Wat (km)	Total length of shoreline at each site	Calculated from the water border shape files	10.68 (0.48–56.45)
	Edge/Area	The shoreline/area ratio of sampled water bodies	Edge_Wat/Area_Wat	0.48 (0.05–4.04)
Water quality indexes	Turb	Turbidity of water	Measured with a turbidometer	8.62 (0.28–28.60)
	DO (mg/L)	Dissolved oxygen	Measured with a portable dissolved oxygen meter	6.03 (1.59–11.18)
	BOD$_5$ (mg/L)	5-day biological oxygen demand	Five-day culture method	4.36 (1.02–10.00)
	TP (mg/L)	Total phosphorus	Measured with the ammonium molybdate and spectrophotometer	0.14 (0.01–0.73)
	TN (mg/L)	Total nitrogen	Measured with the potassium persulfate oxidation and ultraviolet spectrophotometer	5.94 (1.18–14.92)
	Chla (g/L)	Density of Chlorophyll a	Measured with a spectrophotometer	9.88 (0.26–71.75)
Habitat indexes	Islands	Number of islands within the waterbody boundary	Counted in Google earth	3.49 (0.00–51.00)
	Area_Emer (ha)	Surface area covered by emergent vegetation in each site	Measured in Google earth	13.75 (0.00–179.28)
	Spe_Subm	Species richness of submerged plants	Number of submerged plant species	1.56 (0.00–8.00)
	%Cov_Emer	Coverage of emergent vegetation	Area_Emer/Area_Wat	16% (0.00%–83.00%)
	Div_Zoop	Species diversity of zooplankton	Calculated with the Shannon-Wiener index	2.45 (1.17–3.10)
	Den_Zoop (/L)	Individual density of zooplankton	Counted in the 0.1 mL plankton counting chamber	5.58 (0.50–32.15)
	Spe_Zoob	Species richness of zoobenthos	Number of zoobenthos species	5.28 (0.00–13.00)
	Div_Zoob	Species diversity of zoobenthos	Calculated with the Shannon-Wiener index	1.47 (0.00–3.41)
	Den_Zoob (/m^2)	Individual density of zoobenthos	Average number of zoobenthos in each quantitative sample	163.89 (0.00–1276.44)
	Div_Phyt	Species diversity of phytoplankton	Calculated with the Shannon-Wiener index	2.75 (0.52–4.12)
	Bio_Phyt (mg/L)	Biomass density of phytoplankton	Measured as the concentration of Chla	13.72 (0.88–52.48)
	Spe_Fish	Species richness of fish	Number of fish species	6.37 (0.00–16.00)
	Ind_Fish	Fish abundance	Counted in the field survey	222.04 (0.00–1906.00)
	Div_Fish	Species diversity of fish	Calculated with the Shannon-Wiener index	1.34 (0.00–2.78)
	Spe_Tree/Shrub/Herb	Species richness of trees, shrubs, and herbs	Number of tree/shrub/herb species	9.13 (2.00–17.00); 4.13 (0.00–15.00); 16.85 (8.00–30.00)
	Div_Tree/Shrub/Herb	Species diversity of trees, shrubs, and herbs	Calculated with the Shannon-Wiener index of tree/shrub/herb	1.62 (0.18–2.39); 0.77 (0.00–2.18); 2.16 (1.01–2.87)
Landscape pattern indices	Dis_Cent (km)	Linear distance from the geometrical center of each sample site to the geometric center of Tian'an'men Square (city center)	Measured in Google earth	36.98 (1.33–108.88)
	%For_1kmb	% cover of woodland cover within 1000-m radius buffer region of the sample boundary	Obtained from the classified Pleidas/GF-1 image	33.67% (0.78%–96.87%)
	%Art_1kmb	% cover of artificial land within 1000-m radius buffer region of the sample boundary	Obtained from the classified Pleidas/GF-1 image	51.93% (3.34%–93.21%)
	LPIF_1kmb	Largest patch index for woodland patches within 1000-m radius buffer region of the sample boundary	Obtained from the classified Pleidas/GF-1 image	23.76 (0.04–96.76)
	CONECTF_1kmb	Connectivity of woodland patches within 1000-m radius buffer region of the sample boundary	Obtained from the classified Pleidas/GF-1 image	42.61 (0.80–93.17)
	AreaF_200b (ha)	Canopy cover within 200 m buffer region of the waterbody boundary	Obtained from the classified Pleidas/GF-1 image	85.62 (0.30–910.26)

We selected models based on AICc of the bird-environment relationship among forest birds. After species were classified by feeding type, species richness of feeding types showed little variation among sampled sites; therefore, we only modeled species richness and diversity of all forest birds combined. For any given feeding type, we only analyzed abundance data. The relationship between various forest bird community indices (including species richness, species diversity, abundance of granivores, insectivores, insectivore-frugivores, and omnivores) and environmental variables (all indices displayed in Table 1, except for variables for water quality, aquatic plants and animals) were determined with linear regression. We log-transformed response variables (when necessary) to ensure that the initial model fits the basic assumptions of linear regression (including skewness, kurtosis, link function, and heteroscedasticity), which we checked with the gvlma function in R package "gvlma" (Pena and Slate, 2006). We calculated the correlation matrix and variance inflation factors (VIF) to solve severe multicollinearity problems between predictive variables, and we deleted severe outliers based on Bonferroni test ($p < 0.05$). When the $\sqrt{vif}$ of all independent variables in initial models were <2.0 (Kabacoff, 2015), we used stepwise regression to obtain the best model subsets based on AICc. Models with their ΔAICc <2.0 [ΔAICci = AICci − min (AICc)] were chosen as the best subset of models (sensu Burnham and Anderson, 2002).

Based on our food-habitat determination assumptions, we choose aquatic carnivores and piscivores (hereafter referred to as fish foragers) as the response guilds for water birds, because they both forage on fish. Because the other water birds we surveyed were mainly ducks (*Anatidae*), such as Eastern Spot-billed Duck (*Anas zonorhyncha*) and Mallard (*Anas platyrhynchos*), which often acquire supplemental food (such as paddy and leftovers) from humans (and gather at specific sites where they were routinely fed). Therefore, we determined that fish foragers represented the major guild of water birds sampled in most of our sites.

We used a Structural Equation Model (SEM) to analyze the bird-environment relationships for fish foragers. Before conducting the SEM analysis, we first calculated the correlation matrix between fish foragers' community indices (species richness and species abundance) and all of our environment variables listed in Table 1. When the predictive variables reached a suitably high significance level ($p < 0.05$), we performed bivariate regressions between fish foragers' community indices and the significant predictive factors. The procedures were repeated for possible latent variables that were significantly correlated with avian community indices. We primarily selected variables into the SEM that showed a significant contribution in our bivariate analysis.

SEM is a method for establishing, evaluating, and testing causality models, which focus on ecological process (rather than patterns) (Doncaster, 2007). SEM uses a comparative fit index (cfi) to assess closeness of fit (acceptable when cfi ≥ 0.90), use the rooted mean square error of approximation (rmsea) to judge the convergence (acceptable when rmsea ≤0.08), and use the standardized root means square residuals (srmr) to evaluate the convergence of models (acceptable when srmr ≤0.08). In addition, a high p value ($p > 0.05$) indicates that a model cannot be rejected as an explanation of observations. In our SEM analysis, as in the model selection method used for forest birds, we also used the gvlma function to ensure that each single model in the SEM fit the basic assumptions of linear regression, and outliers recognized by Bonferroni test ($p < 0.05$) were excluded.

