Science of the Total Environment 624 (2018) 1561–1576

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Detecting the response of bird communities and biodiversity to habitat loss and fragmentation due to urbanization

Xi Xu [a,b], Yujing Xie [a], Ke Qi [a], Zukui Luo [c], Xiangrong Wang [a,*]

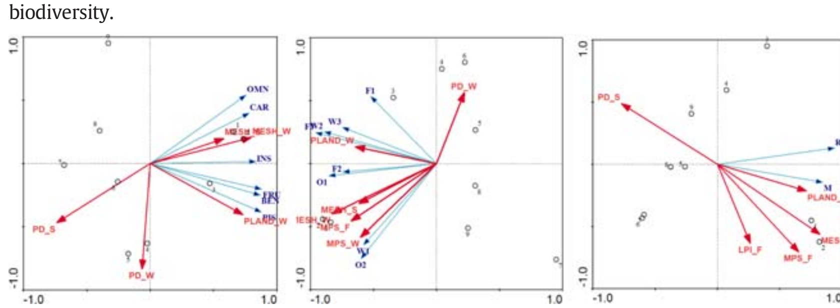
[a] *Department of Environmental Sciences and Engineering, Fudan University, 220 Handan Road, Shanghai 200433, PR China*
[b] *Institute of Green Investment, Detao Group, 2200 Wenxiang Road, Shanghai 201620, PR China*
[c] *School of Environment and Life Sciences, Kaili University, Kaili, Guizhou, 556011, PR China*

HIGHLIGHTS

- Species richness of bird communities varied with the process of urban expansion.
- The MSA could account for the process of species replacement due to urbanization.
- Birds inhabiting wetland and open spaces were mostly affected by the habitat change.
- The area and connectivity of wetlands positively affected bird diversity.

GRAPHICAL ABSTRACT

Fig. 10 Redundancy analysis biplots showing correlation between habitat loss and fragmentation and bird species biodiversity.



Food type Habitat type Seasonal status

ARTICLE INFO

Article history:
Received 20 July 2017
Received in revised form 11 December 2017
Accepted 12 December 2017
Available online xxxx

Editor: Jay Gan

Keywords:
Bird diversity
Landscape change
Mean species abundance
Biodiversity accounting
Urbanization

ABSTRACT

Birds are considered a good model for indicators of biodiversity response to habitat variations, as they are very sensitive to environmental change. However, continuous observations of habitat alterations from undisturbed landscapes to human-dominated ones, as well as the associated effects on bird biodiversity, are lacking. In this study, New Jiangwan Town in Shanghai, China was selected to illustrate the response of bird species, and thus biodiversity, to habitat loss and fragmentation. Land use/land cover (LULC) data and bird records from 2002 to 2013 were collected and analyzed. The results suggested that, due to urban sprawl, the area of wetland and shrub land had dropped by 82.4% and 87.3% by the end of 2013. Four different urbanization stages were identified in terms of the spatio-temporal variations in the landscape. To measure bird biodiversity, species richness and relative abundance were calculated, and they could account for the overall trend in biodiversity but might mask the process of species replacement. As an indicator of biodiversity accounting, the mean species abundance (MSA) of the original species would not include exotic or invasive species in its calculation, and its value decreased from 100% to 76.8% to 52.2% to 24.5% in the four corresponding stages. Finally, suggested by redundant analysis, the effects of habitat loss and fragmentation on bird biodiversity differed in various bird communities, and the area and connectivity of wetlands were the most significant variables. Our findings could provide important information to inform bird biodiversity protection and habitat restoration.

1. Introduction

Biodiversity is a multi-dimensional and complex concept that includes the variations in genes, species, populations, communities, and

* Corresponding author.
E-mail addresses: xuxi@detaoma.com (X. Xu), xieyj@fudan.edu.cn (Y. Xie), xrxrwang@fudan.edu.cn (X. Wang).

https://doi.org/10.1016/j.scitotenv.2017.12.143
0048-9697/© 2017 Elsevier B.V. All rights reserved.

ecosystems from local scales to the living world as a whole (Robinson et al., 1992). Recently, the global decline and the increase in pressures on biodiversity have been widely discussed (Gaston, 2000; Cockell et al., 2006; Schluter and Pennell, 2017), and approximately 12% of birds, 25% of mammals and more than 32% of amphibians are estimated to be under the threat of extinction (Millennium Ecosystem Assessment, 2005b). Additionally, the amount of wildlife is half that of 40 years ago, according to the "The Living Planet Report 2014" (McLellan et al., 2014). The worldwide loss of biodiversity is generally caused by multiple drivers that operate in space and time. In terrestrial ecosystems, landscape change, especially urban expansion, which leads to habitat loss and fragmentation, is recognized as one of the most significant direct drivers (Ortega-Álvarez and MacGregor-Fors, 2011; Reis et al., 2012). It is even considered the primary pressure on biodiversity, as suggested by the UN's 3rd Global Biodiversity Outlook, and its influence continues to intensify (Millennium Ecosystem Assessment, 2005a; Marzluff et al., 2012; Aronson et al., 2014).

Birds offer a good system for understanding the effects of urbanization on habitat structure and composition (Reis et al., 2012; Bregman et al., 2014), as the available details on the characteristics of bird species could provide the most comprehensive information about functional traits (Vandewalle et al., 2010). Additionally, birds comprise a broad range of species, including urban avoiders and urban-tolerant birds that are very sensitive to environmental change (Gregory et al., 2005; Vandewalle et al., 2010). In particular, it is the habitat loss and fragmentation caused by urbanization that significantly threatens birds (González-Oreja, 2011).Habitat loss and fragmentation can disrupt key biological processes by reducing the breeding success of migrant species, limiting dispersal, and decreasing resource acquisition (Bregman et al., 2014). Generally, the process of urbanization can decrease bird species richness as well as affect the species composition gradient and increase the biomass and density of bird communities (Chace and Walsh, 2006; Zhou et al., 2012), and bird community structure might vary significantly between urbanized and non-urbanized areas (Reis et al., 2012; Zhou et al., 2012). Furthermore, natural habitat loss and fragmentation would also pose a significant threat by affecting the composition, structure and ecological functions of biodiversity (Scolozzi and Geneletti, 2012). The importance of understanding habitat changes and bird biodiversity has been highlighted by many researchers (Carrete et al., 2009; Scolozzi and Geneletti, 2012; Quesnelle et al., 2013; Blandón et al., 2016; Keinath et al., 2017). Biodiversity is commonly compared among different sampling areas, such as suburbs (MacGregor-Fors, 2008), urban-rural gradient (Yeh and Huang, 2009), new areas of urban growth (Zhou et al., 2012) and the green spaces in different urban districts (Sandström et al., 2006). Several studies of the change in amphibian species through long-term observation has been conducted to identify the time-lag of the response of biodiversity to human activity (Löfvenhaft et al., 2004; Whitworth et al., 2017). However, such research is still lacking, especially studies examining the influence of habitat loss and fragmentation on bird biodiversity from undisturbed landscapes to human-dominant ones through continuous observation (Jokimäki et al., 2011; Ciach & Fröhlich, 2016).

Many difficulties still exist when measuring the process of biodiversity loss, as biodiversity is a broad concept that refers to all biotic variations from the level of genes to species to ecosystems (Purvis and Hector, 2000). Biodiversity can be measured based on the richness and distribution of species, as well as the change in species "abundance", and it is usually characterized by the decrease in some original species and the increase in some other, opportunistic species(Pfisterer and Schmid, 2002). Although the decrease in original populations is a significant signal of biodiversity loss, highly proliferating species may sometimes invade, and their infestations might even become plagues (UNSD, 2014).

Species richness is the most commonly recognized facet of biodiversity (Purvis and Hector, 2000), and it has been widely applied as an indicator of the effects of urban growth on biodiversity change (Ditchkoff et al., 2006; McKinney, 2008; Reis et al., 2012; Bregman et al., 2014). However, it appears to be an insufficient indicator by itself, as it sometimes increases when new, human-favored species gradually replace the original species, which is termed the "intermediate disturbance diversity peak" (UNSD, 2014). This suggests that biodiversity cannot simply be characterized by a single measure, such as species richness or abundance, that tells us little about the composition and function of the assemblages that persist in human-disturbed landscapes (Purvis and Hector, 2000). Consequently, the index of change in abundance and the distribution of selected species have been chosen by the Convention on Biological Diversity (CBD, VII/30) to track the process of biodiversity loss (UNSD, 2014).

However, biodiversity accounting, which follows the general form of asset accounting in the System of Environmental-Economic Accounting-Experimental Ecosystem Accounting (SEEA-EEA), might shed new light on the measurement of biodiversity. In fact, the SEEA-EEA has been applied by a wide range of organizations (such as UNEP, OECD and EEA) and in numerous studies (Edens and Hein, 2013; Schröter et al., 2014; Remme et al., 2016). Ecosystem accounting is emerging as a promising approach for organizing comprehensive ecosystem data, tracking changes in ecosystems and linking those changes to economic and other human activities (UNSD, 2014; Hein et al., 2015; Schröter et al., 2015). Combining ecosystem accounts with landscape information could provide important data that are directly linked to economic units and that might shed new light on the mitigation of ecosystem degradation (Haines-Young and Páramo, 2006). Two key points make ecosystem accounting innovative compared to traditional research tools: it incorporates opening stocks and closing stocks, which can show a net change, and it allows the relative species abundance to be compared between different periods, different species, and different ecosystems (UNSD, 2014). Similarly, biodiversity accounting has been proposed as one of the tools for measuring and monitoring human impacts on biodiversity (Cai et al., 2011; Hein et al., 2015). The mean species abundance (MSA) of the original species has been suggested as an indicator of ecosystem accounting, as it excludes exotic or invasive species but their impacts can be examined by the decrease in the abundance of original species being replaced (UNSD, 2014).

Here, we present an initial assessment of the effects of recent habitat change in New Jiangwan Town, which is located in the northeast of the central area of Shanghai City, China, focusing on the response of bird communities and biodiversity to habitat loss and fragmentation triggered by urbanization. New Jiangwan Town was a relatively undisturbed area before 2001, but it became urbanized due to extensive construction since the end of 2003, which makes it a good site for detecting the impacts of rapid urbanization on bird biodiversity. More than ten years of land use/land cover (LULC) data and bird records from 2002 to 2013 were collected. We sought to address the following research questions. 1) How have landscape and habitat structure changed across space and time in our study area? 2) Have the number of species of various species groups and their relative abundance varied with different levels of urbanization? 3) What information does the indicator of MSA provide for demonstrating the process of species replacement and biodiversity change? 4) How have bird communities and thus biodiversity responded to habitat loss and fragmentation?

2. Methods

2.1. Study area

New Jiangwan Town (121°29′12″~121°31′47″E and 31°18′39″~31°20′57″N) is in the northeast of Shanghai City, China, the coastal area of which is in the middle of the Asia-pacific bird migration route, making this triangular area a wildlife shelter of local, national and even international significance. New Jiangwan Town is approximately 6 km from the mouth of the Huangpu River and 10 km from the mouth of the Yangtze River (Fig. 1), and it covers an area of 6.56 km^2.

