博士論文（要約）

Evaluation of urban green space in developing Beijing:

based on the matrix of landscape connectivity and bird communities

（景観連結性と鳥類群衆状態とのマトリックス解析による

開発下北京市の都市緑地評価）

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Abstract

Urbanization has been regarded as the main reason for habitat fragmentation and biodiversity loss. Particularly in urbanized areas, urban green space plays an important role in biodiversity conservation. With the increase in human population and land cover change, the evaluation of urban green space is an essential step in landscape planning, which should be conducted based on a deep understanding of the impact of urbanization.

Previous studies have paid considerable attention to the evaluation of urban green space from the perspective of landscape pattern: for instance, landscape connectivity. Green spaces with high connectivity are often considered to provide better conditions for species migration and gene exchange, which contributes to biodiversity conservation. Landscape connectivity has been widely used as an indicator to evaluate the capacity of green space to support biodiversity, referred to as ecological efficiency in this study.

An increasing number of studies have argued that ecological processes should be involved in the evaluation of green space: for example, considering the fact that different species respond to environmental changes at different scales. However, the methods of applying ecological and biological knowledge to the evaluation of green space needs to be explored further. Several studies have modeled the evaluation of green space from the point of view of focal species; however, there are still some limitations to the existing research. First, studies using empirical data of species census in the evaluation of urban green space are limited because the acquisition of empirical data is time-consuming and laborious. Second, long-term observation is necessary to identify the mechanism by which environmental changes resulting from urbanization affect biodiversity. However, most studies use various spaces instead of time to measure the effects of urbanization. Third, since environmental factors in a complex ecosystem vary and interact with each other, multiple spatial scales should be considered in quantifying the response of species to environmental changes.

To fill these gaps in the research, this study attempts to evaluate the green spaces of Beijing, which has been undergoing rapid development and urbanization for decades, by examining species response to environmental changes at multiple spatial scales. The objectives of this study are: 1) to test the advantages and necessity of introducing species in evaluating ecological efficiency of urban green spaces, compared with the widely used landscape connectivity assessment; 2) to explore the variations in bird abundance, richness, and diversity as a function of environmental changes resulting from rapid urbanization at both local and landscape levels; and 3) to identify influential environmental variables that can best account for the variations in bird communities at multiple scales based on long-term observation.

Chapter 1 introduces the academic background, existing research on the effects of urbanization, evaluation of urban green spaces, and the response of bird communities to environmental changes. The literature review revealed some research gaps, and the aims and objectives of this research were defined based on these gaps.

Chapter 2 describes the current situation of urbanization and biodiversity conservation in Beijing and the current green space planning concerning ecological conservation in China. In addition, land cover was interpreted from remote sensing images of Beijing in 1995, 2000, 2005, 2010, and 2015 and the results were used to demonstrate the process of land cover change.

Chapter 3 proposes a multi-species approach synthesizing biological traits of two focal species (i.e., *Nyctereutes procyonoides* and *Phasianus colchicus*) to evaluate urban green spaces in Haidian District, Beijing. Specifically, the range of green space for analyses and the distance threshold of landscape connectivity were defined by the habitat type and dispersal ability of each species, respectively. Finally, the most important green space cores and corridors for each species were identified and overlaid to obtain a final evaluation of urban green spaces. The results supported the hypothesis that the evaluation of urban green space should involve specific species and their biological characteristics to achieve a functional evaluation, not just a structural one.

Chapter 4 presents a case study using riparian areas of the Tsing River to clarify the responses of bird communities to environmental variables at both local and landscape scales along the river. Bird surveys were conducted from May 2016 to April 2017 once a month along 18 transect lines, while local habitat characteristics (i.e., vegetation structure and human disturbance) were collected from field surveys conducted in September 2018. In addition, the surrounding land cover and landscape connectivity were extracted and calculated from Gaofen-2 remote sensing images. Finally, a redundancy analysis was applied to identify important environmental variables that significantly affect the biodiversity of bird communities in urban riparian areas. Results showed that local variables tend to have a more significant effect on bird communities, especially the coverage of grassland and the number of pedestrians passing by.

Chapter 6 presents the conclusions. First, influential environmental variables affecting the biodiversity of bird communities are summarized and discussed in comparison with previous studies. Second, suggestions are put forward for future urban green space planning. Moreover, the literature contribution, limitations, and prospects for future research are clarified to obtain a comprehensive understanding of this study.

This study optimized the existing approach of urban green space evaluation by including empirical data of species living in urban areas. Compared with the traditional landscape pattern analysis, this approach provides a more detailed ecological conservation function for urban green spaces. The results suggest that it is necessary to include species information in the evaluation of urban green spaces, as well as their biological characteristics, such as residential type, diet, home range, and dispersal ability. In addition, the influential variables were identified at both the local and landscape scales. In the case of local biodiversity conservation, field and species surveys are needed to clarify the effects of environmental changes. In terms of biodiversity conservation at the regional level, long-term species census data and remote sensing data will contribute to a comprehensive understanding of the impact mechanism of urbanization on species, which may contribute to more environmentally friendly urban planning in the future.