All statistical analyses for forest birds were conducted with R version 3.5.1 (R Development Core Team, 2018) and the software packages "ape" (Paradis et al., 2004), "MuMIn" (Barton, 2014), "car" (Fox and Weisberg, 2010), and "lavaan" (Rosseel, 2012).

3. Results

3.1. Community structure of surveyed birds

In our field survey, we recorded 88 bird species and 6487 birds, which belonged to 16 orders and 36 families. Forest birds accounted for 60 species and 5134 individuals, whereas water birds accounted for 28 species and 1353 individuals. In the forest bird community, insectivores or resident species each both accounted for nearly half of all species counted, whereas summer breeders and passengers together accounted for a little >50%. In the water bird community, fish forager species accounted for 68% of species richness, whereas summer breeders (which migrated to the study area to breed) and passengers (which migrated through the study area without a long-time stay), both of which migrate, accounted for 88% of water bird species richness (Fig. 2). A bird species list and a classification scheme are provided in Appendix B.

### 3.2. Distribution patterns of bird guilds along an urbanization gradient

The bivariate generalized linear regression shows that species richness of residents (β = −0.008, $p = 0.020$, CI = −0.015, −0.002), fish foragers (β = −0.013, $p = 0.006$, CI = −0.022, −0.004) in water bird community, and insectivore-frugivores (β = −0.015, $p = 0.015$, CI = −0.026, −0.004) in forest bird community, species abundances of insectivores (β = −0.011, $p = 0.010$, CI = −0.019, −0.003), granivores (β = 0.013, $p = 0.048$, CI = 0.001, 0.027), and insectivore-frugivores (β = −0.014, $p = 0.064$, CI = −0.029, 0.000) in forest bird community, are significantly and negatively correlated with the proportion of urban matrix the 1-km landscape circle (Fig. 3). With the exception of granivores' abundance, most bird guilds were negatively correlated with intensity of urbanization. Interestingly, when percent artificial surface exceeded 50% of the buffer region, all the avian community indices exhibited an obvious decline or transition relative to increasing urban matrix proportion (Fig. 4), and such pattern are better fitted than that of GLM given the generally higher R^2 values.

When we analyzed forest birds and water birds separately, we found that some unimodal relationships existed between avian community indices and %Art_1kmb. Such patterns were more significant for water birds, due to the highest explanatory power of the second order model for species richness of water birds ($R^2 = 0.37$), and its p value being lowest ($p = 0.000$). In contrast, the regression model for species richness of forest birds was not significant ($p = 0.36$) and its explanatory power was the lowest ($R^2 = 0.06$) (Fig. 5). Only the model for forest bird species richness (Fig. 5a) is non-significant ($p > 0.05$), and the linear regression (to fit the data set) did not perform better than ($R^2 = 0.0021$, $p = 0.78$).

3.3. Relationship between avian community and environment factors

3.3.1. Key influencing factors for forest birds

Our model selection processes produced 21 models for all forest bird community indices (Table 2). All the models were potentially explanatory ($p < 0.01$), and 10 of the analyzed environment variables were included in the best subsets of models. Among the most explanatory predictors (relative importance ≥0.80), Width_Wat was important for explaining species richness and abundance of insectivores, Dis_Cent was important for explaining abundance of granivores and omnivores, Islands was important for explaining both species diversity and the abundance of insectivores. Only Dis_Cent showed a negative relationship, which was consistent among all models in which the parameter appeared (Table 2). For the five avian community indices, species richness was mainly related to Width_Wat (relative importance = 1.00) and Spe_Shrub (relative importance = 0.87); species diversity was mainly related to number of islands (relative importance = 1.0) and AreaF_200b (relative importance = 0.83); abundance of insectivores
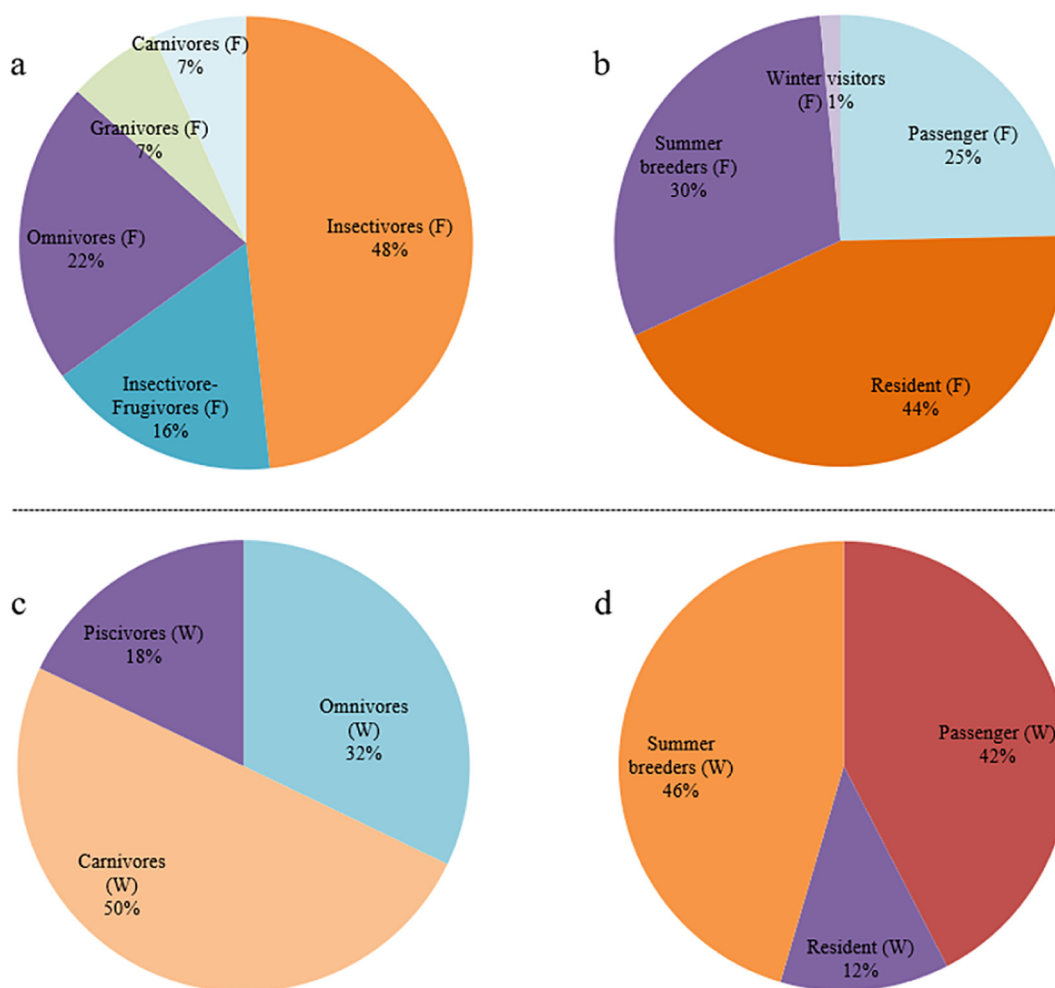
Fig. 2. Proportions of bird species sampled, by feeding and residence types. Panels: (a) feeding types of forest birds, (b) resident types of forest birds, (c) feeding types of water birds, and (d) resident types of water birds. Forest birds denoted with "F"; water birds denoted with "W".

was related to Width_Wat (relative importance = 1.0), Edge_Soft (relative importance = 0.82), and number of islands (relative importance = 0.80); abundance of granivores was related to Dis_Cent (relative importance = 1.0) and Edge_Wat (relative importance = 1.0); abundance of omnivores was related to Dis_Cent (relative importance = 1.0) and Spe_Tree (relative importance = 1.0).