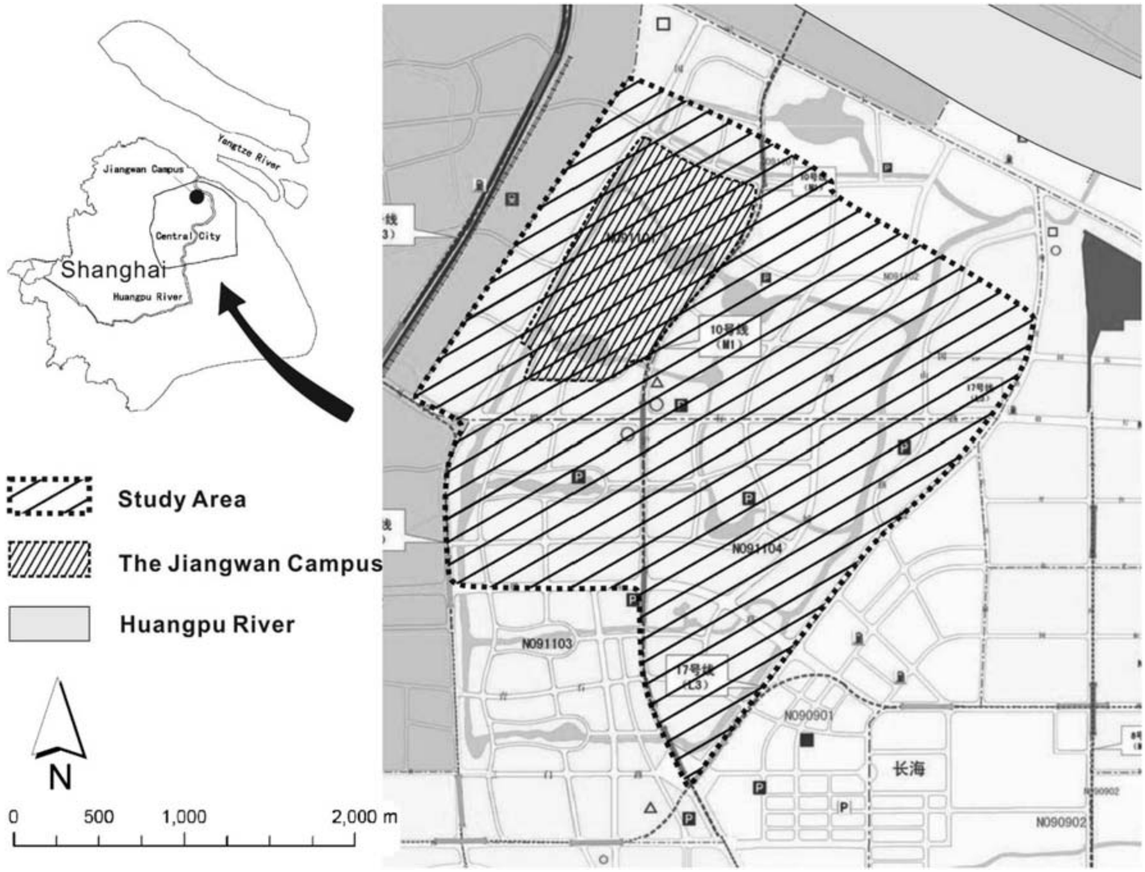


Fig. 1. Location of the study area.

From the 1930s to the 1990s, New Jiangwan Town was a relatively enclosed military airport, and after the airport closed, the landscape remained undisturbed from 1994 to 2001 and provided natural habitat for bird species and other rich flora and fauna that was even better than that of the surrounding parks (Jin et al., 2004). However, at the end of 2003, urban expansion was initiated by the construction of the Jiangwan Campus of Fudan University in the northwest corner of New Jiangwan Town (Fig. 1). The Jiangwan Campus of Fudan University, which has an area of 0.9 km^2, lies in the central area of wetlands in New Jiangwan Town. Continuous bird watching suggests that the composition and number of bird species varied throughout the study period, and *in situ* observations show that avian biodiversity has greatly declined in the coastal areas of Shanghai during recent decades due to rapid urbanization (Cai et al., 2011).

2.2. Data and methods

2.2.1. Land cover and landscape metrics

The land cover information was mainly gathered from historical satellite imagery of New Jiangwan Town from Google Earth, which has been widely used in previous studies (Gulezian and Nyberg, 2010; Pettit et al., 2011), for nine different years: 2000, 2002, 2004, 2005, 2006, 2008, 2009, 2012, and 2013. The land cover maps were rectified by field investigation and comparisons with historical maps, such as the 2002 land use and land cover (LULC) data, obtained from SPOT 5. According to the habitat types of different birds, the land cover datasets were classified into seven types, including built-up land, forest/woodland, shrub land (semi-open areas), grassland (open areas), wetland, water, and bare ground; their detailed characteristics are shown in Table 1. Metrics at the landscape and land cover classification levels were employed to analyze the spatial-temporal patterns of landscape change and were related to the analyses of bird species (Melles et al., 2003; Sundell-Turner and Rodewald, 2008; Wood and Quinn, 2016). Nine landscape metrics were selected, including the proportion of the land type (PLAND), patch density (PD), largest patch index (LPI), edge density (ED), landscape shape index (LSI), Shannon's diversity index

Table 1
Land cover types in New Jiangwan Town.

Land cover type	Detailed description
Built-up area	With no/scattered vegetation (0–10% of total area, excluding biotopes > 100 m^2)
	With sparse vegetation (10–30%)
	With dense vegetation (30–50%)
Forest/woodland	Deciduous forest
	Coniferous forest
	Mixed coniferous and deciduous forest
	Young plantation
Shrub land	Semi-open area, wooded grassland, Boscage: shrub area with high grass
Grassland	Grassland, dry/mesic/moist
Wetland	Wetland with sparse tree cover
	Wetland with dense tree cover
	Wetland with boscage
	Wet Forest
	Grassy shallow water
	Water with Floating vegetation
	Seasonally flooded area
Water	Perennial open water (lakes and streams)
Bare ground	Building yard
	Abandoned area

(SHDI), Shannon's evenness index (SHEI), effective mesh size (MESH), and mean patch size (MPS). The calculation of landscape-level indexes (PD, LPI, ED, LSI, SHDI and SHEI) aimed to give an overview of the variations in landscape patterns. The variations in bird habitats (*i.e.*, wetland, shrub land, and forest land) were measured by class-level metrics (PD, LPI, MESH, MPS) and PLAND of all the types were calculated. The software package FRAGSTATS 4.2(McGarigal et al., 2002) was used to compute the selected metrics (see also in the Appendix A).

2.2.2. Bird data collection and processing

The continuous bird records from 2002 to 2013 were provided by the China Bird Record Center (CBRC), which is the first public and professional NGO for recording birds in China. Bird data were gathered by volunteer citizen-scientists, but each record was under the supervision of a professional ornithologist. The data was collected every 8:00–10:00 am and 4:00–6:00 pm along the line transects. Every line transect was about 500–600 m long, with 50 m wide each side. Field glasses (Kowa 8*BD42) were used during the investigations while walking at a speed 1.0–1.5 km/h, and birds were recorded when seen, heard or flying over. During the line transect investigation the sample point method was also applied to record birds seen or heard within 25 m radius, each point staying 8–10 min. Every report covered 7–10 transects distributed in the north, south, west, east and middle of the site. The line transects were decided by the professional ornithologist in order to represent the bird situation of the total New Jiangwan Town area. Approximately 53 reports of bird observation were obtained in one study area, and more than 4 reports were collected in winter and spring in most years, despite 2010 and 2012 (2 reports in winter and spring). In order to keep the data comparable, the data in winter and spring seasons was used.

Data included the species, number and activities of the birds as well as information about the surrounding environment. Bird species were categorized into different groups with respect to their food types, habitat type and seasonal status according to previous studies (Luan, 2003; Jin et al., 2004; Cai et al., 2011; Zhou et al., 2012). The bird classifications are shown in Table 2, and the bird species lists are shown in Appendix B. The indexes of bird biodiversity, including richness and relative abundance, were calculated, and to avoid masking any change in species, the biodiversity accounting method was applied.

In this study, the natural stock was the bird species, and the opening stock (the reference condition) referred to the condition of the ecosystem under minimal human influence, which was 2002, before the urbanization process began. The mean species abundance (MSA) values of selected species were calculated and compared with the baseline condition:

$$\mathrm{MSA} = \sum_{i=1}^{n} SA_i/n$$

where SA $_i$ is the relative abundance of the ith species which is also among the baseline species, and n refers to the total number of species under the baseline condition.

2.2.3. The Statistical method

The Kolmogorov-Smirnov (K—S) test was used to detect the normality of distribution of the variables for bird species richness and landscape metrics. To meet the requirements for statistical analyses, the raw data of all variables were log-transformed. The PASW Statistic 18.0 was used to calculate the metrics of bird biodiversity, including species richness, relative abundance. Pearson Correlation analysis was applied to examine the relationship and response of bird biodiversity to urbanization process.

Using CANOCO 4.5 (Braak and Smilauer, 2002), a redundancy analysis (RDA) was then employed to investigate how habitat loss and fragmentation explained the variability in bird diversity. Before the RDA, the bird diversity of different food and habitat types were imported into the software to test if the DCA gradient shaft length was less than 3 (Shen et al., 2015). A manual variable selection process was chosen to identify the significant variables at multiple scales based on the results of the Monte Carloper mutation method ($n = 499$).

3. Results

3.1. The changes in landscape pattern from 2000 to 2013

3.1.1. The temporal and spatial variations in land cover types

The temporal and spatial variations in landscape change in New Jiangwan Town from 2000 to 2013 are shown in Fig. 2 and Fig. 3. The results showed that before 2003, more than 50% of the land was covered by wetland (53.9%) or water bodies (6.9%), and the built-up area only occupied approximately 11.3% of the total area. A sharp increase in the built-up area and a decrease in wetland area (to 34.4%) occurred in 2004–2005, especially in the core wetland area. Then the year of 2006–2008 represented a relatively stable phase for all land cover

Table 2
The bird classification criteria and their attributes.

Bird Classification Criteria	Subclassification	Value
Food type	Granivores	GRA
	Insectivores	INS
	Omnivores	OMN
	Frugivores	FRU
	Carnivores	CAR
	Piscivores	PIS
	Benthivores	BEN
Habitat type	Open area species	O1
	Species that prefer open areas but also use forested areas	O2
	Forest species that only use forested areas	F1
	Forest species that also use open areas	F2
	Forest species that use boscage areas	F3
	Swimming birds that use open water	W1
	Waders	W2
	Species that conceal themselves in marshes and aquatic areas with high grass	W3
Seasonal status	Resident	R
	Migratory	M

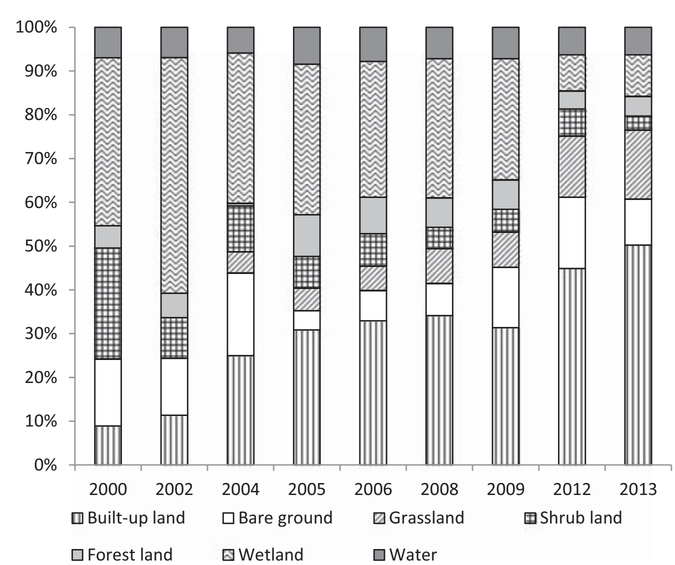


Fig. 2. Proportions (%) of different land cover types (2000−2013).