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Chapter 1 Introduction

1.1 Urbanization

Urbanization is a complex process in which small settlements are developed into small towns and then into cities with increasing population and expanding built-up areas (Bloom et al. 2008). The growth of cities is due to a combination of four forces: natural growth, rural to urban migration, massive migration due to extreme events, and redefinition of administrative boundaries (Elmqvist et al. 2013). Although cities as locations of concentrated economic activities offer large and diversified labor pools, urbanization is thought to depress the functions of ecosystems, for example, biodiversity conservation (Turner et al. 2004) and climate regulation (Oleson et al. 2015). A recent study by Zhang et al. (2018) found that urbanization intensified extreme precipitation and flooding.

From a global perspective, the increasing proportion of urban population has resulted in an increase in built-up areas, such as buildings and roads, which have become one of the fastest growing land-use types. According to the United Nations (2018), there will be 43 megacities with a population of more than 10 million in the world by 2030. By 2018, urban population accounted for 55% of the total global population, and it is expected to increase to 68% by 2050. From 1950 to 2018, urban population increased from 751 million to 4.2 billion, with a large proportion of the increase occurring in Asia and Africa. At present, 82% of the population in North America is urban, making it the most urbanized area in the world. In Europe, 72% of the population lives in cities, while in Asia, which has the world's largest population, the proportion of urban population is only 50%.

Compared with developed countries in Europe and America, urbanization in China started late but progressed rapidly. However, the speed of urbanization in China has been the fastest among all countries in recent years. By 2050, the urban population in China is expected to increase by 255 million. According to the National Bureau of Statistics of China (2018), the total population of China has reached 1.39 billion by 2018. From 2000 to 2018, the population of permanent urban residents increased from 459 million to 831 million, while the population of rural residents decreased from 808 million to 564 million. Since 1985, urbanization in China has been going through a rapid development stage. In 2018, the urbanization rate rose to 60% (**Figure 1-1**).

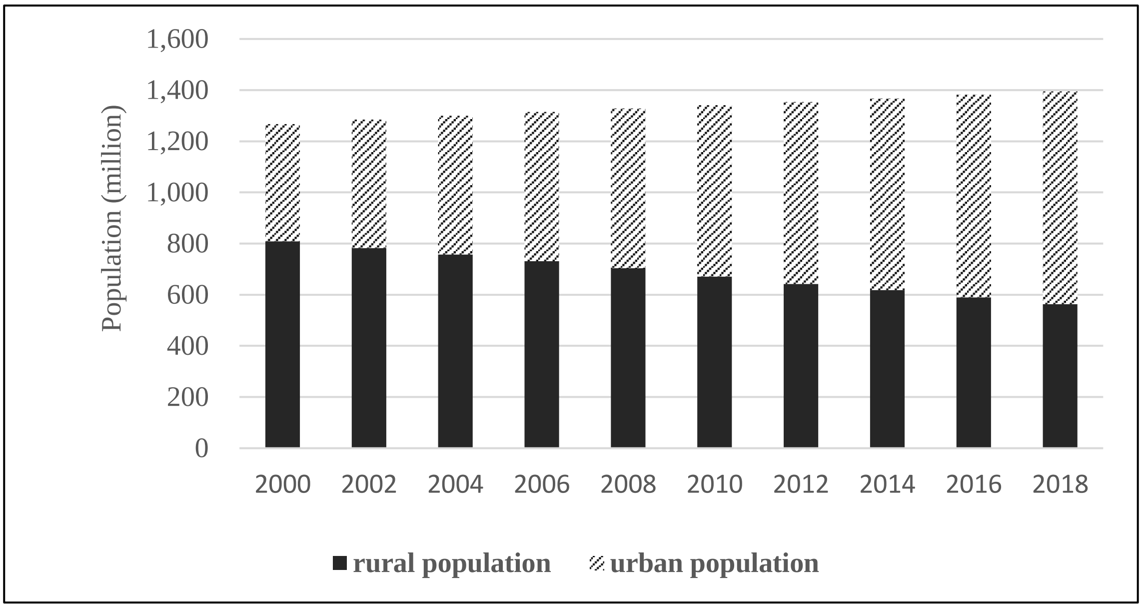


Figure 1-1 Changes of urban and rural population in China (2000-2018).
Data source: National Bureau of Statistics of China (http://www.stats.gov.cn/tjsj/).

1.2 Urban green space

1.2.1 Definition of urban green space

There is no general agreement on a definition of what is urban, and considerable differences exist in the classification of urban and rural areas among countries and continents. In Europe and North America, urban is often defined as an area with human agglomerations and with more than 50% of the surface built, surrounded by areas with 30-50% of the surface built, and an overall population density of more than ten individuals per hectare (Elmqvist et al. 2013). In China, urban areas are defined as residential areas characterized by nonagricultural industries and nonagricultural population agglomerations, including cities and towns established according to the Administration Division (State Bureau of Quality and Technical Supervision 2006).

Urban green spaces are often among the few, or only, places where people can experience nature in the city (Haaland and van Den Bosc 2015). There are various definitions of urban green space in existing research. For example, some studies include green space in suburban areas, while others do not, and there is a debate on whether private gardens should be regarded as urban green spaces. Urban green space is widely defined as “any vegetation found in the urban environment, including parks, open spaces, residential gardens, green roofs, or street trees” (Kabisch and Haase 2013; Wolch et al. 2014). According to the *Standard for Classification of Urban Green Space* [CJJ/T85-2017] in China, green spaces are generally divided into parks, green buffers, public squares, and green spaces attached to public facilities. This classification of green space is effective for urban planning and management, but it ignores the ecological characteristics of green space, such as primary or secondary forests, which are critical for some species living in urban areas.