3.3.2. Key influencing factors for water bird distributions

The bivariate regression showed that species abundance of fish foragers is significantly and positively related to Spe_Fish, Area_Wat, Edge_Wat, Edge_Soft, Islands, %Cov_Emer, Area_Emer, Turb, Bio_Phyt, and Div_Zoob (Fig. 6). Further analysis showed that species richness of fish (Spe_Fish), which is probably a latent variable, is significantly and positively related to Area_Wat (slope = 0.03, R^2 = 0.24, p = 0.00) and Edge_Wat (slope = 0.11, R^2 = 0.11, p = 0.04) and is significantly and negatively related to TP (TP * 1000) (slope = −0.01, R^2 = 0.23, p = 0.00).

To assess the critical paths through which urbanization exerts detrimental effects on water birds, we used SEM to estimate all partial regression coefficients simultaneously to identify probable pathways (Fig. 7). Two similar models were derived from our SEM analysis (Fig. 7a & b), for species richness and abundance of fish foragers. Overall, the models provided highly significant fits for the covariance matrices (species richness: p = 0.333, Cfi = 0.997, rmsea = 0.050, srmr = 0.039; abundance: p = 0.943, Cfi = 1.000, rmsea = 0.000, srmr = 0.053). Estimates for all of the pathway parameters were positive except for that of TP (−0.508). The key structural difference between the two

models was the selection of Area_Wat or Spe_Fish as descriptive variables (Fig. 7). Together, Dis_Cent and Islands explained 43.9% of variation in Edge_Soft among the 39 sampled sites. While the two models each explained more than half of the variation in species richness and abundance of fish foragers. The species richness model implied that the total effect of island number (Islands), waterbody size (Area_Wat), length of unarmored waterbody edge (Edge_Soft), and linear distance from the urban center (Dis_Cent) are 0.43, 0.39, 0.36, 0.16, respectively; and the abundance model suggests that the total effects size of island number (Islands), length of unarmored waterbody edge (Edge_Soft), fish species richness (Spe_Fish), linear distance from the urban center (Dis_Cent), total phosphorus (TP), zoobenthos species richness (Spe_Zoob), and zooplankton species diversity (Div_Zoop) are 0.57, 0.41, 0.21, 0.20, −0.11, 0.08, 0.05, respectively.

3.3.3. Covariates and alternative models

We used several covariates measured during the study to construct alternative multivariate hypotheses and explore whether we could find better explanations for covarying data in our data set. For example, different fish forager species may prefer different fish species; phytoplankton is a major food resource of zooplankton; aquatic organic debris account for a considerable proportion of energy intake for zooplankton zoobenthos; and almost all life entities need oxygen. To account for the potential influence of the mentioned environmental variables, we altered the structural equation models to include additional paths displayed in Table 3. However, all of the changes produced an inferior fit to the data set (see Table 3).
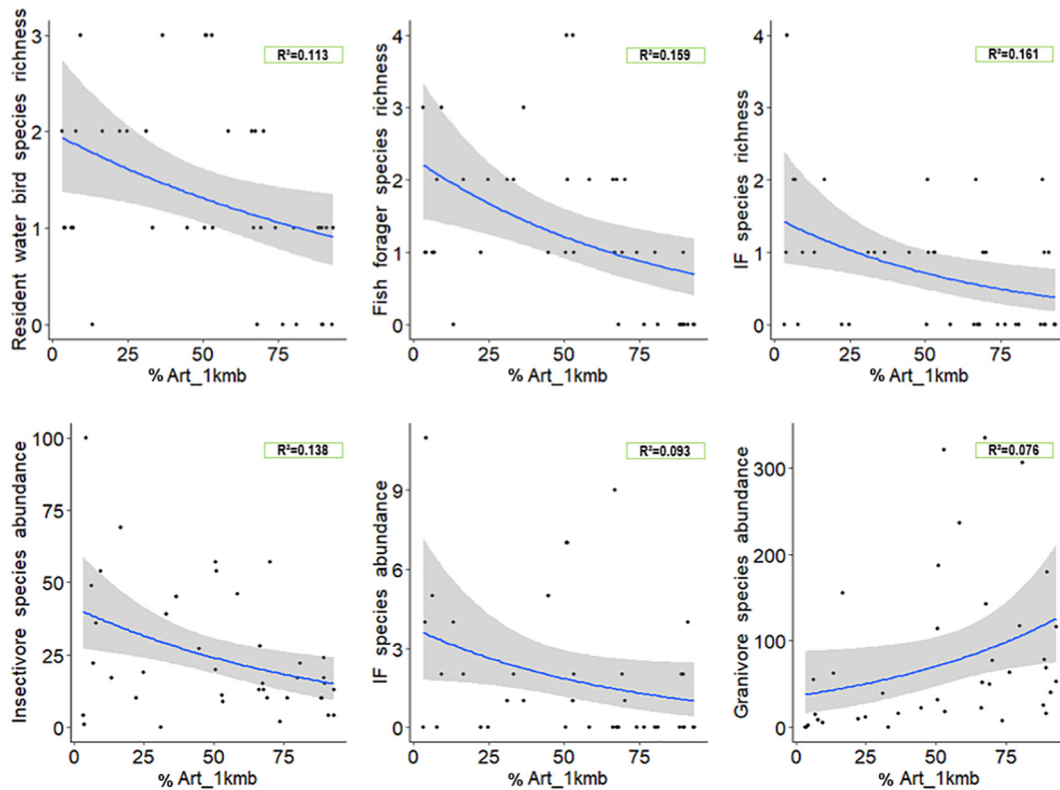
Fig. 3. GLM with quasi-Poisson plotting various bird guilds against proportion of urban cover in a 1000-m radius buffer of sampled sites. R^2 were provided in the box at the top right of each graph.