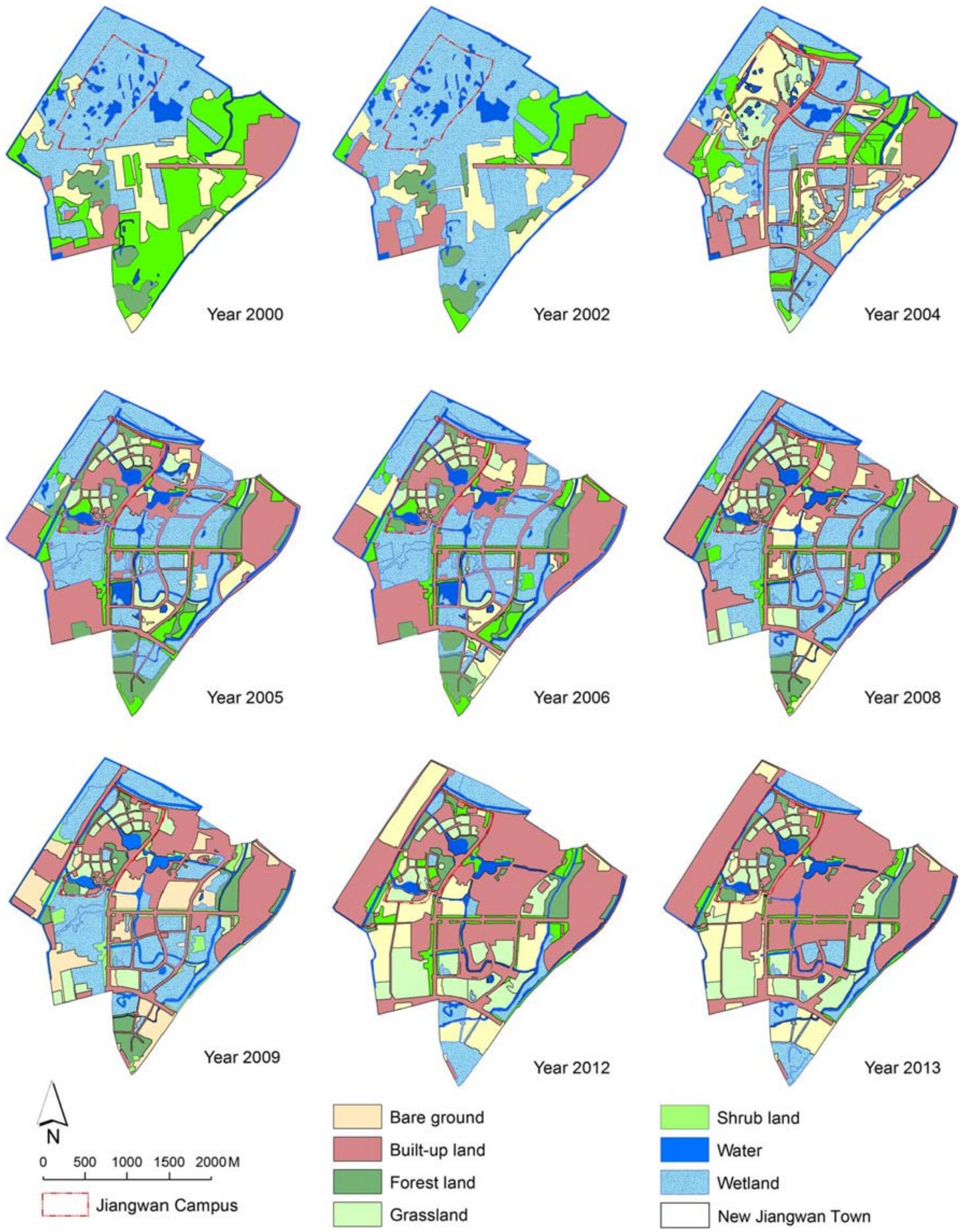


Fig. 3. Spatial distributions of land cover types from 2000 to 2013.

types. However, built-up areas, bare ground and forest land increased during the period of 2009–2013, and by the end of 2013, the proportion of wetland dropped by 9.5%, and the amount of built-up land increased by 50.3%. During the study period, the proportion of grassland increased from 0 to 15.8%, but shrub land decreased from 25.3% to 3.2%.

Accordingly, four stages of urbanization were defined, including the stage before construction (2000 − 2003), the early stage of rapid construction (2004–2005), the middle stage of gradual construction (2006–2008) and the final stage of intensive construction (2009–2013). The spatial distributions of the land cover changes are shown in Fig. 3. During the first stage (2000–2003), the northwest part of the study area was dominated by wetlands and water bodies, and the southeast part was dominated by shrub land and bare ground. During the second stage (2004–2005), the center of the northern wetland area was drained and built up for the Jiangwan Campus, and as the road system was established, the construction of the campus intensified. During the third stage (2006–2008), bare ground increased near the southeast entrance of the campus, and during the final stage (2009–2013), the area around the campus was almost entirely replaced by built-up area, with only a small part of the wetland remaining.

3.1.2. The overview of landscape change as measured by landscape-level metrics

As shown in Fig. 4, the variations in PD and ED appeared as inverted U-shape curves and increased to their highest values during the period of 2004–2008. In 2013, the value of PD decreased to the baseline level, but the value of ED was much higher than that in 2000. Considering the change in the land cover percentages described in the above section, it could be concluded that numerous man-made patches replaced natural patches while patches became fragmented, especially during the period of 2004–2008. The variation in LSI showed a trend similar to that of ED. The lowest LPI value in 2004 suggested some of the largest patches had been converted to urban patches, whereas the highest value in 2013 indicated that the relatively large built-up patches dominated the study area, as shown in Fig. 4. The SHDI and SHEI index values fluctuated during the urbanization process of 2000–2013, appearing as an inverted U-shape curve, but the values of SHDI and SHEI at the landscape level generally ignored the details of the different land cover types by only calculating the overall quantity of landscape diversity. Therefore, the variations in landscape diversity might only indicate the number of activity types that the landscape could support but not whether they were for humans or wildlife.

3.1.3. Habitat loss and fragmentation measured by class-level metrics

As shown in Fig. 5, the most obvious variation occurred in wetlands; from 2000 to 2003, the PD and LPI values of wetlands gently increased and then dramatically declined due to the extensive urban development and construction in 2004. During the period of 2004–2008, the indicator of PD significantly increased and remained high, whereas the LPI remained at a low value, which indicated that the large patches of wetland were fragmented into small ones. The change in the MESH value suggested that patch connectivity declined from 2002 to 2013. As indicated by Fig. 4 and Fig. 5, the loss and fragmentation of wetlands occurred throughout the entire period, so the PD and LPI of wetlands dropped to their lowest levels in the last stage of our study period. In contrast, the increase in the LPI and MPS of built-up land during the entire study period indicated that numerous habitats have been converted to urban area and have become scattered around the large urban patches. For the shrub land and forest land, the decrease in LPI and MPS implied the loss of large patches during the study period. The initial increase and final decrease in the PD values of shrub land and forest land suggested that the large patches were fragmented into small patches at the beginning of the urbanization process, and some small patches even disappeared. Combined with Fig. 2, the increase of grassland in PD, LPI, MPS and MESH was due to the growth of grassland area during the period of 2000–2013.

3.2. Variation in bird species with respect to food type and habitat type

According to the bird records from 2002 to 2013, 53 reports were collected, and approximately 98 species belonging to 31 families were observed. The variations in bird species composition with respect to food type are shown in Fig. 6. And overall, the total number of bird species gradually dropped during our study period. The highest number was 73 in 2003, and the lowest value was 8 in 2010. Omnivores, insectivores and frugivores were the dominant species in terms of bird richness from 2002 and 2013, and omnivores exceeded 50% of the total birds during the 2010–2013 period. The benthic fauna (benthivores) and fish eaters (piscivores) were water birds that generally relied on large areas of wetlands and thus rarely appeared in the city center (Mackinnon et al., 2000). The proportions of benthivores and piscivores decreased from 15% to 6.9%, which might be due to the extensive loss and fragmentation of wetlands under urban construction. The four different food type groups (BEN, PIS, INS and FRU) have temporal-autocorrelations of bird richness. The strength of associations varied from strong (BEN, $R^2 = 0.652$, $p = 0.003$; PIS, $R^2 = 0.787$, $p = 0.000$), to moderately strong (INS, $R^2 = 0.530$, $p = 0.001$; FRU, $R^2 = 0.567$, $p = 0.007$) which approached statistical significance (Fig. 6.). While the OMN ($R^2 = 0.246$, $p = 0.121$) and CAR ($R^2 = 0.302$, $p = 0.08$) did not respond significantly.

The variations in bird species richness according to habitat type are shown in Fig. 7, and it can be seen that birds inhabiting wetlands rapidly decreased during the urbanization process, especially birds inhabiting W2 and W3. Birds inhabiting forest fluctuated but remained slightly decreased, and birds inhabiting F1 decreased to 0 in 2009. Additionally, birds inhabiting open areas greatly decreased during the study period, especially birds living in O2. In general, birds preferring wetland declined fastest at an annual rate of decrease of 7.7% followed by species preferring open areas at an annual rate of decrease of 4.3% and

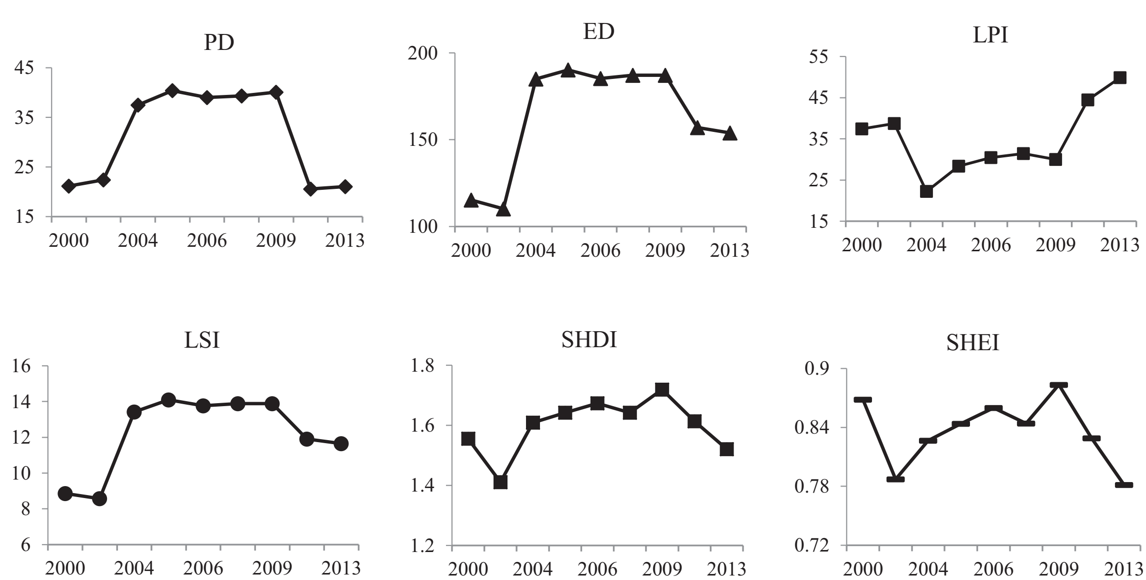


Fig. 4. Landscape change as measured by landscape-level metrics from 2000 to 2013.

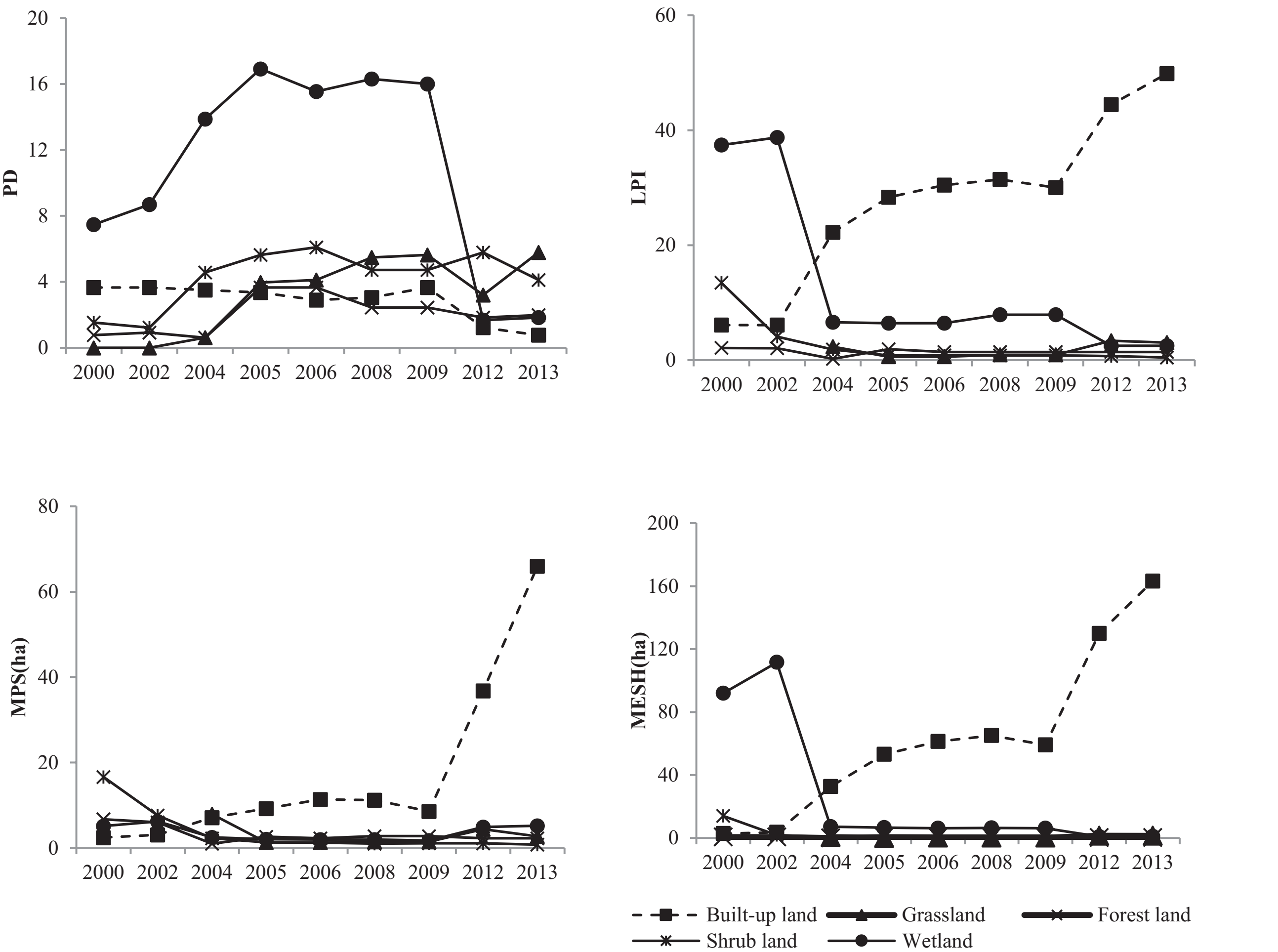


Fig. 5. Habitat loss and fragmentation as measured by class-level metrics from 2000 to 2013.

