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Influence of breeding habitat characteristics and landscape heterogeneity on anuran species richness and abundance in urban parks of Shanghai, China

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ABSTRACT

Urbanization has caused a significant decline in amphibians worldwide due to habitat loss, fragmentation, and degradation of habitat quality. Accordingly, parks have become “islands,” or habitat fragments, for amphibians in highly urbanized areas. Understanding the habitat use pattern of amphibians in fragmented urban environments is essential for biodiversity conservation in urban ecosystems. Several environmental features may affect anuran communities in urban parks, including the characteristics of fragmentation, landscape heterogeneity, breeding habitat, and human disturbance. In particular, it is unclear whether a larger habitat area could lead to higher anuran species richness (species-area relationship) in urban parks. The aim of this study was to examine whether the species-area relationship is relevant for anurans in urban parks, and to determine the environmental characteristics that likely influence anuran communities. We used a visual encounter method to survey anuran communities (species richness and abundance) in 16 parks located in highly urbanized areas of Shanghai, China. Fragmentation characteristics included fragment size and shape index. Landscape heterogeneity was measured as compositional heterogeneity (using the Shannon diversity index of wetland types) and configurational heterogeneity (using edge density of different land use types). We found that compositional and configurational heterogeneity had significant positive effects on anuran species richness and relative abundance, respectively, in the urban parks. We also found that high edge density along streams benefited anuran abundance. However, there was no significant relationship between fragment size and anuran communities, and the abundances of *Pelophylax plancyi* and *P. nigromaculata* were negatively associated with the edge density of large ponds and roads. Our results suggest the importance of landscape structure in urban parks for sustaining anuran persistence. In addition, diverse breeding habitats and landscape heterogeneity should be considered positive indicators of anuran biodiversity conservation in fragmented urban habitats.

1. Introduction

Increasing urbanization has led to a loss of biodiversity, which is currently attracting worldwide concern (Grimm et al., 2008). Amphibians are some of the most vulnerable vertebrates, and many have been brought to the verge of extinction (Baillie et al., 2004). Habitat loss, habitat fragmentation/isolation, and degradation of habitat quality related to urbanization are all threats to amphibian populations (Cushman, 2006; Becker et al., 2007; Hamer and McDonnell, 2008). In this context, parks typically play the role of “islands,” or habitat fragments, for wildlife in urbanized areas. Therefore, managing parks as wildlife habitat is an effective method for biodiversity conservation in urban ecosystems (Alvey, 2006; Scheffers and Paszkowski, 2012). Accordingly, understanding how environmental characteristics affect amphibian communities in parks is important for promoting amphibian biodiversity conservation in urbanized environments (Smallbone et al., 2011; Scheffers and Paszkowski, 2013; Fuyuki et al., 2014).

The species-area relationship (SAR) is one of the most studied patterns in ecology (Franzén et al., 2012; Matthews et al., 2014), and the SAR posits that larger islands/areas generally contain more species than smaller ones (MacArthur and Wilson, 1967). The most common explanation of the SAR is based on the equilibrium theory of island

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biogeography, which assumes that the size of islands/habitat has a strong effect on the dynamic equilibrium between immigration and extinction rates (MacArthur and Wilson, 1967; Mcguinness, 1984). Several studies have indicated a positive relationship between fragment/habitat area and measures of amphibian communities, including species richness (Bell and Donnelly, 2006; Lima et al., 2015; Almeida-Gomes et al., 2016), abundance (Marsh and Pearman, 1997; Guerry and Hunter, 2002), and genetic diversity (Dixo et al., 2009). However, negative or null effects for SAR also have been reported (Hillers et al., 2008, Lion et al., 2014; Wang et al., 2014). Previous studies almost solely focus on the SAR of amphibians in forest fragments or natural islands. Few studies have looked at this relationship in urban fragments. Understanding whether the SAR is relevant for amphibians in urban parks is fundamental to public green space design and thus the conservation biology of amphibians in rapidly urbanized ecosystems, such as Shanghai.

Landscape heterogeneity (including compositional heterogeneity and configurational heterogeneity) also has a positive effect on biodiversity (Tews et al., 2004; Fahrig et al., 2011), and the landscape habitat heterogeneity hypothesis, which predicts that larger areas should have higher species richness due to greater landscape heterogeneity, supports the SAR (Williamson et al., 2001; Tews et al., 2004). Indeed, some studies have found that landscape heterogeneity may be a stronger predictor than SAR in determining animal species richness and abundance (Báldi, 2008; Lizée et al., 2012), and it is clear landscape heterogeneity benefits amphibian biodiversity in agricultural landscapes (Collins and Fahrig, 2017). However, it remains uncertain whether landscape heterogeneity has a greater effect on amphibian communities than the SAR in urban fragments.