Landscape is another important term in this study. Studies in the field of animal ethology and ecology tend to focus on a certain type of habitat or homogenous green space; for example, Fartmann et al. (2018) investigated the impacts of Christmas tree plantations in an intensively used low-mountain landscape on bird assemblages. In contrast, the term landscape in the field of spatial planning is often used to describe green spaces, including various types of habitats (Baguette et al. 2013; Englund et al. 2017). Recently, the definition of landscape has been broadened to not only include physical space but also intangible space, such as cultural landscapes (Li and Mell 2019).

Similar to many studies in landscape ecology, in this study, green space is defined from the perspective of land cover classification extracted from remote sensing images (Buyantuyev and Wu 2010; Li and Yang 2015). It includes outdoor spaces with significant amounts of vegetation and natural or semi-natural areas, including forests, grasslands, and wooded areas composed of forests, shrubs, and grass (Kong et al. 2010). In particular, although agricultural lands and waterbodies are not included in green space planning, they are sometimes regarded as green spaces when they refer to habitats of certain species, especially in landscape ecology studies (Canedoli et al. 2018). In Chapter 3, we define green space according to species habitat type.

1.2.2 Current situation of urban green spaces in China

A review by Haaland and van Den Bosch (2015) shows that there is increasing evidence that urban green spaces are gradually decreasing and becoming fragmented because of global urbanization, especially in Asia and Australia, followed by Europe and North America. However, in some areas, urban green spaces are increasing. There are two possible explanations for this exception. First, urban green spaces are created through the process of urbanization where undeveloped areas in the urban periphery are transformed into urbanized areas, such as in rapidly sprawling cities in China and Europe (Kong and Nakagoshi 2006; Xu et al. 2011; Zhao et al. 2013). Second, some countries, such as China and Singapore, conduct strategic spatial planning encouraging conservation and construction of urban green spaces (The National Bureau of Statistics of China: http://www.stats.gov.cn/; Department of Statistic Singapore: https://www.singstat.gov.sg/). In other words, the establishment of urban green spaces is a part of urbanization.

Table 1-1 Urban green space (UGS) in China (2008-2017)

	Area of UGS (million hectares)	Coverage of UGS (%)	Number of Urban Parks	Area of Parks (million hectares)
2008	174.8	37.4	8,557	0.22
2009	199.3	38.2	9,050	0.24
2010	213.4	38.6	9,995	0.26
2011	224.3	39.2	10,780	0.29
2012	236.8	39.6	11,604	0.31
2013	242.7	39.7	12,401	0.33
2014	252.8	40.2	13,037	0.35
2015	267.0	40.1	13,834	0.38
2016	278.6	40.3	15,370	0.42
2017	292.1	40.9	15,633	0.44

According to data from the National Bureau of Statistics of China, with the development and construction of China, the area of urban green spaces and the number and area of parks have been increasing significantly since 2008 (**Table 1-1**). This stems from research progress in the fields of ecology and biology, which brought global attention to environmental conservation, especially in highly urbanized areas (Haaland and van Den Bosch 2015).

1.2.3 Necessity of evaluating urban green space

Urban green spaces provide various benefits for humans and other organisms, including biodiversity conservation, water management, air quality, human health and well-being, social cohesion, and tourism (Konijnendijk et al. 2013). However, the contradiction between the demand for green space conservation and the shortage of urban land resources has become one of the most challenging problems in densely populated areas (Haaland and van Den Bosch 2015).

Over the past decade, a growing number of studies have quantified the function of biodiversity conservation in urban green space (Humphrey et al. 2015). This is not only due to the increasing effects of urbanization on natural ecosystems, but also because an increasing number of researchers recognize that urban green spaces are a hotspot of biodiversity that should be protected in innovative ways (Savard et al. 2000). Therefore, especially in developing cities, such as Beijing, conflicts between urbanization and habitat protection can be alleviated by more environmentally friendly green space planning based on evaluating the ecological efficiency of urban green spaces.

1.3 Landscape ecology: a way of evaluating urban green space

1.3.1 Landscape ecology

In the past two decades, studies in the field of landscape ecology have attempted to evaluate the ability of green space to conserve biodiversity: that is, ecological efficiency (McGarigal et al. 2002). Metapopulation ecology, on the other hand, focuses on the relationship between population dynamics and migration, colonization and extinction events in spatially structured habitats (Murphy and Lovett-Doust 2004). Compared with metapopulation ecology, which tends to ignore environmental factors beyond the habitat itself, landscape ecology pays more attention to the importance of spatial configuration for ecological processes (Wiens et al. 1993, 1995).

Landscape ecology was introduced in North America, Australia, and China in the early 1980s, but the roots of it are in central and eastern Europe where the environment, biota, and human settlements were viewed as interconnected by biogeographers. Carl Troll, a German biogeographer, coined the term "landscape ecology" as a special viewpoint for understanding complex natural phenomena (Troll 1950, translated in Wiens et al. 2007). It integrates multidisciplinary knowledge from geography, landscape architecture, regional planning, economics, and forestry, and emphasizes broad spatial scales and ecological effects of the spatial patterning of ecosystems (Turner, 1989).