those preferring forest at 3.2%. The bird composition respect to habitat requirements distinctly reflected that birds inhabiting in wetlands (W) decreased more sharply than in forest land (F) and shrub/grassland (O). And the W and O had temporal autocorrelation responded statistical significance while the forest birds ($R^2 = 0.325$, p value $= 0.067$) did not. The strength of association varied from strong

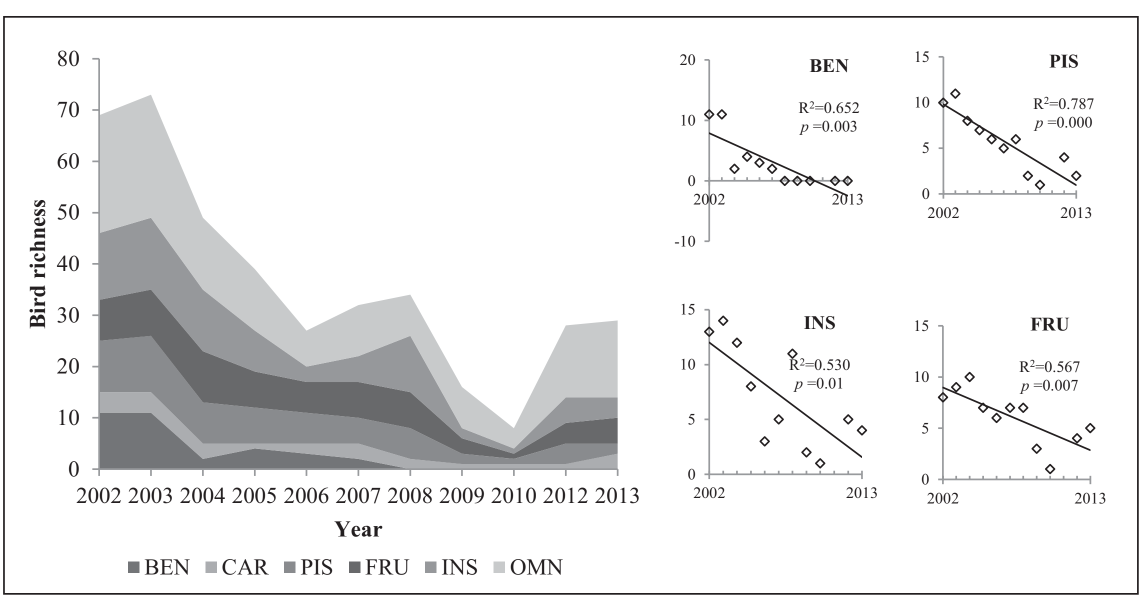


Fig. 6. Bird species richness categorized by food type from 2000 to 2013.

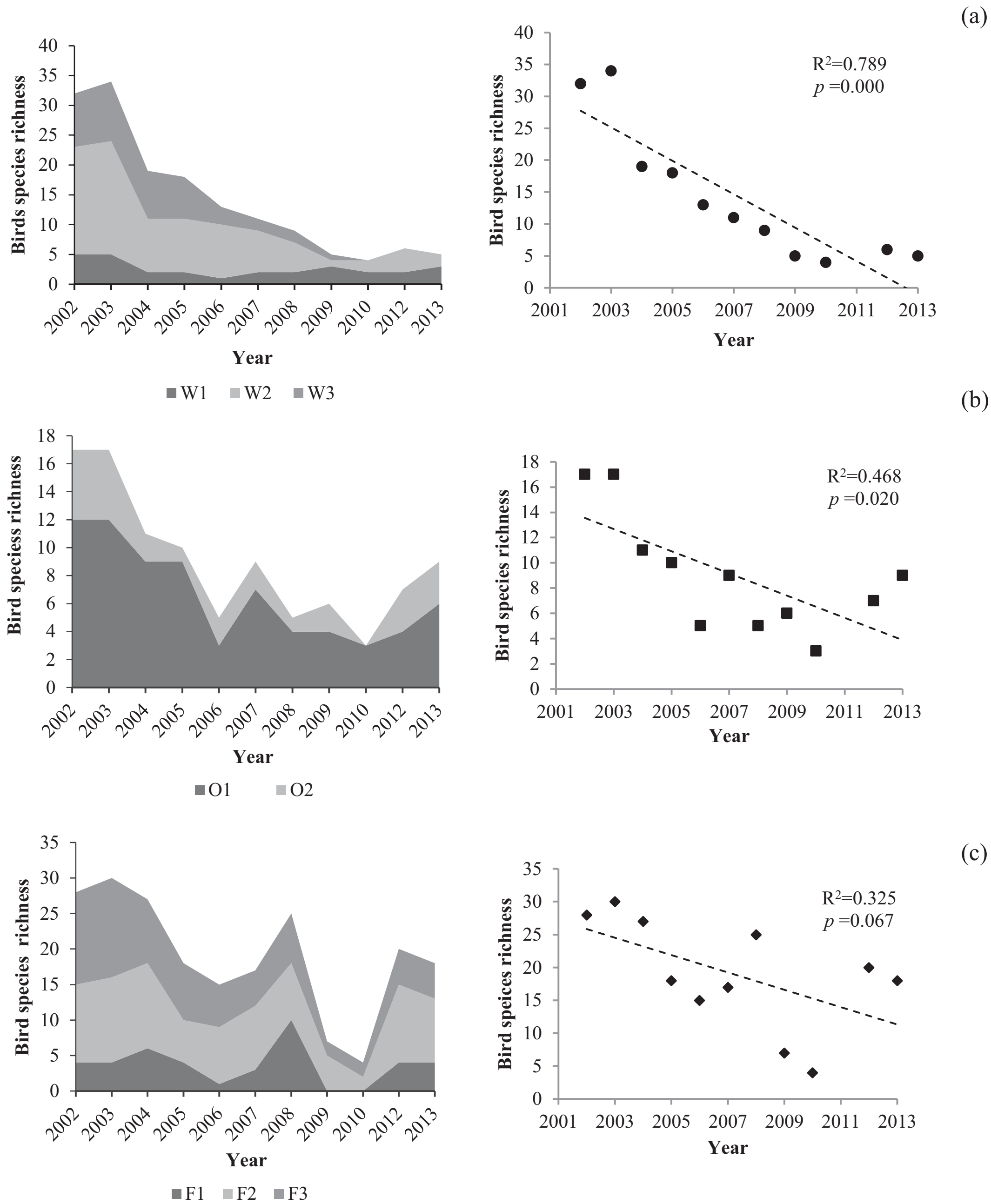


Fig. 7. Bird species richness with respect to habitat type: a) wetland (W); b) shrub/grass land (O); and c) forest land (F) from 2000 to 2013.

(W, $R^2 = 0.789$, $p = 0.000$) to moderate (O, $R^2 = 0.468$, $p = 0.020$). However, the change in the richness of species in different categories may ignore the process of species replacement, such as the turnover from original species to invasive species.

3.3. The dynamics of bird biodiversity from 2002 to 2013

3.3.1. Variation in bird richness and relative abundance

As shown in Fig. 8, the values of bird richness and relative abundance fluctuated but declined during the 2000–2013 study period. The bird richness ($R^2 = 0.567$, $p = 0.007$) was temporal autocorrelation which responded significance while the relative abundance ($R^2 = 0.205$, $p = 0.162$) did not. It should be noted that bird relative abundance peaked in 2005, which might have resulted from the increase in some urban-adaptive birds, such as Eurasian Tree Sparrow and Vinous-throated Parrotbill, as the damage to the wetlands during the process of intensive construction made the food resources available to these birds much richer than before. However, relative abundance quickly declined and never recovered. Generally, with an increase in human disturbance, many endemic species would decrease in abundance, while some species, those that benefit from disturbed habitats in particular, would increase. In our study, the endemic species were gradually replaced by species, either endemic or exotic, favored by human influences. The variation in species richness could imply an overall trend of biodiversity loss in the study area, but the increase in species richness during the 2010–2013 period might have been caused by the significant increase in invasive birds, which will be discussed further in the following analyses. Therefore, the two indexes of richness and relative abundance

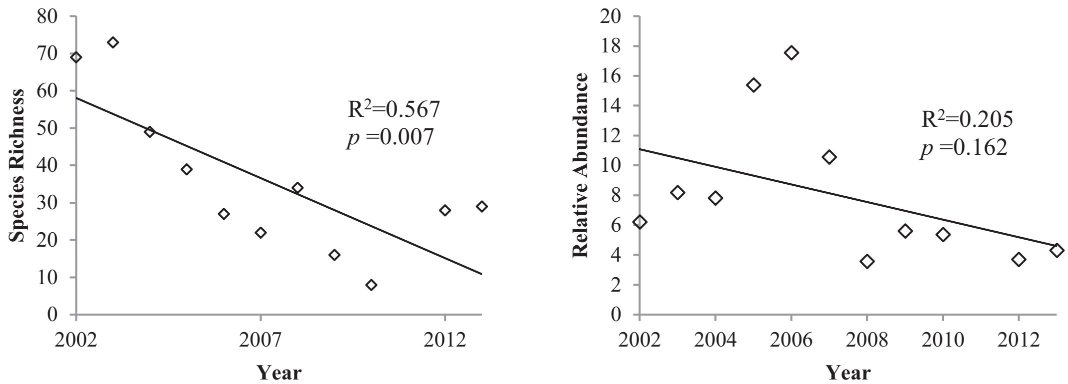


Fig. 8. Variations in bird species richness and relative abundance from 2000 to 2013.

seemed to hardly make clear what happened to species composition and structure under rapid urbanization as bird species were evaluated equally whether they were endemic or invasive. Therefore, these two indexes might not fully account for the process of biodiversity loss and ecosystem degradation.

3.3.2. The process of bird replacement and the change in MSA

According to the four urbanization stages mentioned above, four corresponding stages were defined for biodiversity accounting. The detailed information on bird species in the four stages and the species moving in and out during each stage are shown in Appendix C. During the second (2004–2005), third (2006–2009) and forth (2010−2013) stages, 35, 14, and 23 bird species moved out, while 9,19and 11speices moved in. Considering the detailed information in Appendix D, it can be seen that the urban avoiders moved out and the urban-adapted birds moved into the study area. During the second stage of 2004–2005, the original water birds, such as Mallards, Spot-billed Ducks and Philippine Ducks, were gone, as were the shorebirds, such as the Common Greenshank, Wood Sandpiper, Temminck's Stint, and Pacific Golden Plover. These birds are urban avoiders and could rarely be seen in the city center, only existing in the rural and coastal regions. Meanwhile, more urban-adapted birds (*e.g.*, the Egrets, Eurasian Blackbirds and Sparrows) moved into the study area, which might have led to the peak relative abundance value in 2005 (Fig.8.). During the third stage of 2006–2008, the remaining shorebirds (*e.g.*, the Green Sandpiper and Common Sandpiper) continued to decrease along with the birds using the near-water vegetation (*e.g.*, the Pied Kingfisher, White-breasted Water hen and Yellow Bittern). Simultaneously, birds inhabiting grassland and forest land increased including several kinds of thrushes and buntings. During the last stage, 23 bird species moved out, and even the various Egrets, small Shorebirds, Swallows, Thrushes and Buntings became extinct, and the bird species composition became much more similar to that of a residential area or urban center.