Previous studies have shown that breeding habitat heterogeneity and landscape hydroperiod are critical for the persistence of amphibian populations (Semlitsch, 2000; Snodgrass et al., 2000; Bickford et al., 2010). For example, temporary ponds may dry too rapidly for tadpoles to complete metamorphosis and recruitment, and permanent ponds are not good breeding sites due to cold temperatures and larger numbers of predators (McCaffery et al., 2014). Amphibians also need higher breeding habitat heterogeneity due to different reproductive ecologies. So, breeding habitat heterogeneity may be even more appropriate to explain amphibian species richness and abundance in fragment habitats than the SAR (Bickford et al., 2010). Designing and constructing breeding habitats with characteristics that maintain and enhance anuran biodiversity could be an effective measure for amphibian conservation, assuming we can determine that diverse breeding habitats are positively related to amphibian diversity in urban fragments.

Human population density and socioeconomic factors can be strong predictors of anuran biodiversity in urban ecosystems (Hamer and Parris, 2011; Smallbone et al., 2011; Zhang et al., 2016). Human disturbance could be a negative factor for amphibian communities because of larger human populations around urban parks in Shanghai (Zhang et al., 2016). The shape of fragments also may affect anurans in urban parks, because it is difficult for individual animals to spread from the center to the edge in a complex-shaped habitat (Diamond, 1984). Moreover, fragments with more complex shapes also have stronger edge effects caused by boundaries (Murcia, 1995). Thus, this study included the number of visitors to each park and an index of the fragment shape index as factors to explain amphibian species number and abundance.

Shanghai has the highest urbanization rate of China, with a population of more than 20 million crowded into an area of 6340 km^2. Six anuran species have been recorded in Shanghai parks, including *Pelophylax nigromaculata*, *P. plancyi*, *Fejervarya multistriata*, *Microhyla fissipes*, *Bufo gargarizans,* and *Kaloula borealis*. Previous studies have focused on the effect of landscape and microhabitat factors on anurans in urban ponds, and have suggested that the increasing urbanization of Shanghai has had a negative effect on anuran species richness, abundance, and body condition (Li et al., 2016; Zhang et al., 2016). However, potential solutions to maintaining and enhancing anuran biodiversity in urban fragments experiencing rapid urbanization have been lacking.

Therefore, we surveyed anuran species richness and abundance in 16 Shanghai parks. We also measured environmental variables and landscape features of the parks. Edge density values for different land use types were measured in order to examine their effects on the abundance of dominant anurans within the parks of Shanghai. Our objectives of this study were to understand: (1) whether the SAR is relevant for anurans in urban parks; (2) if anuran species richness and abundance are associated with the landscape heterogeneity of urban parks; and (3) how fragment shape, breeding habitat heterogeneity, and human disturbance affect anuran communities in Shanghai's urban parks.

2. Materials and methods

2.1. Study area and site selection

Shanghai is located in eastern China in the southeastern Yangtze River Delta. A total of 165 urban parks and green spaces form remnant and restored habitats for anurans within the urban environment of this city (Shanghai Municipal Statistics Bureau, 2016). Anuran breeding habitats within these areas are currently limited to diverse permanent and semi-permanent wetlands in the parks, including ponds, streams, and reservoirs (Zhang et al., 2016).

Potential urban parks were identified using aerial imagery and field inspection prior to surveying (Zhang et al., 2016). We selected 16 parks in highly urbanized areas of Shanghai surrounded by impermeable surfaces (buildings and roads) and with wetlands and vegetation coverage rates > 70% (Fig. 1). These were considered to provide suitable habitats for anuran in urban areas and have a gradient of landscape variables (Table A1 in Supplementary materials). These parks ranged in fragment size from between 8 ha and 178 ha (Table A1 in Supplementary materials). Most parks (ten out of 16 parks) opened in the morning at 05:00, although four and two parks opened at 06:00 and 07:00, respectively. Half of these 16 parks closed before 19:00, while seven parks were open until 21:00, and Zhongshan Park remained open all night. Therefore, human disturbance during the night varied significantly in these parks when anurans were actively foraging and breeding.