1.3.2 Landscape connectivity

Landscape models are usually proposed to represent the complex natural landscape; in other words, they simplify the real environment. A classic model in landscape ecology is the Patch-Corridor-Matrix model (**Figure 1-2**), which was proposed to describe how landscape patterns influence ecological processes (Forman and Godron, 1981). A patch is defined as a relatively homogeneous area that differs from its surroundings (Forman, 1995); a corridor is the link connecting patches; and matrix is the background ecological system surrounding patches (Wu and Hobbs, 2007). Ecological networks are typically conceptualized as a suit of core patches of habitats connected by corridors and smaller stepping-stone patches (Lawton et al. 2010).

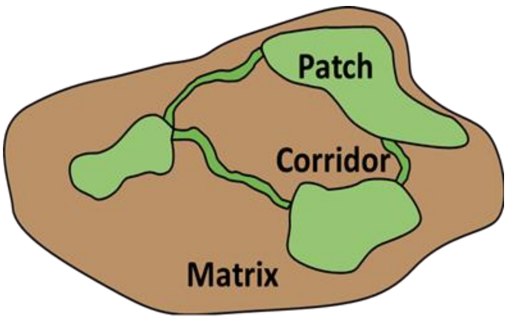


Figure 1-2 The *Patch-Corridor-Matrix* model

Landscape fragmentation, a consequence of urban expansion, is defined as habitat loss through the spread of artificial surfaces and the breakup of the remaining habitat areas into separate patches (Burkey, 1989; Forman and Godron, 1981). Landscape fragmentation is known to constrain movement of animals between habitats and can result in high extinction probabilities of local populations in small isolated patches, which represents a great threat to biodiversity (MacArthur and Wilson, 2001).

On the contrary, landscape connectivity, the opposite of landscape fragmentation, is defined as the degree to which the landscape facilitates or impedes the movement of species among resource patches (Taylor et al. 1993). It is used to evaluate the ecological service of a certain landscape by quantifying landscape patterns from a macro point of view (Tischendorf and Fahrig, 2000; Urban and Keitt, 2001; Bunn et al., 2000). Landscape connectivity simulates ecological processes based on graph theory. The connections between habitat patches are characterized through a probabilistic model, in which there is a certain probability of dispersal among habitat patches, typically modeled as a decreasing function of inter-patch Euclidean or effective distance (Avon and Bergès, 2016; Saura and de la Fuente, 2017). Numerous landscape metrics and models have been developed to quantify landscape configuration and evaluate ecosystem services and forest management (Uuemaa et al. 2013; Reis et al. 2016; Fardila et al. 2017). A review by Kindlmann and Burel (2008) explains various metrics of landscape connectivity and their mathematical connotations.

The development of ecological networks is viewed as an important measure for reversing the effects of fragmentation on biodiversity and is now a major policy driver in green space planning (Humphrey et al. 2015). An increasing number of ecologists have collaborated with other scientists, planners, and engineers to take part in interdisciplinary research focusing on the biodiversity conservation of urban green spaces (Lepczyk et al. 2017). The amount of landscape planning or green space planning using landscape connectivity metrics or models is increasing (Bunn et al. 2000; Saura and Pascual-Hortal, 2007; Saura et al. 2011; Correa et al. 2016).

Table 1-2 Definition of common used terms in landscape ecology (Turner et al. 2015)

Term	Definition
Composition	What and how much is present of each habitat or cover type.
Configuration	A specific arrangement of spatial elements; often used synonymously with spatial structure or patch structure.
Connectivity	The spatial continuity of a habitat or cover type across a landscape.
Corridor	A relatively narrow strip of a particular type that differs from the areas adjacent on both sides.
Fragmentation	The breaking up of a habitat or cover type into smaller, disconnected parcels; often associated with, but not equivalent to, habitat loss.
Heterogeneity	The quality or state of consisting of dissimilar elements, as with mixed habitats or cover types occurring on a landscape; opposite of homogeneity, in which elements are the same.
Landscape	An area that is spatially heterogeneous in at least one factor of interest.
Matrix	The background cover types in a landscape, characterized by extensive cover and high connectivity; not all landscape has a definable matrix.
Patch	A surface area that differs from its surroundings in nature or appearance.

1.4 Evaluating urban green space from the perspective of species

1.4.1 Biodiversity in urban areas

Urban biodiversity is “the variety or richness and abundance of living organisms (including genetic variation) and habitats found in and on the edge of human settlement” (Müller et al. 2013). The impact of environmental changes on biodiversity is the main topic of urban ecology, which is also meaningful for decision-makers. Although urbanization has a variety of impacts on many biological groups, urban green spaces still contain a large part of global biodiversity (Sandström et al. 2006). However, biodiversity is a broad concept that includes all biotic variation in genes, species, communities, and ecosystems from multiple dimensions; thus, it is difficult to measure the loss of biodiversity.

1.4.2 Quantification of biodiversity

Based on the literature review, we concluded two ways of quantifying biodiversity. One is to use the abundance of a certain species to represent the overall biodiversity, i.e. focal species (Lambeck, 1997; Sanderson et al. 2002). Focal species are assumed to have similar requirements for habitats with many other species; thus, the fluctuation of focal species is regarded as the response of the whole to environmental changes (Hess and King 2002). For example, conservation strategies for white-backed woodpeckers in Finland and Russia are also effective for 80% of the threatened beetle populations living in the same forests (Martikainen et al. 1998). This approach allows for quick and easy quantification with limited information. However, the selection of focal species has been a challenge for scientists as it relies on a variety of analyses and verification (Lambeck, 1997).