Furthermore, the MSA was calculated for the four different stages (see Appendix D for more detailed information).As shown in Fig. 9, the MSA values of the original species decreased from 8.28 to 6.36 to 4.32 to 2.00 individuals per survey, respectively. Compared with the original/reference species, the bird biodiversity in the second, third and fourth stages were approximately 76.8%, 52.2% and 24.5% of the baseline condition.

3.4. Effects of habitat loss and fragmentation on bird diversity

Pearson correlation analysis was conducted between biodiversity indexes and PLAND of different types. The richness, birds of BEN, M, R, F3, W2, W3 and W are related in positive linear sense with PLAND of shrub land and wetland but negative with PLAND of built-up area and grassland which approached statistical significance (Table 3). The conclusion could be drawn that the habitat loss of New Jiangwan Town was significantly caused by the increasing of the proportion of built-up area and grassland and the decreasing of shrub land and wetland (Fig.2 and Fig. 3).

As the PLAND of water bodies and bare ground did not vary substantially and grasslands were mainly artificial showing negative correlation with bird richness during the entire study period, their variations have not been included into the RDA. In addition to habitat loss, habitat fragmentation was another important factor that influenced bird diversity in this study. Habitat fragmentation generally includes the following processes (Scolozzi and Geneletti, 2012): 1) a reduction in the total habitat area (measured by PLAND of different habitat types), 2) an increase in the number of habitat patches (measured by PD), 3) an increasing in the isolation of patches (measured by MESH), and 4) the breaking up of one patch of habitat into several smaller patches (measured by LPI and MPS). As shown in Table 4, patch size and connectivity of habitats significantly affected bird biodiversity from the RDA. Loss and fragmentation of wetland, forest land and shrub land could explain more than 80% of the bird diversity variation, with largest explanatory ability of 92.4% for the habitat type in the full model.

However, various bird communities with respect to habitat type, food type and seasonal status were influenced to different degrees. In contrast, wetland area and connectivity was the most significant variable influencing the bird communities followed by shrub land and forest land. The indicator of MESH_W could significantly explain almost half of bird communities' variation, and thus biodiversity, as shown in Fig. 10. The wetland bird groups (BEN, PIS, W2 and W3), which represent the original birds, were significantly negatively affected by loss and fragmentation of wetland and forest land. The birds that preferred O1, O2 were also affected by wetland patch size, as wetland was usually a type of open area, and the boscage area was easily influenced by the wetland. The MESH for shrub land (MESH_S) was tightly correlated with the open-area birds (O1, O2 and W1). With respect to seasonal status, all habitat types were contributors to the variation of bird species. Overall, it might be concluded that a large patch area and well-connected wetlands were the most important habitats for bird biodiversity conservation in our study area.

4. Discussion

4.1. How to measure the dynamic variations in bird diversity due to urbanization

Urban sprawl is currently a worldwide phenomenon, especially in developing countries, concentrating around urban centre and replacing

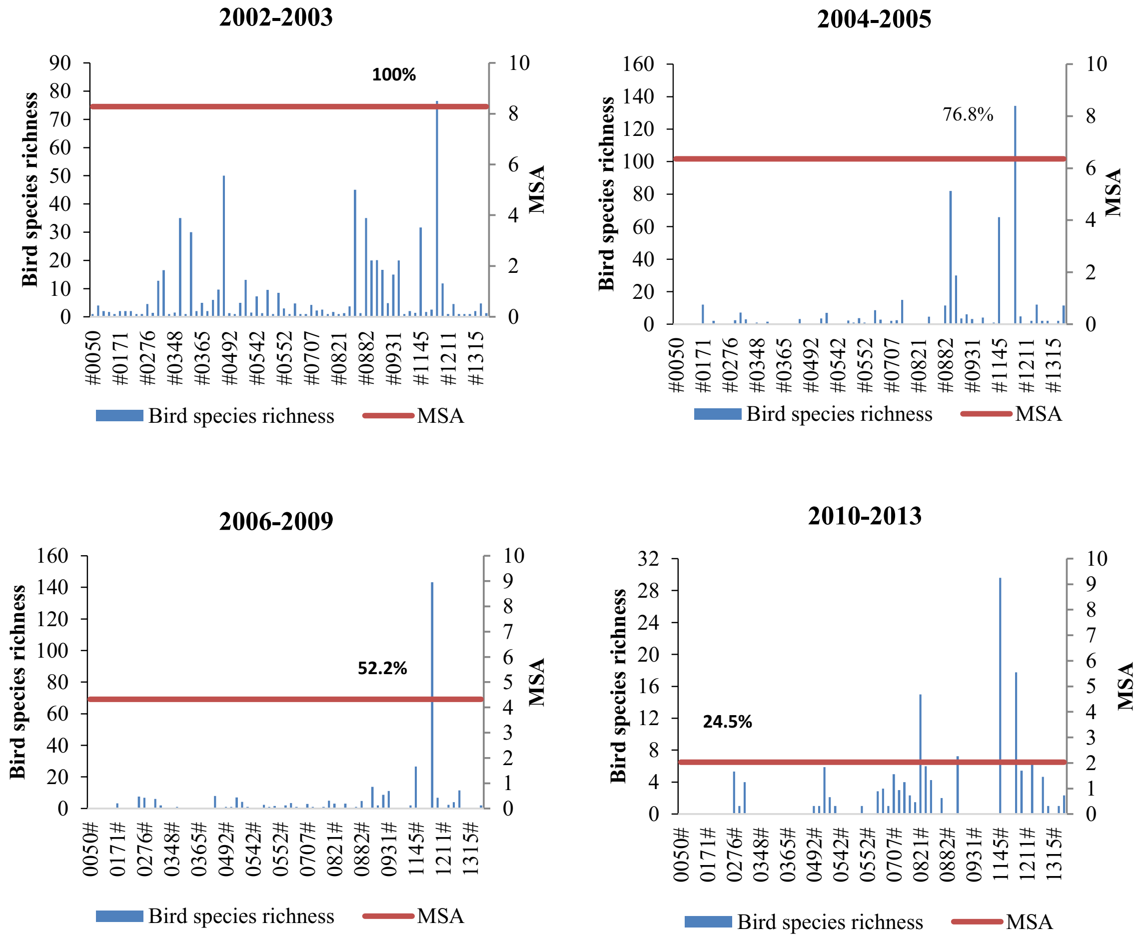


Fig. 9. Value of the MSA indicator at different urbanization stages (the reference species were numbered from #0050 to #1322).

adjacent land cover types such as agriculture and more natural vegetation (McGranahan and Satterthwaite, 2003; Pauchard et al., 2006) that generally dramatically changed ecological conditions (Tomasevic and Marzluff, 2017). Bird species are the most important indicators of ecosystem change. In undisturbed ecosystems, bird communities are often dominated by urban-avoiders, such as aquatic species that

Table 3
The correlation between bird diversity and the proportion of landscape types.

Pearson Correlation	PLAND						
Sig. (2-tailed)	Built-up area	Bare ground	Grassland	Shrub land	Forest	Wetland	Water
Richness	−**0.833****	0.285	−**0.757***	**0.713***	−0.260	**0.720***	−0.118
Abundance	−0.165	−0.546	−0.386	0.013	0.562	0.347	**0.713***
BEN	−**0.880****	0.120	−**0.834****	**0.756***	0.053	**0.759***	0.167
CAR	−0.610	0.339	−0.508	0.588	−0.434	0.495	−0.379
PIS	−**0.873****	0.123	−**0.880****	**0.669***	−0.086	**0.838****	0.109
FRU	−**0.670***	0.152	−**0.697***	0.475	−0.332	**0.709***	−0.106
INS	−**0.741***	0.245	−**0.685***	0.571	−0.322	**0.697***	−0.160
OMN	−0.611	0.442	−0.455	0.632	−0.365	0.403	−0.318
M	−**0.823****	0.277	−**0.729***	**0.706***	−0.216	**0.698***	−0.121
R	−**0.789***	0.277	−**0.748***	**0.673***	−0.311	**0.704***	−0.106
F1	0.001	−0.053	0.014	−0.011	−0.269	0.080	−0.233
F2	−0.352	0.647	−0.227	0.434	−**0.739***	0.177	−**0.677***
F3	−**0.822****	0.165	−**0.780***	**0.688***	−0.165	**0.748***	−0.016
F	−0.556	0.292	−0.485	0.510	−0.450	0.486	−0.335
O1	−**0.773***	0.273	−**0.696***	**0.675***	−0.231	0.652	−0.062
O2	−0.544	0.498	−0.371	0.612	−0.317	0.289	−0.353
O	−**0.767***	0.369	−0.652	**0.714***	−0.279	0.592	−0.161
W1	−0.661	0.360	−0.485	0.589	−0.172	0.468	−0.189
W2	−**0.885****	0.131	−**0.866****	**0.739***	−0.024	**0.796***	0.127
W3	−**0.891****	0.167	−**0.897****	0.662	−0.113	**0.853****	0.122
W	−**0.918****	0.181	−**0.887****	**0.741***	−0.076	**0.829****	0.094

* Correlation is significant at the 0.05 level (2-tailed).
** Correlation is significant at the 0.01 level (2-tailed).

Table 4
Redundancy analysis using the bird diversity and metrics of habitat loss and fragmentation.

	Significant variables	Explained variance (%)	Proportion of total explained variance (%)	*p*-value
Food type	MESH_W	0.459(0.870)	52.8	0.002
	PLAND_W	0.117(0.870)	13.4	
	PD_W	0.107(0.870)	12.3	
	MESH_S	0.132(0.870)	15.1	
	PD_S	0.055(0.870)	0.06	
Habitat type	MESH_W	0.427(0.924)	46.3	0.002
	PLAND_W	0.129(0.924)	14.0	
	PD_W	0.119(0.924)	12.9	
	MESH_S	0.107(0.924)	11.6	
	MPS_F	0.071(0.924)	0.08	
	MPS_W	0.070(0.924)	0.08	
Seasonal status	MESH_W	0.547(0.818)	66.9	0.002
	MPS_F	0.099(0.818)	12.1	
	PLAND_S	0.084(0.818)	10.3	
	LPI_F	0.043(0.818)	0.05	
	PD_S	0.045(0.818)	0.06	

generally feed on fishes, plants or benthonic animals (Palomino and Carrascal, 2006; Matsuba et al., 2016), but in urbanized areas, bird communities are often dominated by urban-tolerant species, such as forest guilds like insectivores, granivores and omnivores (Conole and Kirkpatrick, 2011; Reis et al., 2012).

Species richness and abundance are the two main indicators used for measuring biodiversity, and they have been widely used to be correlated with urban sprawl (MacGregor-Fors, 2008; Yuan et al., 2014; Batáry et al., 2017). However, in our study, we found that species richness alone appeared to be an insufficient indicator of the real status of bird biodiversity, as it mainly represents the number of species while ignoring the process of bird species replacement, especially the original species by urban adaptors. The indicator of MSA of the original species could provide supplemental information by using the original species as a reference condition. Therefore, the combination of different indicators (species richness, relative abundance and MSA) could account for the overall trend of the change in biodiversity, the process of species replacement and the dynamics of biodiversity loss under different levels of urbanization.