2.2. Anuran sampling

Anuran sampling transects was established along permanent and semi-permanent wetlands in the parks (ponds, streams, and reservoirs). Because anurans need water bodies for breeding habitat, we conducted the survey during the breeding season. The length of the sampling transects was determined based on the size of the parks. In small parks (total area ≤ 30 ha or with a perimeter for all water bodies ≤ 3000 m), transects of 1000 m to 3000 m were established along all water bodies and areas with partial vegetation cover. For larger parks (fragment size > 30 ha), we established transects > 3000 m in length and covered ≥ 10% the area of parks in the vicinity of water bodies and vegetation cover (Fig. B1 in Supplementary materials).

We conducted a series of surveys from April to August of 2014 and 2016 to cover the breeding seasons of all anurans present in Shanghai (Zhang et al., 2016). The transects were 5-m wide with a length of at least 1000 m in each park. Surveys were conducted at least 0.5 h after sunset (at 19:00–22:00) when there was no rain and wind speeds < 30 km/h. Visual encounter surveys were chosen to detect anurans (Crump and Scott, 1994). Surveys involved three people walking the transect as a group with flashlights to search for anurans at a steady walking speed of 1.5 km/h. Observers recorded the species, number of individuals, location, and the observed time. Anurans were hand caught for identification if they were hard to identify, then released in the same site where they were captured. Individuals that were only heard were

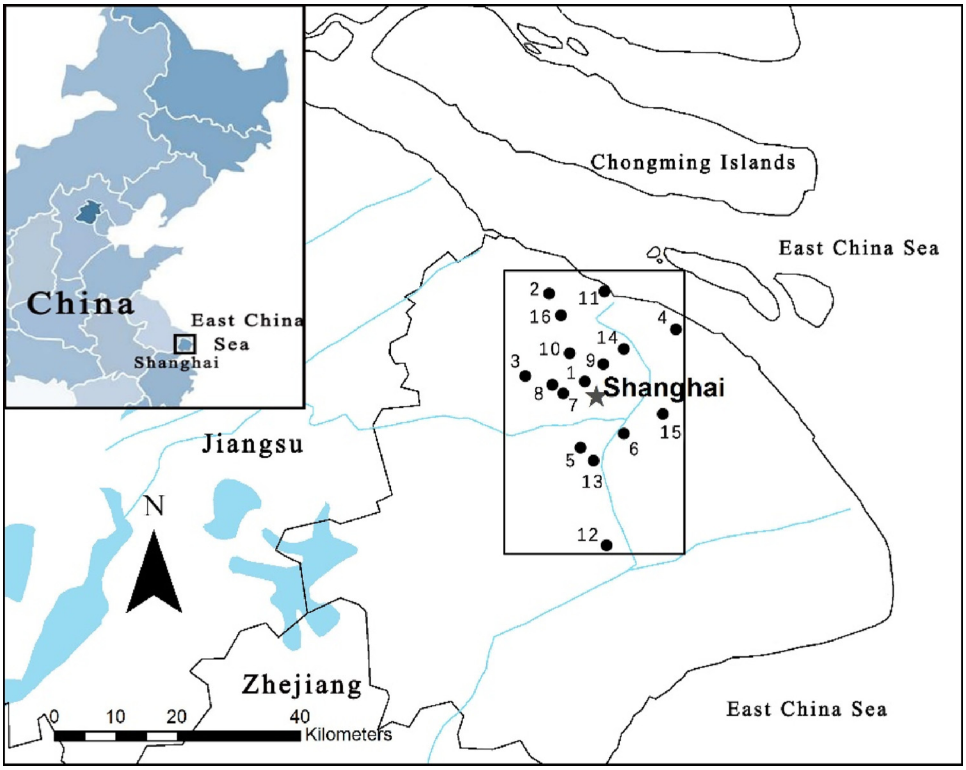

Fig. 1. Location of study region and the 16 urban parks in Shanghai, China. The insert box represents the highly urbanized area of Shanghai. The numbers correspond to the parks listed in Table A1.

left out of the abundance analysis but included in the richness analysis. Each park was surveyed on three or four occasions until the species-accumulation curve reached an obvious asymptote (no additional species were found). Total sampling effort (242 h) ranged from 8 to 36 h for each of the urban parks.