4. Discussion

Our results examining the pattern of urban bird distribution along an urban gradient were consistent with a prior review study that suggested urbanization favors granivores and reduces populations of insectivores and insectivore-frugivores (Chace and Walsh, 2006). We too found differences in distribution patterns among functional guilds and between forest and water birds. For example, our study was similar to, but found a more nuanced trend than did Palacio et al. (2018), which suggested a constant urbanization effect on bird guilds, in that when the
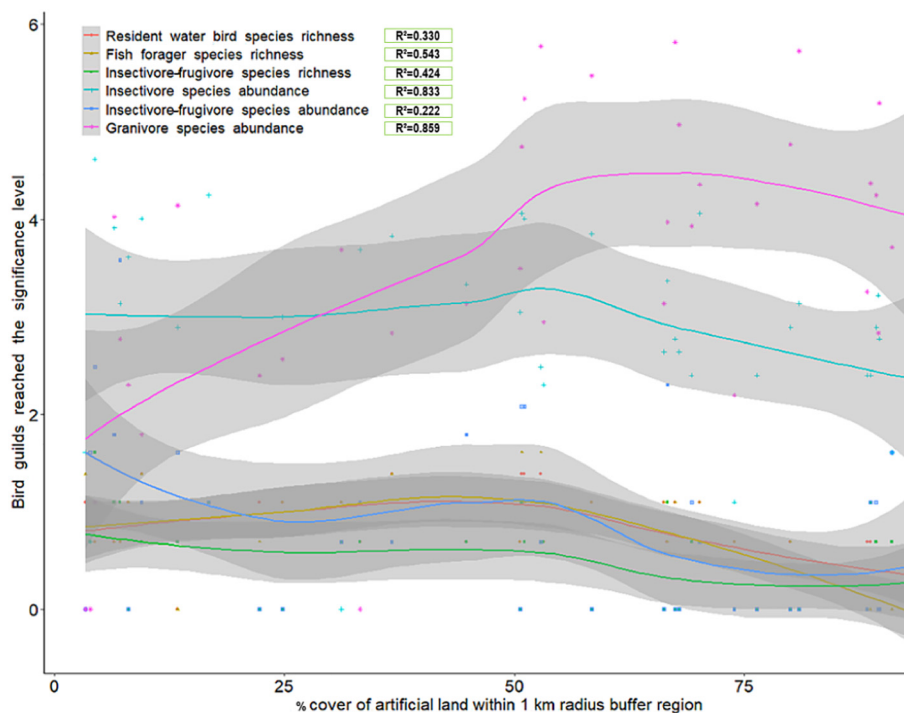
Fig. 4. Nonparametric "loess" regression to improve fit, showing that all bird guilds declined after the 50% threshold of %Art_1kmb. R^2 were provided in the box following each legend. All data were log-transformed [ln (x + 1)]. The gray area stands for the confidential intervals for the fitted lines.
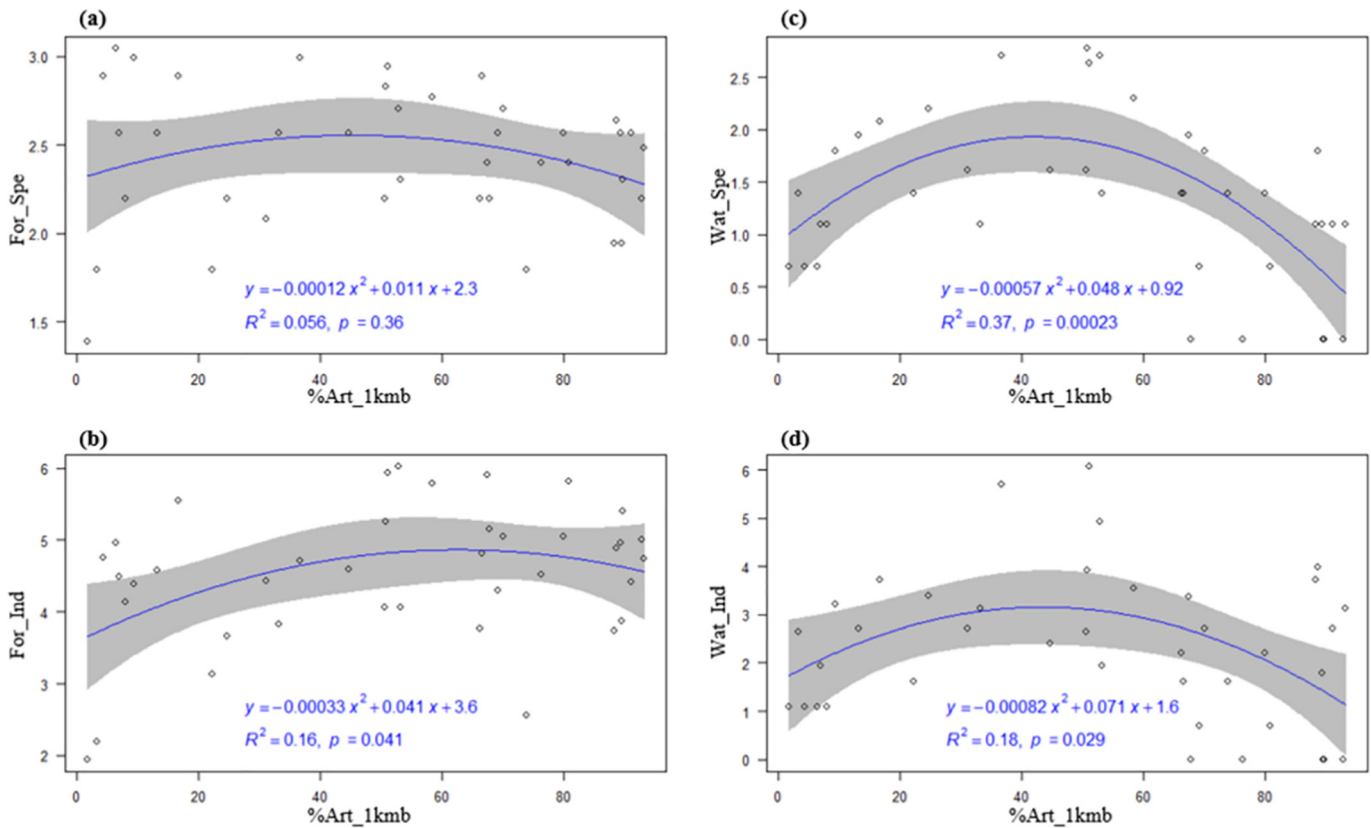
Fig. 5. Non-linear regression between avian communities and proportion of urban cover in a 1000-m radius buffer region of sample sites. Panels: regressions against (a) forest bird species richness, (b) forest bird abundance, (c) water bird species richness, and (d) water bird abundance. The suffix "_Spe" denotes species richness, the suffix "_Ind" denotes species abundance. All avian community indices were log transformed [ln (x + 1)] to improve linearity.

artificial surface exceeded 50% of landscape cover, we found that bird diversity loss by an accelerated speed or even change their attitude towards urbanization, such as the granivore guild. Our differing conclusions from those of Palacio et al. (2018) may be due to the length (urban-rural gradient or urban-rural-natural gradient) of urbanization gradient (Batary et al., 2018).

A recent meta-analysis study by Batary et al. (2018) highlighted the importance of suburban areas along an urbanization gradient in affecting bird species richness. They found that richness declined precipitously in the suburban portion of the gradient while abundances peaked. Although most studies reviewed by the meta-analysis focused on forest birds, our conclusions focusing on water and forest birds are similar. In their meta-analysis study, the authors assumed that the threshold between urban and suburban was 50% impervious surface, whereas our study, which statistically depicted the urbanization gradient with the assistance of valid LULC classification, indicated that the

Table 2
Results of model selection analysis for forest birds based on AICc criterion. All models are very significant ($p < 0.01$) with an R^2 ranging from 0.24 to 0.52 (median = 0.37). The suffix "_Div" denotes species diversity. Descriptions for predictive variables please see Table 1.