4.2. How wetland degradation influences bird biodiversity in urban area?

Wetlands in metropolitan areas are typically the most biologically diverse ecosystems, but they face the greatest degree of degradation, such as through fragmentation and invasion by non-native species as well as destruction (Pauchard et al., 2006; Quesnelle et al., 2013; Yuan et al., 2014). This study detected the dependence of bird biodiversity

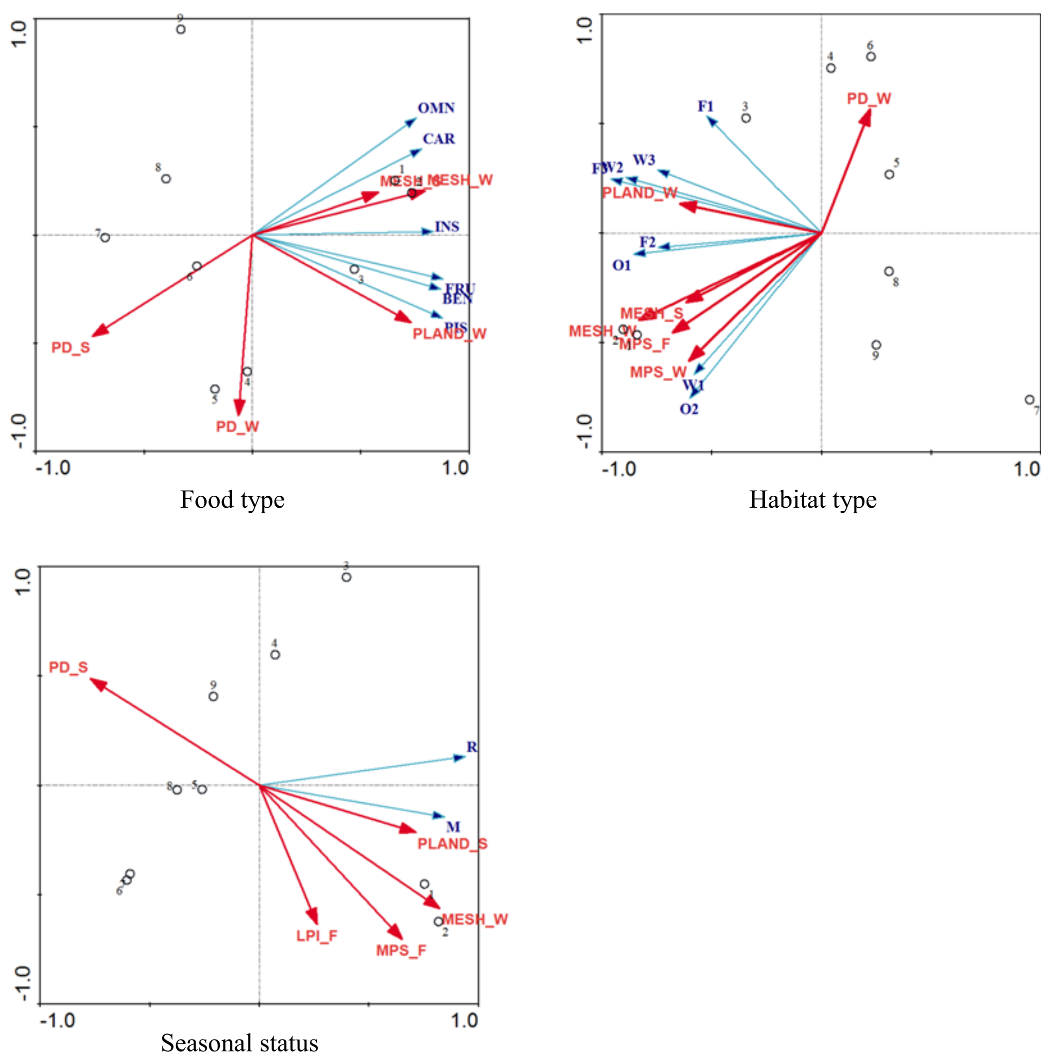


Fig. 10. Redundancy analysis biplots showing correlation between habitat loss and fragmentation and bird diversity.

on habitat conditions, especially in wetlands, so wetland birds are the dominant indicator species for biodiversity conservation, especially at the beginning of human disturbance. Other studies have reached the same conclusion and have noted the distinctiveness of original bird species as indicators because of their specific nesting and habitat requirements as well as the intensity of disturbance to wetlands by human activities (Conole and Kirkpatrick, 2011). In this study, the wetland birds, such as the benthic fauna eaters or fishing birds, rapidly declined at the beginning of the urbanization process, and this phenomenon has been discussed by many other researchers, who suggest that there are critical differences in bird composition between non-urbanized and urbanized areas (Conole and Kirkpatrick, 2011; Reis et al., 2012; Zhou et al., 2012). However, our study area, New Jiangwan Town, was a relatively small and isolated area for wildlife, so broader scales should be considered in further studies, including the three main areas with high bird concentrations on the mouth of the Huangpu River: Gongqing Forest Park, Paotaiwan Wetland Park and Binjiang Forest Park.

4.3. Implications for bird conservation and habitat restoration

Habitat loss and fragmentation have been widely discussed in landscape and ecology studies, and the implementation of these research findings in landscape planning and habitat restoration would contribute to decreasing biodiversity loss and maintaining ecosystem function in increasingly fragmented landscapes (Bregman et al., 2014). By considering different levels of urbanization, we attempted to propose different biodiversity conservation strategies with this study. In the first two stages, which were dominated by natural or semi-natural habitats, conserving the entire area was suggested, and ecological zoning tools could have been applied to prevent habitats from further loss and fragmentation (Löfvenhaft et al., 2004). In the third and fourth stages, temporal conservation could have been implemented (Kattwinkel et al., 2011) to protect habitats, and temporal and transparent planning tools could have been used to increase political awareness to save as many biotopes as possible (Bekessy et al., 2012). In the last stage, habitat reconstruction should have been adopted when the landscape was dominated by built-up area.

Grassland in this study, artificial with intense maintenance, increasing along the built-up areas, turned out to be negative correlation with bird species which implicated that the original and natural habitats with self-maintenance, such as original wetlands, shrub lands could be alternatives to grasslands and lawns in the urbanization process from the perspectives of landscaping. The bird diversity respect to food type and habitat requirement has statistic significant relations with the PLAND of habitats which might also give us suggestions for bird conservation from the landscape level.

The MSA indicator might be used as a reference for determining the different levels of ecosystem degradation. For example, if the indicator is 100%, the species or habitat has a natural or low-impact status, but if the indicator is 50%, the average abundance of the original species is 50% of the natural or low-impact state (UNSD, 2014). However, the question as to what strategies should be adopted if the indicator is 80%, 50% or even lower must be further examined. Additionally, the MSA could be easily compared among different stages and even among different study sites, which might provide significant information to aid decision makers in choosing priority sites for restoration and conservation.

5. Conclusions

Bird diversity is sensitive to environmental change, especially habitat change caused by urban growth. In this study, the influence of habitat loss and fragmentation caused by landscape change from being undisturbed to human-dominated on bird diversity was examined. From our research, we concluded that 1) spatio-temporal variations in habitat loss and fragmentation were triggered by urbanization; 2) the number of species and relative abundance of various species groups varied with different levels of urbanization; 3) when the landscape changed from being undisturbed to human-dominated, the urban avoiders moved out and the urban-adapted birds moved into the study area; 4) the combination of species richness, relative abundance and MSA accounted for the overall trend in the change in biodiversity, the process of species replacement (such as the change from the original species to the invasive species) and the dynamics of biodiversity loss at different levels of urbanization; 5) the impacts of habitat loss and fragmentation on bird species differed among bird communities, and the wetland percent cover and patch size were the most significant factors; 6) with the help of biodiversity accounting, a comprehensive overview of the biodiversity of the ecosystem in response to landscape change could be presented, and the detailed species characteristics could be useful for future habitat restoration under various degradation scenarios in an increasingly urban world.

Acknowledgments

This research was financially supported by the National Key Research and Development Program of China (No. 2016YFC0502705), the National Natural Science Foundation of China (No. 41501194), the Social Science Major Programme Foundation of China (14ZDB140), and the Excellent Scholars Support Plan on Science and Technology of Guizhou Province (KY[2014]252). We acknowledge the China Bird Record Center, the first public and professional NGO for bird records, which provided a large amount of bird census data that were gathered. Thanks to the great efforts of volunteer citizen-scientists. Thanks to Yanxia Zhao for providing the open-source bird data. Finally, we are grateful to those who participated in the bird recording activities for more than ten years and who largely contributed to this research anonymously.

Appendix A. The landscape metrics measured in this paper were listed as below, which were calculated by the software Fragstats 4.2 (McGarigal K. 2015)

Abbr.	Metric name	Unit	Application scale
PLAND	Proportion of the land type	%	Class level
PD	Patch density	#/km^2	Landscape/class level
LPI	Largest patch index	%	Landscape/class level
ED	Edge density	m/ha	Landscape level
LSI	Landscape shape	none	Landscape level
SHDI	Shannon's diversity index		Landscape level
SHEI	Shannon's evenness index		Landscape level
MESH	Effective mesh size	ha	Class level
MPS	Mean patch size	ha	Class level

Appendix B. Bird species observed in and around Jiangwan Campus in 2002–2013, with respect to food type, seasonal status, and habitat type

ID	Family	Name	Species	Food	Seasonal	Habitat
#0014	*Phasianidae*	Japanese quail	*Coturnix japonica*	OMN	R	O1
#0050	*Phasianidae*	Common pheasant	*Phasianuscolchicus*	OMN	R	F3
#0084	*Anatidae*	Mandarin duck	*Aix galericulata*	OMN	M	W1-F2
#0089	*Anatidae*	Mallard	*Anasplatyrhynchos*	OMN	M	W1
#0090	*Anatidae*	Spot-billed duck	*Anaspoecilorhyncha*	OMN	M	W1
#0096	*Anatidae*	Philippine duck	*Anasluzonica*	OMN	M	W1
#0169	*Coraciidae*	Dollarbird	*Eurystomusorientalis*	INS	M/R	F2
#0171	*Alcedinidae*	Common kingfisher	*Alcedoatthis*	PIS	R	W3
#0180	*Cerylidae*	Pied kingfisher	*Cerylerudis*	PIS	R	W3
#0193	*Cuculidae*	Eurasian cuckoo	*Cuculuscanorus*	INS	M	F2-W3
#0195	*Cuculidae*	Lesser cuckoo	*Cuculuspoliocephalus*	INS	M	F2
#0204	*Centropodidae*	Lesser coucal	*Centropusbengalensis*	INS	R	F3-W3
#0221	*Apodidae*	Fork-tailed swift	*Apuspacificus*	INS	M	O1
#0274	*Columbidae*	Oriental turtle dove	*Streptopeliaorientalis*	FRU	R	O2
#0276	*Columbidae*	Spotted dove	*Streptopeliachinensis*	FRU	R	O2
#0313	*Rallidae*	White-breasted waterhen	*Amaurornisphoenicurus*	OMN	R	W3
#0323	*Rallidae*	Common moorhen	*Gallinulachioropus*	OMN	R	W1-O1
#0324	*Rallidae*	Common coot	*Fulicaatra*	OMN	R	W1
#0334	*Scolopacidae*	Common snipe	*Gallinagogallinago*	BEN	M	W3
#0345	*Scolopacidae*	Common greenshank	*Tringanebularia*	BEN	M	W2
#0348	*Scolopacidae*	Green sandpiper	*Tringaochropus*	BEN	M	W2
#0349	*Scolopacidae*	Wood sandpiper	*Tringaglareola*	BEN	M	W2
#0351	*Scolopacidae*	Common sandpiper	*Actitishypoleucos*	BEN	M	W2
#0363	*Scolopacidae*	Rufous-necked stint	*Calidrisruficollis*	BEN	M	W2
#0364	*Scolopacidae*	Temminck's stint	*Calidristemminckii*	BEN	M	W2
#0365	*Scolopacidae*	Long-toed stint	*Calidrissubminuta*	BEN	M	W2
#0380	*Jacanidae*	Pheasant-tailed jacana	*Hydrophasianuschirurgus*	OMN	M	W3
#0388	*Charadriidae*	Pacific golden plover	*Pluvialisfulva*	BEN	M	W2
#0392	*Charadriidae*	Little ringed plover	*Charadriusdubius*	BEN	M	W2
#0393	*Charadriidae*	Kentish plover	*Charadriusalexandrinus*	BEN	M	W2
#0492	*Accipitridae*	Common buzzard	*Buteobuteo*	CAR	M	O1
#0508	*Falconidae*	Common kestrel	*Falco tinnunculus*	CAR	R	O1
#0518	*Podicipedidae*	Little grebe	*Tachybaptusruficollis*	PIS	R	W1
#0535	*Ardeidae*	Little egret	*Egrettagarzetta*	PIS	R	W2-F2
#0539	*Ardeidae*	Grey heron	*Ardeacinerea*	PIS	R	W2-F2
#0542	*Ardeidae*	Large egret	*Casmerodiusalbus*	PIS	R	W2
#0543	*Ardeidae*	Intermediate egret	*Mesophoyxintermedia*	PIS	R	W2
#0544	*Ardeidae*	Cattle egret	*Bubulcus ibis*	PIS	R	W2
#0545	*Ardeidae*	Chinese Pond-heron	*Ardeolabacchus*	PIS	R	W2
#0546	*Ardeidae*	Striated heron	*Butoridesstriatus*	PIS	R	W2-W3
#0547	*Ardeidae*	Black-crowned night-heron	*Nycticoraxnycticorax*	PIS	R	W2-F2
#0552	*Ardeidae*	Yellow bittern	*Ixobrychussinensis*	PIS	R	W3
#0614	*Laniidae*	Brown shrike	*Laniuscristatus*	CAR	M	F3
#0616	*Laniidae*	Long-tailed shrike	*Laniusschach*	CAR	R	F3
#0630	*Corvidae*	Azure-winged magpie	*Cyanopicacyana*	OMN	R	F2
#0636	*Corvidae*	Black-billed magpie	*Pica pica*	OMN	R	F2
#0691	*Muscicapidae*	White-throated rock thrush	*Monticolagularis*	OMN	M	F1
#0700	*Muscicapidae*	Scaly thrush	*Zootheradauma*	OMN	M	F1
#0702	*Muscicapidae*	Grey-backed thrush	*Turdushortulorum*	OMN	M	F1
#0707	*Muscicapidae*	Eurasian blackbird	*Turdusmerula*	OMN	R	F2
#0712	*Muscicapidae*	Eyebrowed thrush	*Turdusobscurus*	OMN	M	F1
#0714	*Muscicapidae*	Pale thrush	*Turduspallidus*	OMN	M	F1
#0717	*Muscicapidae*	Dusky thrush	*Turdusnaumanni*	OMN	M	O1
#0730	*Muscicapidae*	Asian brown flycatcher	*Muscicapadauurica*	INS	M	F1
#0734	*Muscicapidae*	Narcissus flycatcher	*Ficedulanarcissina*	INS	M	F1
#0735	*Muscicapidae*	Mugimaki flycatcher	*Ficedulamugimaki*	INS	M	F2
#0745	*Muscicapidae*	Blue-and-white flycatcher	*Cyanopticacyanomelana*	INS	M	F1
#0767	*Muscicapidae*	Bluethroat	*Lusciniasvecica*	INS	M	F3
#0773	*Muscicapidae*	Orange-flanked bush-robin	*Tarsigercyanurus*	INS	M	F1
#0787	*Muscicapidae*	Daurian redstart	*Phoenicurusauroreus*	INS	R	F3
#0821	*Sturnidae*	White-cheeked starling	*Sturnuscineraceus*	OMN	M	O2
#0823	*Sturnidae*	Black-collared starling	*Sturnusnigricollis*	OMN	R	O2
#0829	*Sturnidae*	Crested myna	*Acridotherescristatellus*	OMN	R	O1
#0850	*Remizidae*	Chinese penduline tit	*Remizconsobrinus*	OMN	M	W3
#0862	*Paridae*	Great tit	*Parus major*	OMN	R	F2
#0882	*Hirundinidae*	Barn swallow	*Hirundorustica*	INS	M	O1
#0884	*Hirundinidae*	Red-rumped swallow	*Hirundodaurica*	INS	M	O1
#0898	*Pycnonotidae*	Light-vented bulbul	*Pycnonotussinensis*	FRU	R	F2
#0913	*Cisticolidae*	Zitting cisticola	*Cisticolajuncidis*	FRU	R	O1
#0922	*Cisticolidae*	Plain prinia	*Priniainornata*	FRU	R	W3
#0931	*Sylviidae*	Manchurian bush warbler	*Cettiacanturians*	INS	M	F3
#0932	*Sylviidae*	Japanese bush-warbler	*Cettiadiphone*	INS	M	F3