We measured anuran species richness and relative abundance. Anuran species richness was the total number of anuran species observed at each park on all visits. Because of the different transect lengths, anuran population densities were used to estimate relative abundance, which was calculated as:

$$D_i = \frac{\sum N_{ik}}{L_i} / K_i$$

where *Di* is the anuran population density in park *i*, *Nik* is the number of individuals in park *i* on occasion *k*, *Li* is the length of transect in park *i*, and *Ki* is the total number of sampling occasions in park *i* (Zhang et al., 2016).

2.3. Landscape data

Six landscape variables were defined and investigated to explain variance in anuran communities of urban parks: (1) fragment size (Fs), (2) fragment shape index (Fsi), (3) breeding site heterogeneity (Bsh), (4) compositional heterogeneity (Cph), (5) configurational heterogeneity (Cfh), and (6) human disturbance (Hd) (Table 1).

Fragment size and the perimeter of the 16 urban parks were obtained from Google Earth Plus 6.0.1. Land cover data were obtained from satellite images of Formosat-2 (June 2012; 2-m resolution), and land use types included woodlands, grasslands, buildings, roads, and diverse water bodies (Table 2). Fragstats 4.0 (McGarigal et al., 2002) and ArcMap 10.0 (ESRI) were used to derive and process the landscape heterogeneity metrics. All landscape data were provided by Zhang et al. (2016). Five types of breeding sites were defined in this study based on anuran reproduction ecologies (Bickford et al., 2010), and breeding site heterogeneity score was the number of different breeding habitats (ranging from 1 to 5) found in the urban parks (Table 2). Human disturbance was measured as the number of people entering each of the parks during a 10-min period when we surveyed for anurans in the evening. We defined three different levels (ranging from 1 to 3) of human disturbance. Level 1 means no people entered the park, level 2 means less than 50 individuals entered the park, and level 3 means more than 50 people entered the park. High human disturbance scores represent more human disturbance.

To further explore the effect of configurational heterogeneity of different patch types (Fragstats 4.0: class scale) (McGarigal et al., 2002) on anuran relative abundance, the edge density of each patch type was also measured in each park. The common patch types included woodlands, grasslands, roads, buildings, small ponds, large ponds, and long streams (short streams and artificial reservoirs were rare in the 16 parks, see Table 2).

2.4. Statistical analyses

A simple linear regression was used to test the effect of fragment size on anuran species richness and relative abundance in urban parks, which was based on the relation $\log(Y) = \log(c) + z\log(A)$ (Arrhenius, 1921). 'Y' represents species richness or abundance, 'A' represents the size of the urban park, and 'c' represents an empirically determined constant (Lizée et al., 2012).

In order to explore the relationship between the remaining five predictor variables (fragment shape index, breeding site heterogeneity, compositional heterogeneity, configurational heterogeneity, and human disturbance) and two response variables (species richness and relative abundance), multiple linear regression was used to determine which predictor variables were the best predictors of anuran species richness and relative abundance. Configurational heterogeneity was $\log_{10}$ transformed; Fragment shape index and anuran relative abundance were standardized to zero and unit variance to improve the linear

Table 1
List of the six landscape variables used in this study.

Variable	Description
Fragment size (Fs)	The area of each park (m^2).
Fragment shape index (Fsi)	$Fsi = \frac{0.25Fp}{\sqrt{Fs}}$, where Fp = the perimeter of each park (m), Fs = the area of each park (m^2). When the score of Fsi is close to 1, the shape of the park is close to square.
Breeding site heterogeneity (Bsh)	Diversity of anuran breeding sites in each park, five breeding site types were defined in order to measure this variable in each park.
Compositional heterogeneity (Cph)	The Shannon diversity index of wetland types in the parks was chosen to represent compositional heterogeneity of breeding habitats. $SHDI = -\sum_{i=1}^{m} Pi\ln(Pi)$, where Pi = proportion of the landscape of wetlands occupied by different breeding sites (class) i.
Configurational heterogeneity (Cfh)	The edge density of land use types at the landscape scale was chosen to represent configurational heterogeneity. $ED = \frac{Fe}{Fs}10^6$, where Fe = the total length of the patch border in each park (m), Fs = the area of each park (m^2).
Human disturbance (Hd)	Number of visitors that entered each park during the time of the survey.

Table 2
Description of breeding site heterogeneity used in this study.