	Model	Islands	Edge_Soft	Edge_Wat	Width_Wat	Spe_Tree	AreaF_200b	Dis_Cent	%For_1kmb	Spe_Shrub	Div_Shrub	AICc/Wi
For_Spe	1		0.56		0.35					0.32		100.6/0.34
	2	0.19	0.49		0.34					0.28		101.3/0.24
	3	0.24		0.46	0.33					0.24		102.0/0.17
	4			0.53	0.34					0.29		102.6/0.13
	5		0.55		0.29	0.26						102.6/0.13
For_Div	1	0.40					0.29		0.27			107.1/0.28
	2	0.34					0.40					107.4/0.25
	3	0.42							0.40			108.1/0.17
	4	0.37					0.32		0.26		0.17	108.3/0.16
	5	0.31					0.43				0.18	108.4/0.14
For_O_Ind	1			0.46		0.36		−0.48				104.6/0.33
	2					0.39	0.50	−0.57				105.5/0.21
	3			0.30		0.38	0.27	−0.57				106.0/0.16
	4	0.20				0.34	0.50	−0.57				106.1/0.16
	5		0.40			0.37		−0.41				106.1/0.15
For_I_Ind	1	0.21	0.40		0.35							79.5/0.42
	2	0.26	0.37		0.31				0.13			80.9/0.20
	3		0.47		0.36							81.0/0.20
	4	0.25		0.37	0.34							81.1/0.18
For_G_Ind	1			0.44				−0.65				103.4/0.56
	2			0.46		0.20		−0.63				103.8/0.44
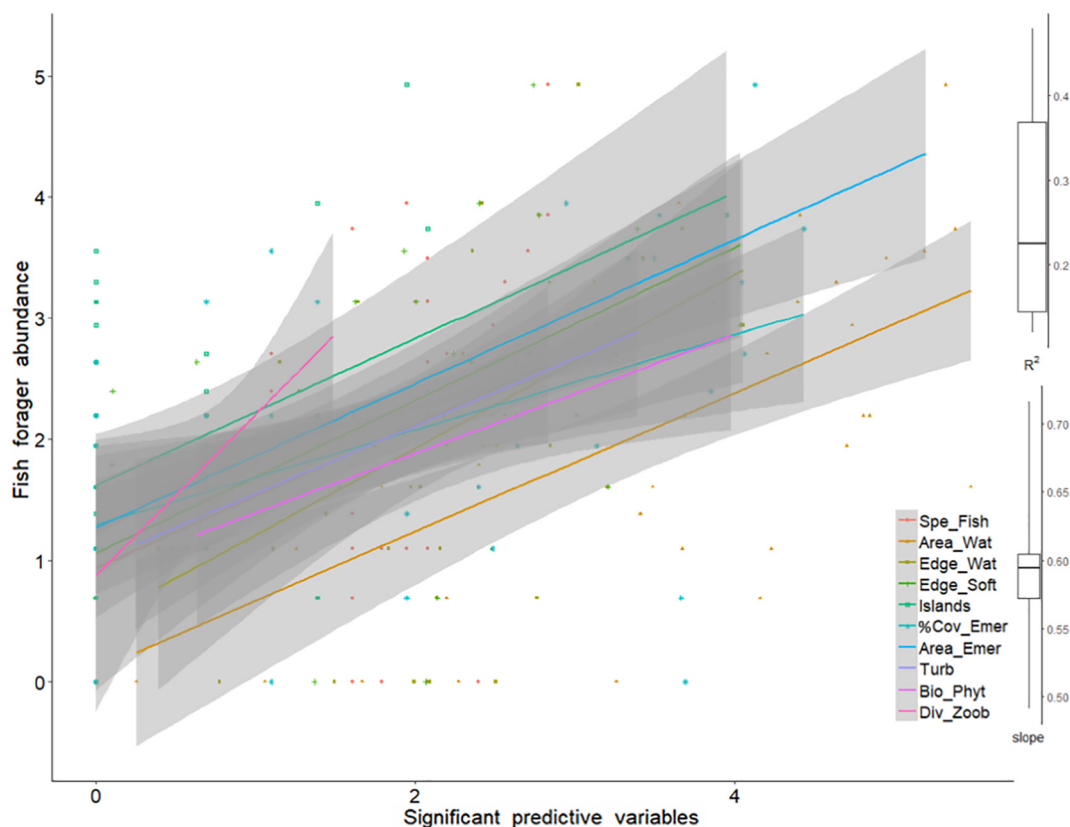
Fig. 6. Bivariate regression between species abundance of fish foragers and predictive factors that were significant ($p < 0.05$) in the correlation matrix. Slopes for the exhibited bivariate models ranged from 0.49 to 0.72, and R^2 ranged from 0.11 to 0.48 (depicted on right side of figure). Regressions for species richness and abundance of fish foragers were redundant and so only results of species abundance are displayed. All variables were log transformed [$\ln (x + 1)$] to improve linearity.
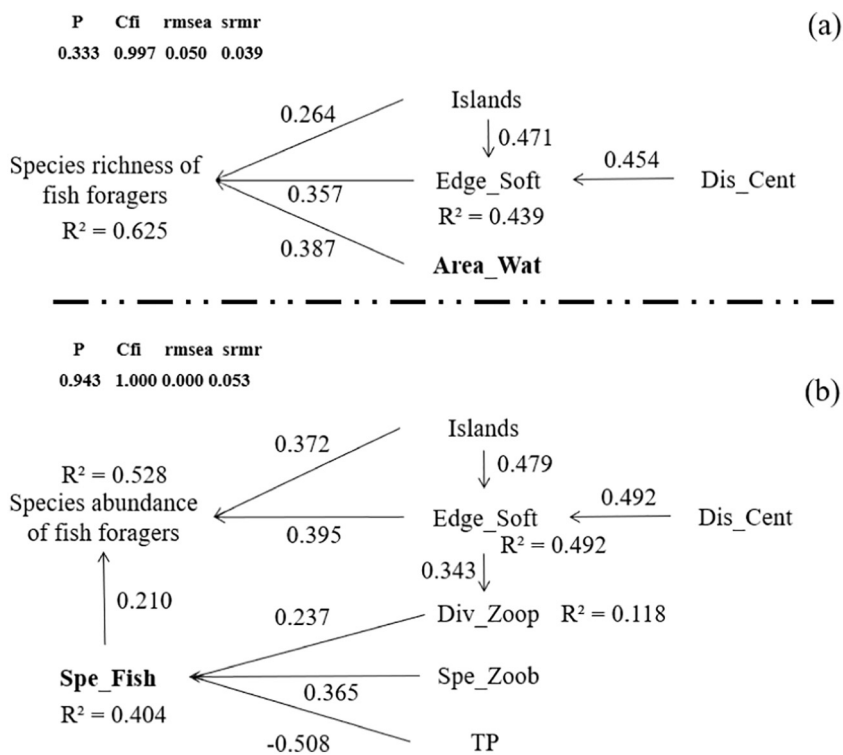
Fig. 7. Results of SEM analysis used to explore the key environment factors affecting fish foragers. Panels: (a) species richness and (b) abundance. Standardized regression coefficients are provided for each path (near arrows), which provide the change in standard deviation of Y for any given change in standard deviation for X. Model fit metrics are provided at the top left of each panel. Each latent variable or response variable in this figure is accompanied by R^2 values, depicting the amount of variation explained by the models. The best-fit metrics indicated that our models cannot be rejected as a potential explanation for the variation in the fish forager data sets. All variables were log transformed [$\ln (x + 1)$] to improve linearity.

Table 3
Additional pathways for SEM analysis and the comparison with the selected models.

	M	Additional effect	Description	df	χ^2	p	AIC	R^2	cfi; rmsea; srmr	Notes[a]
Species richness of fish foragers	1		Base model in Fig. 6 (a).	2	2.20	0.333	502.73	0.63	0.997; 0.050; 0.039	
	2	Dis_Cent	Species richness of fish foragers = f(Dis_Cent)	1	2.00	0.158	504.53	0.63	0.983; 0.160; 0.037	Dis_Cent (0.668)
	3	Spe_Fish	Species richness of fish foragers = f(Spe_Fish)	3	2.22	0.527	590.46	0.63	1.000; 0.000; 0.034	Spe_Fish (0.836)
Species abundance of fish foragers	1		Base model in Fig. 6 (b).	13	6.07	0.943	408.28	0.53	1.000; 0.000; 0.053	
	2	Div_Phyt	Div_Zoop = f(Div_Phyt)	16	9.51	0.891	424.81	0.53	1.000; 0.000; 0.057	Div_Phyt (0.267)
	3	BOD5	Div_Zoop = f(BOD5)	16	7.73	0.956	451.67	0.53	1.000; 0.000; 0.055	BOD5 (0.706)
	4	DO	Div_Zoop = f(DO)	16	7.38	0.965	421.97	0.53	1.000; 0.000; 0.052	DO (0.653)
	5	Spe_Zoob~Edge_Soft	+f(Spe_Zoob~Edge_Soft)	15	10.63	0.778	408.84	0.53	1.000; 0.000; 0.073	
	6	Spe_Zoob~BOD5	+f(Spe_Zoob~BOD5)	20	21.55	0.365	457.49	0.51	0.976; 0.047; 0.144	Div_Zoop (0.060); Spe_Zoob (0.052); BOD5 (0.303)
	7	Spe_Zoob~TP	+f(Spe_Zoob~TP)	15	18.05	0.26	416.25	0.51	0.955; 0.076; 0.154	Div_Zoop (0.060); Spe_Zoob (0.053); BOD5 (0.134)
	8	Spe_Zoob~DO	+f(Spe_Zoob~DO)	20	19.92	0.463	426.52	0.51	1.000; 0.000; 0.139	Div_Zoop (0.059); Spe_Zoob (0.050); BOD5 (0.120)