(continued on next page)

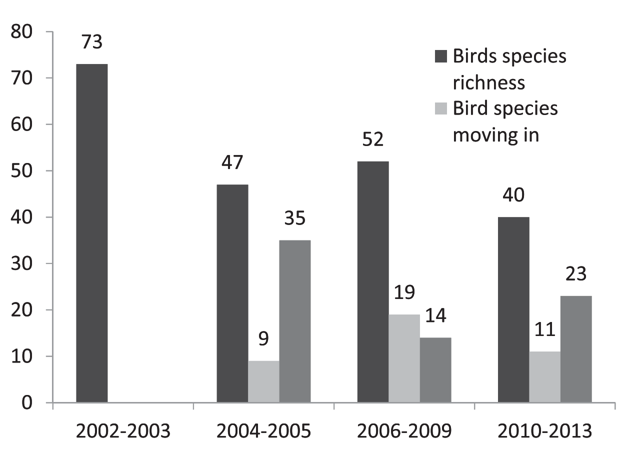
(*continued*)

ID	Family	Name	Species	Food	Seasonal	Habitat
#0933	*Sylviidae*	Brownish-flanked bush-war	*Cettiafortipes*	INS	R	F3
#0961	*Sylviidae*	Oriental reed warbler	*Acrocephalusorientalis*	INS	M	W3
#0984	*Sylviidae*	Yellow-rumped warbler	*Phylloscopusproregulus*	INS	M	F3
#0988	*Sylviidae*	Yellow-browed warbler	*Phylloscopusinornatus*	INS	M	F1
#0993	*Sylviidae*	Pale-legged warbler	*Phylloscopustenellipes*	INS	M	F1-F3
#0995	*Sylviidae*	Eastern crowned warbler	*Phylloscopuscoronatus*	INS	M	F1
#1013	*Sylviidae*	Masked laughingthrush	*Garrulaxperspicillatus*	INS	R	F3
#1017	*Sylviidae*	Greater necklaced laughingthrush	*Garrulaxpectoralis*	INS	R	F1
#1036	*Sylviidae*	Hwamei	*Garrulaxcanorus*	INS	R	F3
#1145	*Paradoxornis*	Vinous-throated parrotbill	*Paradoxorniswebbianus*	INS	R	F3
#1172	*Alaudidae*	Eurasian skylark	*Alaudaarvensis*	OMN	M	O1
#1174	*Alaudidae*	Oriental skylark	*Alaudagulgula*	OMN	M	O1
#1198	*Passeridae*	Eurasian tree sparrow	*Passer montanus*	OMN	R	F2
#1207	*Passeridae*	White wagtail	*Motacilla alba*	OMN	R	O1-W2
#1211	*Passeridae*	Yellow wagtail	*Motacillaflava*	OMN	M	O1-W2
#1212	*Passeridae*	Grey wagtail	*Motacillacinerea*	OMN	M	O1-W2
#1218	*Passeridae*	Orienfnl tree pipit	*Anthushodgsoni*	OMN	M	F2
#1239	*Passeridae*	White-rumped munia	*Lonchurastriata*	OMN	R	F3
#1246	*Fringillidae*	Grey-capped greenfinch	*Carduelissinica*	FRU	R	F2
#1293	*Fringillidae*	Yellow-billed grosbeak	*Eophonamigratoria*	FRU	M	F2
#1307	*Fringillidae*	Meadow bunting	*Emberizacioides*	FRU	R	F2
#1311	*Fringillidae*	Tristram's bunting	*Emberizatristrami*	FRU	M	F3
#1314	*Fringillidae*	Yellow-browed bunting	*Emberizachrysophrys*	FRU	M	F3
#1315	*Fringillidae*	Rustic bunting	*Emberizarustica*	FRU	M	F3
#1316	*Fringillidae*	Yellow-throated bunting	*Emberizaelegans*	FRU	R	F1
#1322	*Fringillidae*	Black-faced bunting	*Emberizaspodocephala*	FRU	M	F3

Resident status: R = resident, M = migratory; Food Type: OMN = omnivores, INS = insectivores, FRU = frugivores, PIS = piscivores, BEN = benthivores, CAR = carnivores; Habitat type: F1 = species that only use forested areas, F2 = forest species that also use open areas, F3 = species that use boscage areas, O1 = open area species, O2 = species that prefer open areas but also use forested areas, W1 = swimming birds that use open water, W2 = waders, W3 = species that conceal themselves in marshes and aquatic areas with high grass.

Appendix C

Bird species moving into and out of the study area in different urbanization stages.



Appendix D

The change of bird species and the MSA in the 2nd stage (2004–2005), with the 1st stage (2002 − 2003) as the baseline.

		Species ID (according to the bird list Appendix A)	Richness	MSA Using 2002–2003 species as baseline
Opening species	2002–2003 Baseline	#0050#0089#0090#0096#0169#0171#0195#0204#0221#0274#0276#0313#0323#0334#0345#0348#0349#0351#0363#0364#0365#0380#388#0392#0393#0492#0508#0518#0535#0539#0542#0543#0545#0546#0547#0552#0614#0616#0636#0700#0707#0714#0717#0773#0787#0821#0823#0829#0850#0862#0882#0884#0898#0913#0922#0931#0932#0933#0961#1036#1145#1172#1174#1198#1207#1211#1218#1239#1293#1314#1315#1316#1322	73	8.28 (100%)
Species change	Moving into (+)	#0014#0193#0544#0730#0745#0995#1017#1212#1307	9	
	Moving out (−)	#0050#0089#0090#0096#0169#0195#0221#0274#0345#0349#0363#0364#0365#0380#0388#0393#0492#0508#0539#0542#0543#0614#0700#0773#0787#0821#0823#0850#0862#0932#0961#1172#1174#1211#1315	35	
Closing species	2004–2005	#0014#0171#0193#0204#0276#0313#0323#0334#0348#0351#0392#0518#0535#0544#0545#0546#0547#0552#0616#0636#0707#0714#0717#0730#0745#0829#0882#0884#0898#0913#0922#0931#0933#0995#1017#1036#1145#1198#1207#1212#1218#1239#1293#1307#1314#1316#1322	47	6.36 (76.8%)

The change of bird species and the MSA in the 3rd stage (2006–2008).

		Species ID (according to the bird list Appendix A)	Richness	MSA Using 2002–2003 species as baseline
Opening species	2004–2005	#0014#0171#0193#0204#0276#0313#0323#0334#0348#0351#0392#0518#0535#0544#0545#0546#0547#0552#0616#0636#0707#0714#0717#0730#0745#0829#0882#0884#0898#0913#0922#0931#0933#0995#1017#1036#1145#1198#1207#1212#1218#1239#1293#1307#1314#1316#1322	47	6.36 (76.8%)
Species change	Movinginto (+)	#0274#0349#0492#0508#0539#0614#0691#0702#0712#0734#0735#0773#0787#0821#0862#0988#0993#1246#1311	19	
	Moving out (−)	#0014#0193#0204#0313#0348#0351#0552#0717#0884#0933#1017#1307#1314#1316	14	
Closing species	2006–2008	#0171#0274#0276#0323#0334#0349#0392#0492#0508#0518#0535#0539#0544#0545#0546#0547#0614#0616#0636#0691#0702#0707#0712#0714#0730#0734#0735#0745#0773#0787#0821#0829#0862#0882#0898#0913#0922#0931#0988#0993#0995#1036#1145#1198#1207#1212#1218#1239#1246#1293#1311#1322	52	4.32 (52.2%)

The change of bird species and the MSA in the 4th stage (2009–2013).

		Species ID (according to the bird list Appendix A)	Richness	MSA Using 2002–2003 species as baseline
Opening species	2006–2008	#0171#0274#0276#0323#0334#0349#0392#0492#0508#0518#0535#0539#0544#0545#0546#0547#0614#0616#0636#0691#0702#0707#0712#0714#0730#0734#0735#0745#0773#0787#0821#0829#0862#0882#0898#0913#0922#0931#0988#0993#0995#1036#1145#1198#1207#1212#1218#1239#1246#1293#1311#1322	52	4.32 (52.2%)
Species change	Moving into (+)	#0103#0163#0313#0324#0630#0700#0717#0823#0984#1314#1316	11	
	Moving out (−)	#0171#0274#0334#0349#0392#0544#0545#0546#0614#0691#0712#0730#0734#0745#0882#0913#0922#0931#0993#0995#1036#1212#1246	23	
Closing species	2009–2013	#0103#0163#0276#0313#0323#0324#0492#0508#0518#0520#0535#0539#0547#0616#0630#0636#0700#0702#0707#0714#0717#0735#0773#0787#0821#0823#0829#0862#0898#0984#0988#1145#1198#1207#1218#1293#1311#1314#1316#1322	40	2.00 (24.5%)

References

Aronson, M.F.J., La Sorte, Frank A., Nilon, Charles H., Katti, Madhusudan, Goddard, Mark A., Lepczyk, Christopher A., Warren, Paige S., Nicholas, S.G. Williams, Cilliers, Sarel, Clarkson, Bruce, Dobbs, Cynnamon, Dolan, Rebecca, Hedblom, Marcus, Klotz, Stefan, Kooijmans, Jip Louwe, Kühn, Ingolf, MacGregor-Fors, Ian, McDonnell, Mark, Mörtberg, Ulla, Pyšek, Petr, Siebert, Stefan, Sushinsky, Jessica, Werner, Peter, Winter, M., 2014. A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. Proc. R. Soc. Lond. B Biol. Sci. 281.