Category	Description	Distribution in parks
Large pond	Still water > 100 m^2	15/16
Small pond	Still water ≤100 m^2	11/16
Long stream	Running water > 100 m long and > 4 m wide	14/16
Short stream	Running water ≤100 m long and ≤4 m wide	3/16
Artificial reservoir	Reservoir and fountain, still water ≤100 m^2	4/16

relationship between predictor variables and response variables. To avoid the collinearity of variables, Spearman correlation test was used for pairwise comparisons of five predictor variables. Pairs with significant correlation ($|r| > 0.7$) were judged as having strong collinearity (Dormann et al., 2013), and thus were not used in the same model. A multi-model inference approach using Akaike information criterion corrected (AICc) was utilized to calculate and compare standardized model-weight mean coefficients to determine the direction and relative importance of the predictor variable on response variables. Differences of AICc (△AICc) were used to determine the level of support for each model in the candidate set. Candidate models with △AICc < 4 were considered the competing models for making inferences. Akaike weights (W*i*) were additionally calculated to explain the probability that any given model was the best model in the entire of set of candidate models (Burnham and Anderson, 2002). Model averaging was used to provide model coefficients and variances because the W*i* suggested no individual model was clearly the best (W*i* > 0.9) (Anderson et al., 2001).

Pearson correlation was used to test the relationship between edge density of different patch types and anuran relative abundance (especially three individual anuran species *P. plancyi*, *B. gargarizans*, and *P. nigromaculata*) in urban parks. We used the square root transformation for edge density of woodlands, long streams, and large ponds, and the log10 transformation for overall anuran relative abundance of the three dominant anuran species for normality.

All statistical analyses were performed using R 3.3.2 (R Core Team, 2016). The "glmulti" (Calcagno, 2013) and "MuMIn" packages (Barton, 2016) were used to carry out model selection and average. Normality was tested using Shapiro–Wilk tests.

3. Results

3.1. Anuran survey

We observed six anuran species and 2314 individual anurans in 2014 and 2016 in the parks (Table A2 in Supplementary materials). Three species accounted for 90.35% of individuals: *P. plancyi* (52.07%; 1205 individuals; recorded in 15 of 16 parks, 15/16), *B. gargarizans* (21.69%; 502 individuals; 16/16), and *P. nigromaculata* (16.59%; 384 individuals; 14/16). *F. multistriata* (8.21%; 190 individuals; 7/16), *M. fissipes* (1.34%; 31 individuals; 3/16), and *K. borealis* (0.09%; two individuals; 1/16) were seldom recorded in the parks (Table A2 in Supplementary materials). The highest species richness and relative abundance of anurans at a single park were five species and 0.123 individuals per m (Table A2 in Supplementary materials). The average number of species and relative abundance observed per park were 3.438 ± 1.031 and 0.030 ± 0.032 individuals per m (SD), respectively.

3.2. Species-area relationship of anuran

First, fragment size had no significant influence either on species richness ($R^2 = 0.143$, $P = 0.149$, Fig. 2a) or relative abundance ($R^2 = 0.127$, $P = 0.176$, Fig. 2b) of anurans in urban parks.

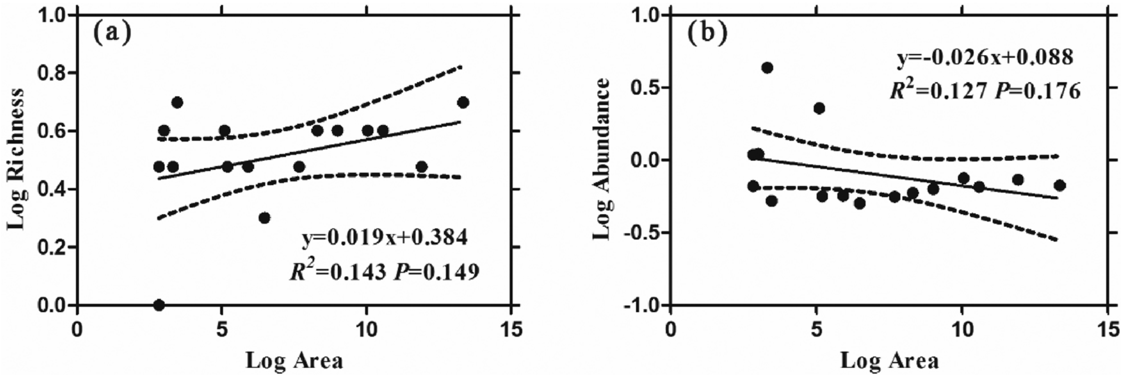

Fig. 2. Simple linear regression of anuran species richness, relative abundance and urban park area. The upper and lower limits of the 95% confidence intervals were showed with dotted lines.