Another way is to investigate the composition of a group of species, such as birds, bees, mammals, amphibians, or vascular plants (Humphrey et al. 2015). Birds are often selected as an indicator of biodiversity because their autecology is well studied, and they are relatively easy to observe and identify in urban areas (Liang et al. 2018). As highly mobile species, birds are relatively sensitive to the environment, and they are particularly responsive to environmental changes at different scales (Cunningham et al. 2014; Burgess and Maron 2016; Guttery et al. 2017).

There are various metrics to describe the change in biodiversity. Villéger et al. (2016) attempted to identify a nonredundant set of key biodiversity metrics within 12 common metrics (**Table 1-3**), which included four aspects: abundance, taxonomic diversity, functional diversity, and phylogenetic diversity. Total abundance and geometric mean abundance are both typical ways of quantifying abundance. Total abundance is the sum of the population of all species, while geometric mean abundance reflects relative rather than absolute abundance by using a multiplicative calculation. Taxonomic diversity is the most typical way to quantify biodiversity based on the number of species and their relative abundances, such as species richness, Shannon diversity index, and Simpson Diversity Index. Functional traits are “the morphological, physiological, or phenological characteristics of an organism that strongly influence its performance" (McGill et al. 2006). Functional diversity represents the distribution of the functional traits of the organisms present in an assemblage (Villéger et al. 2008), such as functional richness, functional divergence, and functional evenness. Phylogenetic diversity is calculated based on the total length of the phylogenetic branches connecting all species of a given assemblage (Faith, 1992), which describes its evolutionary divergence.

Table 1-3 Overview of the biodiversity metrics, adapted from Villéger et al. (2008, 2016).

Biodiversity metric (abbreviation)	Description
Total abundance (TA)	$\sum_{i=1}^{i=n} a_i$
Geometric mean abundance (GMA)	$\left(\prod_{i=1}^{i=n} \ln(a_i + c)\right)^{1/n}$
Species richness (SR)	Total number of species present.
Shannon index (Shan)	$\exp(-\sum_{i=1}^{i=\mathrm{SR}} p_i \cdot \log p_i)$
Simpson index (Simp)	$\frac{1}{\sum_{i=1}^{i=n} p_i^2}$
Functional richness (FRic)	The convex hull volume of the individual species in multidimensional trait space
Functional evenness (FEve)	The regularity with which species abundance is distributed along the minimum spanning tree which links all the species in the multidimensional functional space.
Functional divergence (FDiv)	Species deviance from the mean distance to the center of gravity within multidimensional trait space, weighted by relative abundance.
Functional dispersion (FDis)	The weighted mean distance in multidimensional trait space of individual species to the centroid of all species. Weights are species' relative abundances.
Proportion of carnivores (PC)	The abundance of carnivorous species (i.e., species with a diet consisting of at least 60% meat, fish, and /or carrion) relative to the total abundance of all species (%).
Community-weighted mean body mass (CWMm)	The mean body mass of species weighted by the species' abundances.
Phylogenetic diversity (PD)	The total length of the phylogenetic branches connecting all species of a given assemblage.

Villéger et al. (2016) compared these biodiversity metrics and found that total abundance and the Shannon Index can represent biodiversity change at a considerable level; species richness and metrics of functional diversity showed different importance in explaining the variance of biodiversity. Therefore, this study only selected three biodiversity metrics to describe the biodiversity of birds: bird abundance (BA), bird richness (BR), and bird diversity (BD) (Shih, 2018). BA refers to the sum of all bird individuals, BR is the number of bird species, and BD is calculated using the Shannon Index, which reflects the evenness of species composition:

$$BD = -\sum_{i=1}^{S}(p_i \ln p_i)$$

where p_i is $a_i/\sum a_i$, a_i is the number of the ith specie, and i is one through S.

1.4.3 Level, scale, and biological characteristics of species

To evaluate urban green space from the perspective of species, it is essential to establish a model between changes in biodiversity and in the environment. Before measuring environmental changes, it is necessary to define the boundary of the investigated areas. Species likely go through a hierarchy of decisions when selecting a habitat (Johnson 1980, Hutto 1985); thus, it is necessary to understand the size of the area evaluated by different species when selecting a habitat.

The grain and extent of data used in any analysis of landscape patterns influence the numerical results obtained for a given metric and must be defined explicitly (Turner et al. 1989; Wu 2004). Although “level” and “scale” are often used inconsistently in existing research, some researchers (Mayor et al. 2009, Wheatley and Johnson 2009, and McGarigal et al. 2016) made a critical distinction between them. The term “level” can describe cases when a) the investigated environment is hierarchically organized in space (e.g., forest, tree, leaf) or time (e.g., annual, seasonal, daily light cycles), or b) the behavior of targeted species is regarded as hierarchical in space (e.g., dispersal range, home range, resource patch) or time (e.g., generation, breeding cycle, foraging period). In comparison, the definition of “scale” is broader than “level”, including organizational scale (i.e. “level”) and absolute scale in space (500 m, 1000 m, 3000 m, and so on) or time (one day, one month, one year). Based on the above definition, in this study, we use “local level” and “regional level” to distinguish the hierarchy of environmental variables, such as a riparian habitat and a city, and “multiscale” to refer to multiple levels and multiple absolute scales.