Batáry, P., K., K., S.-R., M., C., D.E., 2017. Non-linearities in bird responses across urbanization gradients: a meta-analysis. Glosb. Chang. Biol. 1–9.

Bekessy, S.A., White, M., Gordon, A., Moilanen, A., Mccarthy, M.A., Wintle, B.A., 2012. Transparent planning for biodiversity and development in the urban fringe. Landsc. Urban Plan. 108, 140–149.

Blandón, A.C., Perelman, S.B., Ramírez, M., López, A., Javier, O., Robbins, C.S., 2016. Temporal bird community dynamics are strongly affected by landscape fragmentation in a Central American tropical forest region. Biodivers. Conserv. 25, 311–330.

Braak, C.T., Smilauer, P., 2002. CANOCO reference manual and CanoDraw for Windows user's guide: software for canonical community ordination (version 4.5). Microcomputer Power Ithaca, NY.

Bregman, T.P., Sekercioglu, C.H., Tobias, J.A., 2014. Global patterns and predictors of bird species responses to forest fragmentation: implications for ecosystem function and conservation. Biol. Conserv. 169, 372–383.

Cai, Y.T., Tang, S.M., Yuan, X., Wang, J.J., Ma, Z.J., 2011. Checklist and change of birds in Shanghai. Journal of Fudan University(Natural Science) 50, 334–343 [In Chinese].

Carrete, M., Tella, J.L., Blanco, G., Bertellotti, M., 2009. Effects of habitat degradation on the abundance, richness and diversity of raptors across Neotropical biomes. Biol. Conserv. 142, 2002–2011.

Chace, J.F., Walsh, J.J., 2006. Urban effects on native avifauna: a review. Landsc. Urban Plan. 74, 46–69.
Ciach, M., Fröhlich, A., 2016. Habitat type, food resources, noise and light pollution explain the species composition, abundance and stability of a winter bird assemblage in an urban environment. Urban Ecosystems 20 (3), 547–559.
Cockell, C., Koeberl, C., Gilmour, I. (Eds.), 2006. Biological Processes Associated With Impact Events. Springer, Berlin.
Conole, L.E., Kirkpatrick, J.B., 2011. Functional and spatial differentiation of urban bird assemblages at the landscape scale. Landsc. Urban Plan. 100, 11–23.
Ditchkoff, S.S., Saalfeld, S.T., Gibson, C.J., 2006. Animal behavior in urban ecosystems: modifications due to human-induced stress. Urban Ecosys. 9, 5–12.
Edens, B., Hein, L., 2013. Towards a consistent approach for ecosystem accounting. Ecol. Econ. 90, 41–52.
Gaston, K.J., 2000. Global patterns in biodiversity. Nature 405, 220–227.
González-Oreja, J.A., 2011. Birds of different biogeographic origins respond in contrasting ways to urbanization. Biol. Conserv. 144, 234–242.
Gregory, R.D., Van Strien, A., Vorisek, P., Meyling, A.W.G., Noble, D.G., Foppen, R.P.B., Gibbons, D.W., 2005. Developing indicators for European birds. Philosophical Transactions of the Royal Society of London B: Biological Sciences. 360, pp. 269–288.
Gulezian, P.Z., Nyberg, D.W., 2010. Distribution of invasive plants in a spatially structured urban landscape. Landsc. Urban Plan. 95, 161–168.
Haines-Young, R., Páramo, F., 2006. Land Accounts for Europe 1990–2000: Towards Integrated Land and Ecosystem Accounting. Office for Official Publ. of the European Communities.
Hein, L., Obst, C., Edens, B., Remme, R.P., 2015. Progress and challenges in the development of ecosystem accounting as a tool to analyse ecosystem capital. Curr. Opin. Environ. Sustain. 14, 86–92.
Jin, X.B., Zhou, B.C., Qin, X.K., Cui, Z.X., Xia, J.H., Si, Q., Liu, M.P., 2004. The biodiversity of the derelict Jiangwan airport in Shanghai. The 6th Conference Of National Biodiversity Conservation and Sustainable Development Lijiang, Yunnan, China.
Jokimäki, Jukka, Kaisanlahti-Jokimäki, Marja-Liisa, Suhonen, Jukka, Clergeau, Philippe, Pautasso, Marco, Fernández-Juricic, Esteban, 2011. Merging wildlife community ecology with animal behavioral ecology for a better urban landscape planning. Landsc. Urban Plan. 100 (4), 383–385.
Kattwinkel, M., Biedermann, R., Kleyer, M., 2011. Temporary conservation for urban biodiversity. Biol. Conserv. 144, 2335–2343.
Keinath, D.A., Doak, D.F., Hodges, K.E., Prugh, L.R., Fagan, W., Sekercioglu, C.H., Buchart, S.H.M., Kauffman, M., 2017. A global analysis of traits predicting species sensitivity to habitat fragmentation. Glob. Ecol. Biogeogr. 26, 115–127.
Löfvenhaft, K., Runborg, S., Sjögren-Gulve, P., 2004. Biotope patterns and amphibian distribution as assessment tools in urban landscape planning. Landsc. Urban Plan. 68, 403–427.
Luan, X., 2003. Studies on Avian Community of Shanghai and Planning of Conservation. [Thesis]. East China Normal University Shanghai, China.
MacGregor-Fors, I., 2008. Relation between habitat attributes and bird richness in a western Mexico suburb. Landsc. Urban Plan. 84, 92–98.
Mackinnon, J., et al., 2000. A field guide to the birds of China. Hunan Educational Press, Hunan, China.
Marzluff, J., Bowman, R., Donnelly, R. (Eds.), 2012. Avian Ecology and Conservation in an Urbanizing World. Springer Science & Business Media.
Matsuba, M., Nishijima, S., Katoh, K., 2016. Effectiveness of corridor vegetation depends on urbanization tolerance of forest birds in central Tokyo, Japan. Urban For. Urban Green. 18, 173–181.
McGarigal, K., Cushman, S.A., Neel, M.C., Ene, E., 2002. FRAGSTATS: Spatial Pattern Analysis Program for Categorical Maps. University of Massachusetts, Amherst. In.
McGranahan, G., Satterthwaite, D., 2003. Urban centers: an assessment of sustainability. Annu. Rev. Environ. Resour. 28, 243–274.
McKinney, M.L., 2008. Effects of urbanization on species richness: a review of plants and animals. Urban Ecosys. 11, 161–176.
McLellan, R., Iyengar, L., Jeffries, B., Oerlemans, N., 2014. Living Planet Report 2014: Species and Spaces, People and Places. World Wide Fund for Nature.
Melles, S., Glenn, S., Martin, K., 2003. Urban bird diversity and landscape complexity: species–environment associations along a multiscale habitat gradient. Conserv. Ecol. 7.
Millennium Ecosystem Assessment, 2005a. Current State and Trends, Ecosystems and Human Well-being. Island Press, Washington, DC.
Millennium Ecosystem Assessment, 2005b. Ecosystems and Human Well-Being: Biodiversity Synthesis. World Resources Institute, Washington, DC.
Ortega-Álvarez, R., MacGregor-Fors, I., 2011. Dusting-off the file: a review of knowledge on urban ornithology in Latin America. Landsc. Urban Plan. 101, 1–10.
Palomino, D., Carrascal, L.M., 2006. Urban influence on birds at a regional scale: a case study with the avifauna of northern Madrid province. Landsc. Urban Plan. 77, 276–290.
Pauchard, A., Aguayo, M., Peña, E., Urrutia, R., 2006. Multiple effects of urbanization on the biodiversity of developing countries: the case of a fast-growing metropolitan area (Concepción, Chile). Biol. Conserv. 127, 272–281.
Pettit, C.J., Raymond, C.M., Bryan, B.A., Lewis, H., 2011. Identifying strengths and weaknesses of landscape visualisation for effective communication of future alternatives. Landsc. Urban Plan. 100, 231–241.
Pfisterer, A.B., Schmid, B., 2002. Diversity-dependent production can decrease the stability of ecosystem functioning. Nature 416, 84–86.
Purvis, A., Hector, A., 2000. Getting the measure of biodiversity. Nature 405, 212–219.
Quesnelle, P.E., Fahrig, L., Lindsay, K.E., 2013. Effects of habitat loss, habitat configuration and matrix composition on declining wetland species. Biol. Conserv. 160, 200–208.
Reis, E., López-Iborra, G.M., Pinheiro, R.T., 2012. Changes in bird species richness through different levels of urbanization: implications for biodiversity conservation and garden design in Central Brazil. Landsc. Urban Plan. 107, 31–42.
Remme, R.P., Hein, L., van Swaay, C.A.M., 2016. Exploring spatial indicators for biodiversity accounting. Ecol. Indic. 70, 232–248.
Robinson, G.R., Holt, R.D., Gaines, M.S., Hamburg, S.P., Johnson, M.L., Fitch, H.S., Martinko, E.A., 1992. Diverse and contrasting effects of habitat fragmentation. Science 524–526.
Sandström, U.G., Angelstam, P., Mikusiński, G., 2006. Ecological diversity of birds in relation to the structure of urban green space. Landsc. Urban Plan. 77, 39–53.
Schluter, D., Pennell, M.W., 2017. Speciation gradients and the distribution of biodiversity. Nature 546, 48–55.
Schröter, M., Barton, D.N., Remme, R.P., Hein, L., 2014. Accounting for capacity and flow of ecosystem services: a conceptual model and a case study for Telemark, Norway. Ecol. Indic. 36, 539–551.
Schröter, M., Remme, R.P., Sumarga, E., Barton, D.N., Hein, L., 2015. Lessons learned for spatial modelling of ecosystem services in support of ecosystem accounting. Ecosyst. Serv. 13, 64–69.
Scolozzi, R., Geneletti, D., 2012. A multi-scale qualitative approach to assess the impact of urbanization on natural habitats and their connectivity. Environ. Impact Assess. Rev. 36, 9–22.
Shen, Z.Y., Hou, X.S., Li, W.K., Aini, G., Chen, L., Gong, Y.W., 2015. Impact of landscape pattern at multiple spatial scales on water quality: a case study in a typical urbanised watershed in China. Ecol. Indic. 48, 417–427.
Sundell-Turner, N.M., Rodewald, A.D., 2008. A comparison of landscape metrics for conservation planning. Landsc. Urban Plan. 86, 219–225.
Tomasevic, J.A., Marzluff, J.M., 2017. Cavity nesting birds along an urban-wildland gradient: is human facilitation structuring the bird community? Urban Ecosys. 20, 435–448.
UNSD, 2014. System of Environmental-Economic Accounting 2012—Experimental Ecosystem Accounting (In: New York).
Vandewalle, M., Bello, F., Berg, M.P., Bolger, T., Dolédec, S., Dubs, F., Feld, C.K., Harrington, R., Harrison, P.A., Lavorel, S., Silva, P.M., Moretti, M., Niemela, J., Santos, P., Sattler, T., Sousa, J.P., Sykes, M., Vanbergen, A.J., Woodcock, B.A., 2010. Functional traits as indicators of biodiversity response to land use changes across ecosystems and organisms. Biodivers. Conserv. 19, 2921–2947.
Whitworth, A., Villacampa, J., Serrano Rojas, S.J., Downie, R., MacLeod, R., 2017. Methods matter: different biodiversity survey methodologies identify contrasting biodiversity patterns in a human modified rainforest — a case study with amphibians. Ecol. Indic. 72, 821–832.
Wood, J.M., Quinn, J.E., 2016. Local and landscape metrics identify opportunities for conserving cavity-nesting birds in a rapidly urbanizing ecoregion. J. Urban Econ. 1, 10.
Yeh, C.-T., Huang, S.-L., 2009. Investigating spatiotemporal patterns of landscape diversity in response to urbanization. Landsc. Urban Plan. 93, 151–162.
Yuan, Y.J., Zeng, G.M., Liang, J., Li, X.D., Li, Z.W., Zhang, C., Huang, L., Lai, X., Lu, L.H., Wu, H.P., 2014. Effects of landscape structure, habitat and human disturbance on birds: a case study in East Dongting Lake wetland. Ecol. Eng. 67, 67–75.
Zhou, D.Q., Fung, T., Chu, L.M., 2012. Avian community structure of urban parks in developed and new growth areas: a landscape-scale study in Southeast Asia. Landsc. Urban Plan. 108, 91–102.