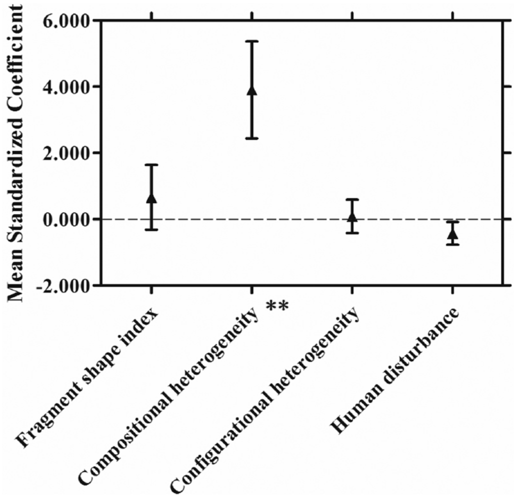

Fig. 3. Model averaged standardized coefficients and standard errors, from the best models ($\triangle$AICc < 4), showing the direction and relative magnitude of the effects of each predictor variable (fragment shape index, compositional heterogeneity, configurational heterogeneity, and human disturbance) on anuran species richness in urban parks of Shanghai (**$P < 0.01$).

3.3. *Anuran species richness and relative abundance response to landscape variables*

Pairwise correlation showed that the correlation between compositional heterogeneity and breeding site heterogeneity was strong and significant ($r = 0.833$, $P < 0.001$) (Table A3 in Supplementary materials), and both of these two variables represent breeding habitat diversity. Therefore, compositional heterogeneity, which could be a more useful measure for managers, was chosen in the multiple linear regression.

When four variables were included as predictor variables in the multiple linear regressions for species richness, five top models were chosen using AICc ($\triangle$AICc < 4), which contained all four predictor variables (fragment shape index, compositional heterogeneity, configurational heterogeneity, and human disturbance). In addition, all of these five top models included compositional heterogeneity (positive effect). The R^2 of the five top models ranged from 0.262 to 0.532 (Table A4 in Supplementary materials). Compositional heterogeneity was the only variable that was positive significantly associated with species richness in model averaging of the top five models (estimate 3.900 ± 1.465 SE, $P = 0.008$) (Fig. 3).

Model selection using AICc ($\triangle$AICc < 4) produced nine models for anuran relative abundance. These nine top models also contained all four predictor variables. Furthermore, five of these nine top models included configurational heterogeneity (positive effect). The R^2 of these top nine models ranged from 0.093 to 0.356 (Table A4 in Supplementary materials). The model average also indicated that configurational heterogeneity had a significant positive relationship with anuran relative abundance (estimate 0.652 ± 0.559 SE, $P = 0.038$) (Fig. 4).

3.4. *Effect of configurational heterogeneity*

Anuran relative abundance increased with higher edge density along streams in urban parks ($r = 0.725$, $P = 0.008$). The relative abundance of *P. plancyi* was negatively related to large pond edge density ($r = -0.776$, $P = 0.003$). *P. nigromaculata* relative abundance was negatively correlated with road edge density ($r = -0.682$, $P = 0.030$). We did not find any corresponding relationship between *B. gargarizans* relative abundance and edge density of different land use types (Table 3).

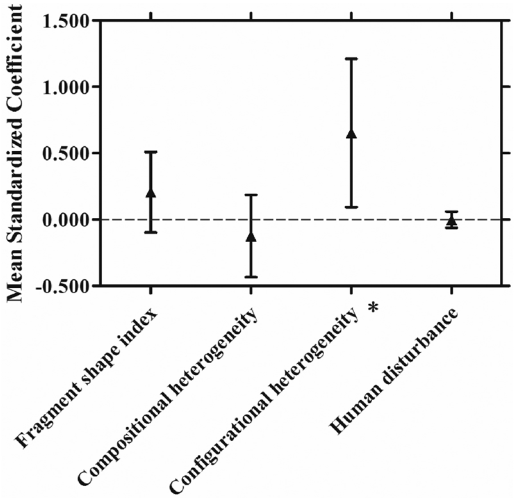

Fig. 4. Model averaged standardized coefficients and standard errors, from the best models ($\triangle$AICc < 4), showing the direction and relative magnitude of the effects of each predictor variable (fragment shape index, compositional heterogeneity, configurational heterogeneity, and human disturbance) on anuran relative abundance in urban parks of Shanghai (*$P < 0.05$).