To identify the best scale at which the response to an environmental variable is the most significant, called “the scale of effect” (Jackson and Fahrig, 2012), it is necessary to build a multiscale research framework (Wiens, 1989; Levin, 1992). Different taxa will perceive and respond to landscape structure at multiple scales, depending on a range of parameters, such as body size, life history characteristics, life stage, and season (e.g., birds defend a smaller home range when breeding than when foraging more widely in winter) (Hostetler and Holling, 2000). In addition, the scale of sampling can confound ecological processes (Goddard et al. 2010). For example, the relationship between human population and biodiversity may depend on the study extent: negative at fine scales, where built-up areas occupy habitats, and positive at large scales, where both humans and wildlife tend to live in places with abundant resources (Pautasso, 2007). Therefore, to properly consider the scale dependencies of a large number of taxa, it is necessary to discuss the effects of environmental variables at multiple scales.

1.5 Research gap

1.5.1 Utilization of long-term empirical data

As explained in Section 1.4.2, one way to quantify biodiversity is using census data of a group of species. Some studies use virtual instead of real species, whose habitat selection can cover all kinds of ecosystems, to model connectivity networks of green spaces for each species (Girardet et al. 2013; Mimet et al. 2016; Sahraoui et al. 2017). The result is regarded as the evaluation of green space from the perspective of the whole group of species. Some studies have investigated the response of a certain group of species (e.g., species composition, bird diversity, bird abundance, and bird richness) to various environmental variables (Banville et al. 2017). Influential environmental variables are then identified as a reference for biological conservation and green space planning.

However, most studies focus on an individual or a group of species with empirical data of less than three years or even a breeding season, which limits the application of research results to the evaluation and planning of green space (Xu et al. 2018; Garcês et al. 2019). Long-term monitoring helps to understand the temporal and spatial impact of environmental changes on biodiversity (Baumgardt et al. 2019). It has substantial value for detecting relationships between populations of species and environmental changes, and provides further suggestions for green space management regarding biodiversity conservation (Pollock et al. 2002).

Several countries have national long-term monitoring programs, such as the North American Breeding Bird Survey (https://www.pwrc.usgs.gov/bbs/) and the Japan Long Term Ecological Research Network (http://www.jalter.org/en/). Species data in these programs are collected by professionals in a structured way, which is time- and labor-intensive. Therefore, many studies have been compromised by the limited availability of long-term data, which motivated the development of citizen science data. Many citizen-based observation networks have been established to gather information on a diverse array of taxa and ecological processes (Sullivan et al. 2009). Citizen science data could fill the research gap in studies integrating large-scale environmental changes and long-term biodiversity changes. However, the application of citizen science data in this direction is still limited.

1.5.2 Limited empirical evidence of landscape connectivity

In the past decades, although models and indices of landscape patterns based on graph theory have been developed and widely used in landscape ecology, uncertainties remain in the interpretation of biodiversity variance (Grafius et al. 2017). Local variables, such as history and vegetation composition of habitats, have been shown to affect biodiversity (Huang et al. 2015; Banville et al. 2017). A review by Humphrey et al. (2015) summarizes the responses of individual species and communities to both local and regional variables. Among the 104 selected articles, 40 articles showed a negative effect on patch isolation. However, there are various approaches to measure patch isolation and species response (e.g., bird richness), and the mathematic relationship between landscape connectivity and patch isolation is blurred. Moreover, local variables also have an impact on biodiversity, which should be separated from the overall effects to show the role of landscape connectivity. Unfortunately, studies comparing empirical observations of species with landscape connectivity remain limited.

1.5.3 Limited multiscale syntheses of environmental variables

Although ecological analyses of green spaces have benefited from the development of remote sensing technology, there is still a debate on the relative importance of local versus regional environmental variables affecting biodiversity. For example, some avian studies have suggested that local variables are more influential than regional variables in describing the variation of bird species richness in urban green spaces (Evans et al. 2009). However, some studies show that the heterogeneity of the surrounding environment is significant (Kadlec et al. 2008), because there are various habitat types satisfying different species. This debate and uncertainty may hinder the effectiveness of the concept of ecological networks.

When it comes to the relationship between species and environment, it is necessary to define the scale or level at which the relationship is described. On the one hand, the species-habitat relationship is the basis of discussion at local scales in relation to wildlife reserve planning (Carbó-Ramírez and Zuria 2011; Fardila et al. 2017). On the other hand, regional analyses are usually related to ecological network principles in landscape ecology, developed from the island biogeography theory (MacArthur and Wilson 2001; Margaritis and Kang 2017). Only if the management scale matches the scale of ecological patterns and processes, can conservation actions be effective (Sanderson et al. 2002). However, studies covering the impacts of environmental changes on biodiversity at both, the local and regional, levels are still lacking (Humphrey 2015).