[a] The special note showed the p value of the given predictive variables which are non-significant within the given algorithms.

urbanization threshold should be a little >50% imperviousness. Thus, our results align well with the assumption set forth by Batary et al. (2018). Comparing with other threshold studies, which usually choose different habitat patches to stand for urbanization gradient (Blair, 1996; With and Crist, 1995), we suggested that the definition of 'urbanization' in this study could be more statistically accurate. And habitat loss probably one of the major drivers underling the bird-urbanization trend (Richmond et al., 2015; Yin et al., 2017). Also, prior threshold researches often found differentiated threshold values for species guilds with different habitat preferences (Swift and Hannon, 2010), however, it is interesting that in this study all bird guilds classified as different categories (including residence type and feeding type) respond to urbanization with the same step, which is valuable for future landscape planning and avian biodiversity conservation in urban area. For example, the top planners could efficiently conserve avian biodiversity within citizen's neighborhood through the control of the intensity of urbanization to be less than the threshold (~50% impervious surface) beyond which avian communities are difficult to sustain their population.

Resident birds have a greater opportunity to adapt to urban conditions, due to their year-round presence, whereas migratory birds tend to be more flexible in habitat selection and foraging behaviors (Heim et al., 2018; Salewski and Jones, 2006). Thus, we assumed that habitat selection by water birds could be well represented by the preference of the resident guild. On the other hand, all other guild indexes which showed a significant correlation with urban cover in the 1 km radius buffer region were defined by feeding types, including both forest and water birds. Based on our data, we suggested that the influence of urbanization gradient towards avian biodiversity could be well reflected by different feeding guilds, which indicate that the large scale distribution pattern of food resources probably described a major part of the variation in sympatric avian community structure. Indeed, there is a lack of direct demonstration on the food-bird relationship for forest birds, which has been carefully considered and finally abandoned because of on the following concerns. According to our field survey, several plant species (e.g., *Lonicera maackii* (Rupr.) Maxim. and *Koelreuteria paniculate*), which could provide abundant seeds for many frugivore and omnivore bird species, were planted as roadside green in Beijing. Therefore, these plants were common and widely distributed around all of our randomly selected study sites, which is the so-called food resource homogenization. As a result, the food-bird relationship for forest birds could not be clarified by diligently counting the

exact number of food resource trees because every single large tree with abundant fruits/seeds would able to attract and sustain a significant part of the bird species and population of a given site, while the rest same trees remain untouched within our survey period. Yet, we are not negating the importance of food for forest birds, we suggest there probably be a tradeoff between available food (associated with high-level disturbance) and eligible habitat (with less food), which could address critical drivers for the city-scale distribution of forest birds thus desire further study.

The contrasting gradient distribution patterns we identified between water birds and forest birds has rarely been examined in previous studies (Andrade et al., 2018). When analyzed by functional guilds, we found that water bird species and abundance varied similarly to forest birds when considered as a whole; however, water birds showed a more statistically significant unimodal pattern than did forest birds, relative to both species richness and abundance.

Among the key predictive factors we determined for avian communities, distance to the urban center, number of islands, gross and unarmored shoreline length were important for both forest and water bird species richness and abundance. Distance from the urban center indicated the resource distribution patterns along an urbanization gradient (Start et al., 2018) and so our results suggest that both forest and water birds avoid the most-highly urbanized city center given the background that some of our study sites are large in size (both the forest and waterbody area). However, with our SEM analysis, we determined that distance from the urban center could probably only indirectly affect water bird species and populations via its impact on the length of unarmored shoreline. Andrade et al. (2018) suggested that shoreline complexity plays a key role in shaping water bird diversity and abundance, we found that shoreline features (as measured by gross and unarmored shoreline length) could influence both water and forest bird species richness and abundance. Forest birds may respond to shoreline features partly in reaction to edge effects (Drinnan, 2005), whereas available water edge could also provide water drinking sites. For water birds, unarmored shoreline provides available habitat, thus playing a key role in shaping water bird community structure.

The species-area relationship is not only applicable to forest birds, as shown by a variety of urban park bird studies (Xie et al., 2016; Zhou and Chu, 2012), but the concept is also applicable to water birds. That is, species richness of water birds shows a positive relationship with waterbody size. However, as is reflected by the algorithms for species

abundance of fish foragers, further enhancement of water bird populations depends on habitat condition (e.g., fish species richness indicates food resource availability for fish foragers, which were the most abundant species of water birds in our study, and in this study, fish individual number account for 66% of the variation in fish species richness ($p < 0.001$)), which could be well demonstrated by the food chains and the connection between primary consumer assemblages and the so-called high quality habitats. While prior studies have demonstrated the critical negative influence of excessive phosphorus on macroinvertebrate assemblage (which could be fish foragers' food) (Pan et al., 2000) and the potential effects on endangered species and ecosystem services (Putt et al., 2019), our results suggested that it could exert direct negative influence on fish species richness, and after transition process, its negative effects on fish forager abundance is still considerable.

5. Conclusions

Our study confirms our hypothesis that food (though only for water birds) and available habitat play the key roles in determining avian community structure along an urban intensity continuum. Our study provides an urbanization threshold beyond which species richness and abundance declines. The study also specifies a process through which urbanization likely exerts its detrimental effects on water birds. Both findings should be useful in planning future urban landscapes in ways that could be used to conserve avian species diversity. The differences in the strength of response to urbanization and the similarities in the gradient distribution pattern between forest and water birds should receive more attention in future urbanization gradient studies. We emphasize that rivers and lakes (including their fringing wetlands and open water, aquatic habitats) provide exceptional habitat in urban settings relative to built-out infrastructure) and so play a key role in urban biodiversity. We recommend that aquatic environments and their wetlands be identified and protected as an integrated part of urban environments. Based on our results, there are several recommendations we want to supply to possible stakeholders or policy makers, including the efficient application of the half percent impervious surface threshold in the future urban planning, reduced use of armored river or lake banks, the improvement of the surrounding terrestrial vegetation (especially the expansion of forest area, and the enrichment of the shrub species diversity), and the protection of the aquatic food chains should receive more attention aimed at water bird conservation.