4. Discussion

Our data indicated compositional heterogeneity of breeding habitats was significantly positively correlated with anuran species richness in urban parks (Fig. 3). This result overrode the SAR, and is in agreement with the findings of other studies (Zimmerman and Bierregaard, 1986; Bickford et al., 2010). In addition, we found that landscape configurational heterogeneity had a significant positive effect on anuran relative abundance in urban parks (Fig. 4), as was the case for butterflies (Lizée et al., 2012), bees (Holzschuh et al., 2010), beetles (Sanchez-de-Jesus et al., 2016) and birds (Morelli et al., 2013). Other studies have found similar relationships between anuran abundance and landscape configurational heterogeneity (Knutson et al., 1999; Guerry and Hunter 2002; Collins and Fahrig, 2017).

4.1. *Effect of diverse breeding habitats on anuran species richness*

Previous studies have shown that the SAR has clear effects on anuran species richness in forest landscapes (Lima et al., 2015; Almeida-Gomes et al., 2016); however, in this study, compositional heterogeneity of breeding habitats better explained anuran species richness as a more direct mechanism than the SAR alone in urban parks (Fig. 3). Many anuran species are associated with specific types of water bodies for breeding and habitat (Semlitsch, 2000). For example, breeding habitats with high vegetation and water coverage, such as small ponds in urban parks, are usually more suitable for *P. plancyi* (Shu et al., 2016). Water bodies with high vegetation coverage but deep water, such as long streams, are more ideal habitat for *P. nigromaculata* (Wang et al., 2008). To our knowledge, the toad *B. gargarizans* usually lives in artificial reservoirs. Therefore, urban parks with diverse breeding habitats contain more species regardless of their fragment sizes.

Moreover, diverse breeding habitats in different hydroperiods reduces the negative effects of environmental and climatic stochasticity (McCaffery et al., 2014). During our survey, *M. fissipes*, and *K. borealis* were usually found in the temporary water bodies created by rain. Therefore, maintaining diverse breeding habitats as complementary breeding habitats can reduce anuran population variability and promote persistence of species richness (Snodgrass et al., 2000; Werner et al., 2007; Venne et al., 2012).

Previous studies have suggested different relationships between breeding habitat size and anuran diversity: larger breeding habitats

Table 3
Pearson correlation coefficients (r) between anuran relative abundance and edge density of diverse land use types in 16 urban parks (**P < 0.01, *P < 0.05).

Species	Woodland	Grassland	Small pond	Large pond	Long stream	Building	Road
Relative abundance	0.038	−0.268	0.358	−0.300	**0.725****	0.233	−0.105
P. plancyi	0.180	0.035	0.249	**−0.776****	0.481	0.041	0.033
B. gargarizans	−0.114	0.266	0.610	0.350	0.277	0.214	−0.134
P. nigromaculata	−0.295	−0.295	−0.120	0.066	0.449	0.419	**−0.682***

have positive correlation with anuran species richness (Babbitt, 2005; Villaseñor et al., 2017), intermediate-sized breeding habitats support more anuran species richness (Semlitsch et al., 2015), and even small breeding habitats benefit anuran species richness (Scheffer et al., 2006). We concluded that diverse breeding habitats of different sizes in urban parks (Table 2) could benefit anuran species richness.

4.2. *Effect of landscape configurational heterogeneity on anuran abundance*

Our results showed landscape configurational heterogeneity was positively correlated with anuran relative abundance (Fig. 4), indicating that landscapes with higher edge density benefit anuran abundance. Moreover, our results support the prediction that anuran abundance increased with increasing landscape configurational heterogeneity. Kisel et al. (2011) proposed that positive edge effects increase niche availability. Likewise, higher edge density of fragmented urban habitats leads to more prey resources, such as arthropods, which may also facilitate anuran foraging (Bolger et al., 2000). We speculate the edge effect is a significant factor influencing the abundance of anuran species, especially in artificial landscapes.