1.6 Aims, objectives, and research framework

This study aims to examine a new approach to urban green space evaluation, which is based on the identification of influential environmental variables that best describe the variation in the diversity of bird species living in urban areas. This study is based on the hypothesis that although landscape connectivity is a widely used indicator to represent the ecological efficiency of green space, the evaluation of urban green spaces should involve ecological processes, such as species response to environmental changes. The species-environment relationship should be investigated at multiple scales/levels as it contains biological characteristics of a group of species and various environmental variables. The objectives of this study are as follows:

1) to examine the advantages and necessity of introducing species in evaluating urban green space, compared with the widely applied landscape connectivity assessment (presented in

Chapter 3).

2) to explore the relationship between the variation in biodiversity and environmental changes at both the local and regional levels, using bird communities in riparian areas of the Tsing River in Beijing (presented in Chapter 4).

3) to evaluate urban green space in central Beijing after identifying influential regional environmental variables that can best describe the variation of bird communities based on long-term bird monitoring data (presented in Chapter 5).

The research framework is shown in **Figure 1-4**. Chapter 1 introduces the academic background, aims, and objectives. Chapter 2 explains the current situation in the study area: Beijing, China. Chapters 3-5 describe the process of including species in urban green space evaluation. Chapter 3 applies the widely used landscape connectivity metric with two focal species as a case study. Chapter 4 uses one-year empirical bird data (May 2016-April 2017), and clarifies the responses of bird communities to environmental variables at both the local (i.e., vegetation composition and human disturbance) and regional levels (i.e., land cover and landscape connectivity surrounding the waterfront). Chapter 6 summarizes the results, literature contribution, and limitations of this research. Additionally, suggestions are put forward for future work.

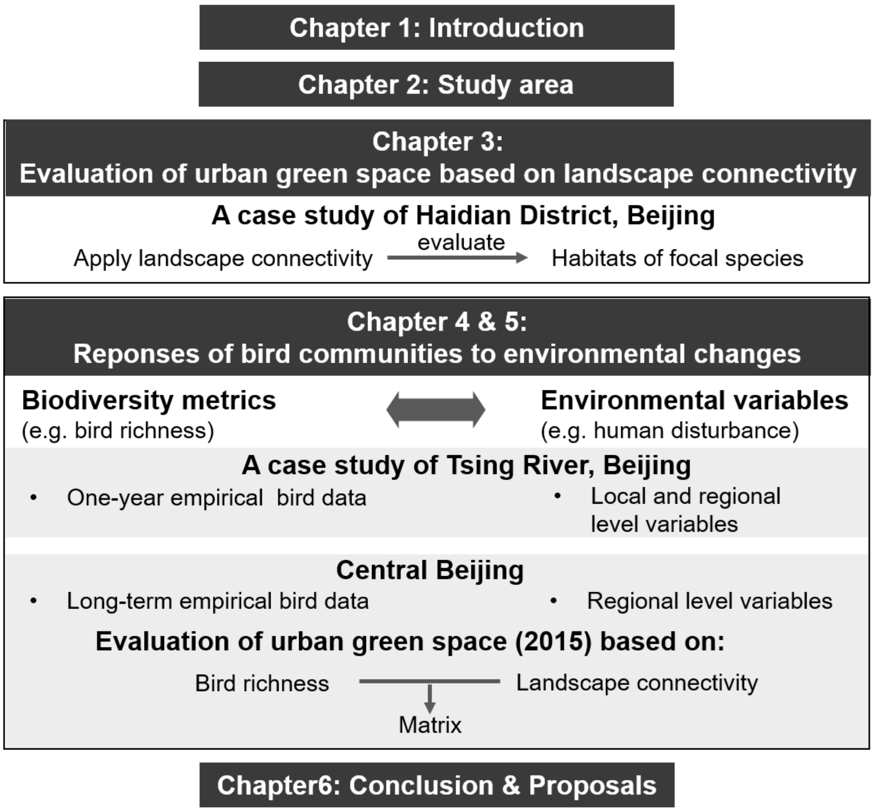


Figure 1-3 The research framework

Chapter 2 Study area

2.1 Study area

2.1.1 Location and topography

Beijing (39°38′–41°05′N) is in the north of China and occupies a total area of 16,410 km^2. The city lies on a plain surrounded by mountains on three sides. Its administrative area comprises 16 districts that are divided into three zones in *the Master Plan of Beijing (2016-2035)*: the urban area (Dongcheng, Xicheng, Chaoyang, Fengtai, Haidian, and Shijingshan), the suburban area (Changping, Daxing, Fangshan, Tongzhou, and Shunyi), and the outer suburban area (Huairou, Mentougou, Pinggu, Miyun, and Yanqing). The area of the central city is 1369 km^2, accounting for 8.34% of the total area of Beijing. Among them, Chaoyang District is the largest, accounting for 455 km^2, and Dongcheng District is the smallest, accounting for 42 km^2. The main development pattern is “single center-ring circuit” with the Forbidden City acting as the single center, ringed by five highways.

Beijing has an average altitude of 43.5m, plain altitude of 20-60m and mountain altitude of 1000-1500m, with an overall trend of higher in Northwest and lower in Southeast. Mountainous areas (10,200 km^2) are in the western and northern parts of Beijing, including hills and shallow transition areas that connect mountainous areas and plain areas (6,200 km^2). The overall trend is higher in the northwest and lower in the southeast.