CRediT authorship contribution statement

Shilin Xie: Methodology, Formal analysis, Writing - original draft, Writing - review & editing.**Xiaoke Wang:**Methodology, Formal analysis, Writing - original draft, Writing - review & editing.**Yufen Ren:**Methodology, Investigation, Data curation.**Zhimin Su:**Investigation, Data curation.**Yuebo Su:**Investigation, Data curation.**Siqi Wang:**Investigation, Data curation.**Weiqi Zhou:**Methodology, Investigation, Data curation, Formal analysis, Writing - original draft, Writing - review & editing.**Fei Lu:**Methodology, Investigation, Data curation, Formal analysis, Writing - original draft, Writing - review & editing.**Yuguo Qian:**Investigation, Data curation.**Cheng Gong:**Formal analysis, Writing - original draft, Writing - review & editing.**Binbin Huang:**Investigation, Data curation.**Zhiyun Ouyang:**Methodology, Formal analysis, Writing - original draft, Writing - review & editing.

Declaration of competing interest

The authors of this study declare no competing interests.

Acknowledgements

Appendix A. Field survey methods except for avian survey

We conducted all field surveys in August to September 2018, each survey was conducted once at each sample site.

A.1. Water quality survey

We sampled three to about five fixed transects in each water body, based on the length and area of the sampled water body. (We typically sampled the upper, middle, and lower reaches of a river or the third to fifth sections of a lake's main axis.) We used a discrete layer sampler to collect water samples at every 0.5- or 1-m depth, based on the maximum depth of the waterbody sampled. Where the maximum water depth was <3 m, we sampled at every 0.5-m depth interval; where water depth was >3 m, we sampled at 1-m depth interval. [Water sample collection procedures followed the *Technical specifications for surface water and sewage monitoring* of China (HJ/T91-2002; State Environmental Protection Administration, 2002b). Water chemical analyses followed procedures of the *Environmental quality standards for surface water* of China (GB 3838-2002; State Environmental Protection Administration, 2002a)]. We measured depth of water, turbidity (Turb), dissolved oxygen (DO) profiles, and Chlorophyll a (Chla) with an MI-parameter meter (AQUAREAD, EXO2). Turb and DO were measured every 0.5 or 1 m from the water surface to the bottom. We measured 5-d biological oxygen demand (BOD5) using the five-day culture method (HJ 505-2009, 2009), total phosphorus (TP) using the ammonium molybdate spectrophotometric method (GB 11893-1989, 1989), and total nitrogen (TN) with a potassium persulfate oxidation UV spectrophotometer (GB 11894-1989, 1989). We measured both TP and TN with an automatic continuous flow analyzer (SEAL, AA3).

A.2. Phytoplankton survey

Phytoplankton survey was conducted in the same transects with water quality survey. Quantitative survey was conducted in the inner waterbody, we used a vertical sampler (1 L volume) to collect water samples from a portable rubber raft, at different depths. That is, when the water depth <2 m, only water samples at 0.5 m depth were collected; when 2 m ≤ water depth < 5 m, water samples below 0.5 m of water surface and 0.5 m above the bottom were collected; and when water depth >5 m, an additional collection was conducted at the middle depth other than that of 2 m – 5 m depth. Qualitative survey was conducted following the quantitative survey. Standing on the raft, the surveyor swept (by a speed of <0.3 m s^{-1}) a plankton net (0.064 mm mesh size) by a "∞" route, and the captures were frequently collected.

All of the collected samples were treated by fixatives (Lugol's solution and 4% of formalin) within 5 min. After sedimentation and agitation, each collected water sample was censused at least twice within the 0.1 mL plankton counting chamber (20 * 20 mm), by species, and we averaged the individual number of the same species within the quantitative samples. The biomass (mg/L) of phytoplankton was measured as the concentration of Chlorophyll a (see the method in water quality survey). Species diversity of phytoplankton was calculated with the Shannon-Wiener index.

A.3. Zooplankton survey

Zooplankton survey was conducted by a similar method and in the same transects with that of phytoplankton survey. For quantitative survey, water samples were also collected by a vertical sampler (1 L volume) at the same depths, but the difference in this part being that 1 L water was collected for small zooplanktons (protozoa, rotifers, and immature crustacean), while 20 L water was collected for large zooplanktons (mature crustacean) in each

sample point. Qualitative sampling and taxonomical classification are the same with that of phytoplankton survey, but duplicate sampling time within the counting process for qualitative samples is at least five times, and the number is two for quantitative samples. Species diversity of zooplankton was calculated with the Shannon-Wiener index.

A.4. Zoobenthos survey

Zoobenthos community was censused in the transects used for water quality survey. We conducted qualitative sampling at shallow water, and each habitat type (including micro-habitats with stones, submerged plants, sands, grass, and mud) was censused within one-two quadrats (4 m^2). In this method, the surveyor agitated the benthic deposits, and simultaneously sweep a net (0.064 mm mesh size) to capture the escaping zoobenthos within 30 min, with a 5-min interval between two successive processions within the same sample point. Quantitative sampling was conducted in the inner waterbody. From a boat, the surveyor lowered the Peterson dredger (1/16 m^2) to the bottom, four to about six times per transect, to quantitatively collect the benthic deposits. After filtration by a decimate sift (0.45 mm mesh size), taxonomic information of zoobenthos species was recognized by naked eyes or a microscope. Breakable annelids, etc., were counted by head, and the dead shell of mollusks were excluded from our count. Individual number was also counted to calculate individual density. Species diversity of zoobenthos was calculated with the Shannon-Wiener index. We show the immediate picking of macrobenthos after capture in Fig. A1.

Fig. A1. Immediate picking of macrobenthos after capture.

A.5. . Aquatic vegetation survey

We censused macrophytes (including emergent plants and submerged plants) near the transects we sampled for water quality. From the rubber raft, we placed a floating wooden sampling frame (2 × 2 m) on the water and counted macrophytes within the frame, by species. (Composition and dominance were later determined by calculating relative abundance). Submerged macrophytes were visually surveyed along the same transects as those sampled for macrophytes (Van Onsem and Triest, 2018). Short vegetation (including submerged macrophytes and the young plant of emergent plants) rooted below water or in turbid water was pulled to the surface with an iron tong with a net.

A.6. . Fish survey

We collected fish using the cage-trap and net-casting methods. We placed the net cage traps (1.0 cm diameter opening, 8 m long) near the ends of the transects within which we measured water quality and sampled aquatic organisms. At least one cage (with bait) was placed in each transect line, which we withdrew the next day. We casted cast nets (1.5 cm mesh size, 2 m radius) at one to three points along each transect. Taxonomic information and number of each species were recorded for each fish captured. We also measured the weight of each individual, but for species with >20 individuals, we weighed all individuals at once and divided by the number of fish captured (to determine mean weight). After taking digital images, we released all fish on site. We show the cage-trap method in Fig. A2.

Fig. A2. Deployment and retrieval of cage nets used in fish community surveys.

A.7. . Terrestrial vegetation survey

We surveyed terrestrial vegetation within 100-m of the shoreline along each transect. We established 4–6 quadrats at each sample site to measure plant community composition. Quadrat dimensions were 10 m by 50–100 m (length laid parallel to shore) and established at the forest edge near the shoreline. Within each quadrat, we established 3–5 subplots (1 × 1 m) by alternated plot placement equidistant along both sides of quadrat center-lines in which we tallied herbaceous plants, by species.

We counted and tallied all tree and shrub species in each quadrat, height of each tree and shrub species, diameter at breast height (DBH) of trees (woody stems >1.5 m tall), and the height and cover of each herb species (in subplots). Shrub and herb height were measured with a tapeline, and tree height was visually estimated. We measured DBH with a ruler at 1.3 m above the ground (Canedoli et al., 2018). Species diversity of trees, shrubs, and herbs were calculated with the Shannon-Wiener index.