We further found that edge density of long streams has positive significant relationships with anuran relative abundance (Table 3), which indicated that long streams were the most important wetland landscape for anurans in urban parks during the breeding season. Because anurans breed near wetland edges, the high vegetation cover on the edge of long streams supported more beneficial breeding microhabitats and shelter with less predation (Martín et al., 2005; Kisel et al., 2011). Greater amounts of wetland edges also possess greater habitat connectivity, which assists anuran dispersal and provides habitat for anuran foraging and hibernation (Burne and Griffin, 2005; Hamer and Parris, 2011; Zhang et al., 2016). Knutson et al. (1999) found anurans appeared to benefit from forest wetland edges, a result consistent with our findings.

In this study, the relative abundance of *P. plancyi* was negatively related to the edge density of large ponds (Table 3). However, since *P. pl*ancyi prefer to forage and breed in small permanent wetlands, the landscape configurational heterogeneity of large ponds had a negative relationship with that of small ponds, which may explain the lesser abundance of *P. pl*ancyi in urban parks with a high edge density of large ponds. Several studies have shown that road density has a negative effect on anuran relative abundance (Fahrig et al., 1995; Carr and Fahrig, 2001; Eigenbrod et al., 2008). As a result of weak vagile ability, *P. nigromaculata* abundance declined with the increasing road edge density associated with many urban parks (Table 3). However, toads are more vagile compared to frogs, dispersing up to 5–6 km (Hamilton, 1934). Therefore, *B. gargarizans* are regarded as “urban adapters” that overcame barriers, such as roads and buildings in urban parks (Zhang et al., 2016). Also, we did not find a significant effect of the different land use types present in the parks on *B. gargarizans* (Table 3). In short, the ecological characteristics of the anurans determined the different responses to different land use types in the parks.

4.3. *No significant species-area relationship of anuran in urban parks*

Finally, there was no significant relationship between anuran species richness or relative abundance and park size in this study (Fig. 2). Previous studies have shown that habitat heterogeneity (Tews et al., 2004; Báldi, 2008; Lizée et al., 2012), habitat quality (Fleishman et al., 2002), and landscape fragmentation (Andren, 1994) can be a stronger predictor than SAR. As MacArthur and Wilson (1967) suggested, area was simply a surrogate factor for determining species richness, and specific richness could be area independent in small and unstable islands. According to the “small island (fragment) effect” hypothesis, species number does not increase with increasing area in small islands (Lomolino and Weiser, 2001). Moreover, specific richness may be more influenced by environmental characteristics, landscape structure, isolation, occasional disturbance and/or human impacts than by the effect of patch area in small islands (Triantis et al., 2006). Our results were also in agreement with the “fragmentation threshold hypothesis” (Pardini et al., 2010), which emphasizes that landscape structure (landscape heterogeneity in this study) could explain the “patch-area effect.” Moreover, several land use types (e.g. buildings and roads) in the urban parks could not be considered as resources and habitats for amphibians. Therefore, urban park size in our study could only partly reflect suitable habitats for amphibians.

Our results are consistent with the prediction that human disturbance has negative effects on anuran diversity but is not significant (Figs. 3 and 4). The most likely explanation for this discrepancy was a lack of a wide range of human disturbance in the 16 parks and the low levels of human disturbance observed in most parks (Table A1 in Supplementary materials). The fragment shape index also did not show a negative effect on anuran species richness and abundance (Figs. 3 and 4). This could be because the area surrounding the urban parks in this study was all impermeable surfaces, which results in less difference in edge effects caused by fragment shape.

Our results recognize that compositional heterogeneity of breeding habitats and configurational heterogeneity could benefit anuran species richness and abundance in urban parks. In addition, we suggest that constructing diverse breeding habitats and higher long stream edge density in urban parks could be an effective management strategy for restoring and protecting amphibian biodiversity. There are few published studies on habitat ecology for anuran species; therefore, future studies should focus on the relationship between targeted anuran species and specific types of water bodies in urban parks, which may have applicability for anuran conservation. In addition, anuran morphology (such as size and body condition) could reflect population health status more accurately than abundance alone, but more research is needed. Understanding the effect of park landscape structure on anuran morphology should be a priority for future research to improve understanding of how landscape structure affects anurans in urban parks.

5. Conclusions

In conclusion, we found that: (1) there was no significant species-area relationship of anurans in urban parks, (2) improving compositional heterogeneity of breeding habitats and landscape configurational heterogeneity can benefit anuran species richness and abundance in urban parks, and (3) maintaining higher edge density of long streams should be encouraged as a method to increase anuran abundance in the urban parks.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.ufug.2018.03.017.

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