Appendix B. . Bird species recorded in our field survey and the classification on residence and feeding types

Taxonomy	Number	English name	Scientific name	Residence types	Feeding types
Water birds					
Charadriiformes					
Scolopacidae	1	Green Sandpiper	*Tringa ochropus*	P	O
	2	Common Snipe	*Gallinago gallinago*	P	O
	3	Common Sandpiper	*Actitis hypoleucos*	SP	O
	4	Common Redshank	*Tringa totanus*	P	O
Laridae	5	Caspian Tern	*Hydroprogne caspia*	S	O
	6	Siberian gull	*Larus smithsonianus*	P	O
Anseriformes					
Anatidae	7	Eastern Spot-billed Duck	*Anas zonorhyncha*	S	O
	8	Mallard	*Anas platyrhynchos*	R	O
	9	Mandarin Duck	*Aix galericulata*	SP	O
	10	Swan Goose	*Anser cygnoid*	P	O
	11	Ruddy Shelduck	*Tadorna ferruginea*	P	O
	12	bar-headed goose	*Anser indicus*	P	O
Pelecaniformes					
Ardeidae	13	Gray Heron	*Ardea cinerea*	S	O
	14	Chinese Pond Heron	*Ardeola bacchus*	S	O
	15	Intermediate Egret	Ardea intermedia	SP	O
	16	Little Egret	*Egretta garzetta*	S	O
	17	Striated Heron	*Butorides striata*	P	O
	18	Black-crowned Night Heron	*Nycticorax nycticorax*	S	O
	19	Yellow Bittern	lxobrychus sinensis	S	O
	20	Cinnamon Bittern	*Ixobrychus cinnamomeus*	S	O
Ciconiiformes					
Ciconiidae	21	Black Stork	*Ciconia nigra*	S	O
Podicipediformes					
Podicipedidae	22	Great Crested Grebe	*Podiceps cristatus*	P	P
	23	Little Grebe	*Tachybaptus ruficollis*	R	P
Gruiformes					
Rallidae	24	Common Coot	*Fulica atra*	S	O
	25	Common Moorhen	*Gallinula chloropus*	SP	O
Suliformes					
Phalacrocoracidae	26	Great Cormorant	*Phalacrocorax carbo*	SP	P
Coraciiformes					
Alcedinidae	37	Common KingPer	*Alcedo atthis*	R	P
	38	Crested KingPer	*Megaceryle lugubris*	R	P
Forest birds					
Accipitriiformes					
Accipitridae	27	Eurasian Sparrowhawk	*Accipiter nisus*	SP	C
Strigiformes					
Strigidae	28	Eurasian Eagle-Owl	*Bubo bubo*	R	C
Falconiformes					
Falconidae	29	Amur Falcon	*Falco amurensis*	S	C
	30	Common Kestrel	*Falco tinnunculus*	R	C
Columbiformes					
Columbidae	31	Oriental Turtle Dove	*Streptopelia orientalis*	R	G
	32	Spotted Dove	*Streptopelia chinensis*	R	G
	33	Eurasian Collared Dove	*Streptopelia decaocto*	R	G
Galliformes					
Phasianidae	34	Common Pheasant	*Phasianus colchicus*	R	O
	35	koklass pheasant	*Pucrasia macrolopha*	R	O
Bucerotiformes					
Upupidae	36	Common Hoopoe	*Upupa epops*	S	I
Piciformes					
Picidae	39	Gray-capped Woodpecker	*Dendrocopos canicapillus*	R	I
	40	Great Spotted Woodpecker	*Dendrocopos major*	R	I
	41	Gray-headed Woodpecker	*Picus canus*	R	I
	42	Eurasian wryneck	*Jynx torquilla*	WP	I
Passeriformes					
Corvidae	43	Azure-winged Magpie	*Cyanopica cyanus*	R	O
	44	Common Magpie	*Pica pica*	R	O

(continued)

Taxonomy	Number	English name	Scientific name	Residence types	Feeding types
	45	Red-billed Blue Magpie	*Urocissa erythroryncha*	R	O
	46	Carrion Crow	*Corvus corone*	P	O
	47	Large-billed Crow	*Corvus macrorhynchos*	R	O
	48	Rook	*Corvus frugilegus*	RP	O
	49	Eurasian Jay	*Garrulus glandarius*	R	IF
Paridae	50	Marsh Tit	*Poecile palustris*	R	I
	51	Cinereous Tit	*Parus cinereus*	R	I
	52	Silver-throated Bushtit	*Aegithalos glaucogularis*	R	I
Pycnonotidae	53	Light-vented Bulbul	*Pycnonotus sinensis*	S	O
	54	Collared Finchbill	*Spizixos semitorques*	R	O
Phylloscopidae	55	Yellow-browed Warbler	*Phylloscopus inornatus*	P	I
	56	Dusky Warbler	*Phylloscopus fuscatus*	S	I
	57	Arctic Warbler	*Phylloscopus borealis*	P	I
Sylviidae	58	Vinous-throated Parrotbill	*Sinosuthora webbiana*	R	I
	59	Chinese Hill Babbler	*Rhopophilus pekinensis*	R	I
Sturnidae	60	Crested Myna	*Acridotheres cristatellus*	R	O
	61	White-cheeked Starling	*Spodiopsar cineraceus*	S(SW)	O
	62	Silky Starling	*Spodiopsar sericeus*	S	O
Turdidae	63	Chinese Blackbird	*Turdus mandarinus*	R	IF
Muscicapidae	64	Siberian Stonechat	*Saxicola maurus*	P	I
	65	Taiga Flycatcher	*Ficedula albicilla*	P	I
	66	Daurian Redstart	*Phoenicurus auroreus*	S	I
	67	Plumbeous Water Redstart	*Rhyacornis fuliginosa*	R	I
Leiothrichidae	68	Plain Laughingthrush	*Garrulax davidi*	R	IF
Acrocephalidae	69	Black-Browed Reed Warbler	*Acrocephalus bistrigiceps*	S	I
	70	Oriental Reed Warbler	*Acrocephalus orientalis*	S	I
Cisticolidae	71	Zitting Cisticola	*Cisticola juncidis*	R	I
Passeridae	72	Eurasian Tree Sparrow	*Passer montanus*	R	G
Motacillidae	73	Water Pipit	*Anthus spinoletta*	P	I
	74	White Wagtail	*Motacilla alba*	PS	I
	75	Gray Wagtail	*Motacilla cinerea*	PS	I
Fringillidae	76	Gray-capped Greenfinch	Chloris sinica	R	IF
Emberizidae	77	Little Bunting	*Emberiza pusilla*	P(WP)	IF
	78	Yellow-throated Bunting	*Emberiza elegans*	SP	IF
	79	Chestnut-eared Bunting	*Emberiza fucata*	P	IF
	80	Meadow Bunting	*Emberiza cioides*	R	IF
Hirundinidae	81	Barn Swallow	*Hirundo rustica*	SP	I
	82	Red-rumped Swallow	*Cecropis daurica*	S	I
	83	Eurasian Crag Martin	*Ptyonoprogne rupestris*	S	I
Laniidae	84	Brown Shrike	*Lanius cristatus*	SP	I
	85	Long-tailed Shrike	*Lanius schach*	S	I
Dicruridae	86	Black Drongo	*Dicrurus macrocercus*	S	I
Oriolidae	87	Black-naped Oriole	*Oriolus chinensis*	S	IF
Zosteropidae	88	Japanese White-eye	*Zosterops japonicus*	SP	IF

Residence types: R = Resident, P = Passenger, S = Summer breeders, W = Winter visitors; feeding types: P = piscivorous, G = granivores, I = insectivores, IF = insectivore-frugivores, O = omnivores, and C = carnivores.

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