There are five main river system in Beijing: Juma River System, Yongding River System, North Canal River System, Chaobai River System and Ji Canal River System. Most of them originated from the Northwest Mountainous Areas and meandered southeast through the plain areas and finally joined the Bohai Sea in the Haihe River (except Ji Canal River System). According to the main river systems, there are more than 80 rivers in Beijing. Among them, there are mainly four rivers flowing through the central urban area, namely, Qing River, Ba River, Tonghui River, Liangshui River and more than 120 related tributaries.

2.1.2 Climate and precipitation

The climate in Beijing is a typical semi-humid continental monsoon climate in the north temperate zone. The average annual temperature is 13.1°C. The highest temperature is seen in July, at an average of 26.9°C. The lowest temperature is seen in January, at an average of -2.9°C. In terms of precipitation, the spatial and temporal distribution of annual precipitation in Beijing is uneven. The northeastern and southwestern Piedmont windward slopes are relative precipitation centers, with precipitation ranging from 600 to 700 mm, the northwestern and northern deep mountains get less than 500 mm, and the plains and some mountainous areas get between 500 and 600 mm. 80% of the annual precipitation is concentrated in the summer months of June, July and August. Among them, the highest precipitation for the year is in July and the lowest is in December.

2.2 Urbanization in Beijing

2.2.1 Population and economy

Beijing, the capital of China, is a typical case of rapid urbanization. Since 2000, the gross domestic product (GDP) in Beijing has experienced significant growth **(Figure 2-1**). In addition, there was a continuous increase in the population of permanent residents from 2000 to 2014, and the population remained stable from 2014 to 2018.

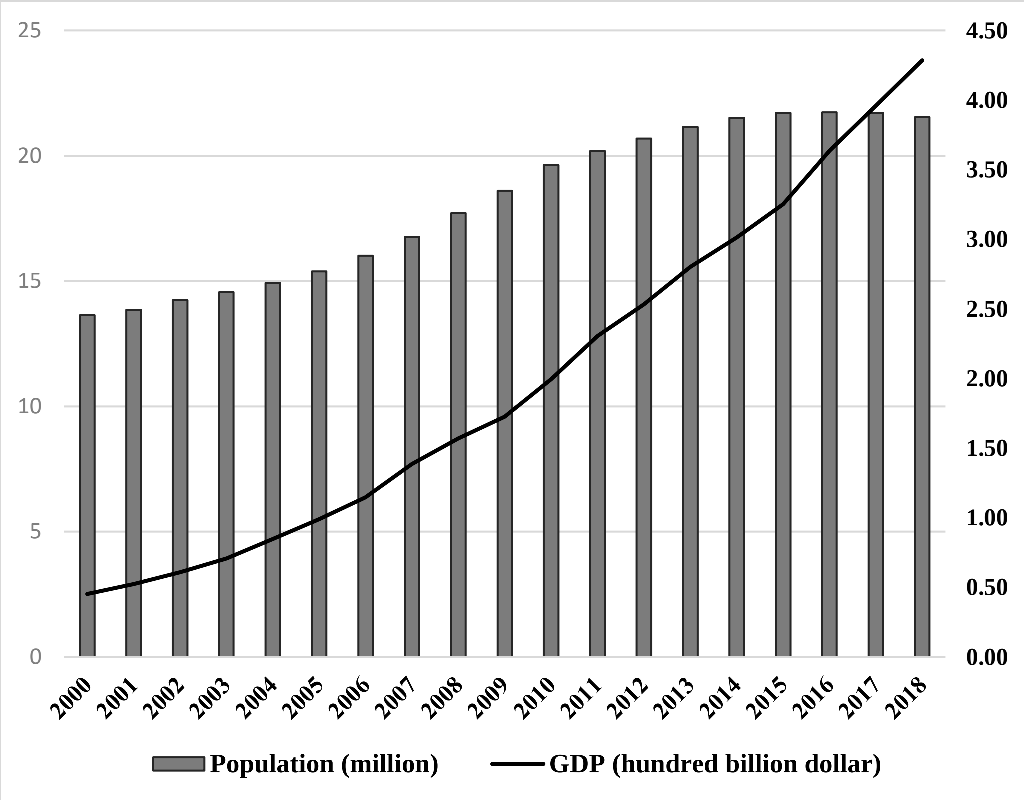


Figure 2-1 The population and GDP of Beijing (2000-2018)

2.2.2 Land cover change (1995-2015)

Historical changes in land cover of Beijing were interpreted using remote sensing images from 1995, 2000, 2005, 2010, and 2015 (**Figures 2-2, 2-3, 2-4**). Detailed information of remote sensing data is shown in **Table 2-1** (downloaded from Geospatial Data Cloud: http://www.gscloud.cn/search). Object-based image analysis (OBIA) was applied to extract land cover information, performed in ENVI and eCognition. Indexes such as the Normalized Difference Vegetation Index (NDVI), geometry, and brightness were included in the unsupervised classification. Five land cover types were identified: green space (including forest and grassland), agricultural land, waterbody, built-up area, and bare land. A visual analysis of the images, as shown on **Figure 2-5**, indicates that the area of green space is decreasing, while that of built-up area is increasing. In 2010, the area of bare land increased significantly, but decreased to the level of 2005 in 2015. This may be due to rapid urbanization and dense construction activities around 2008, when the Olympic Games were held in Beijing.

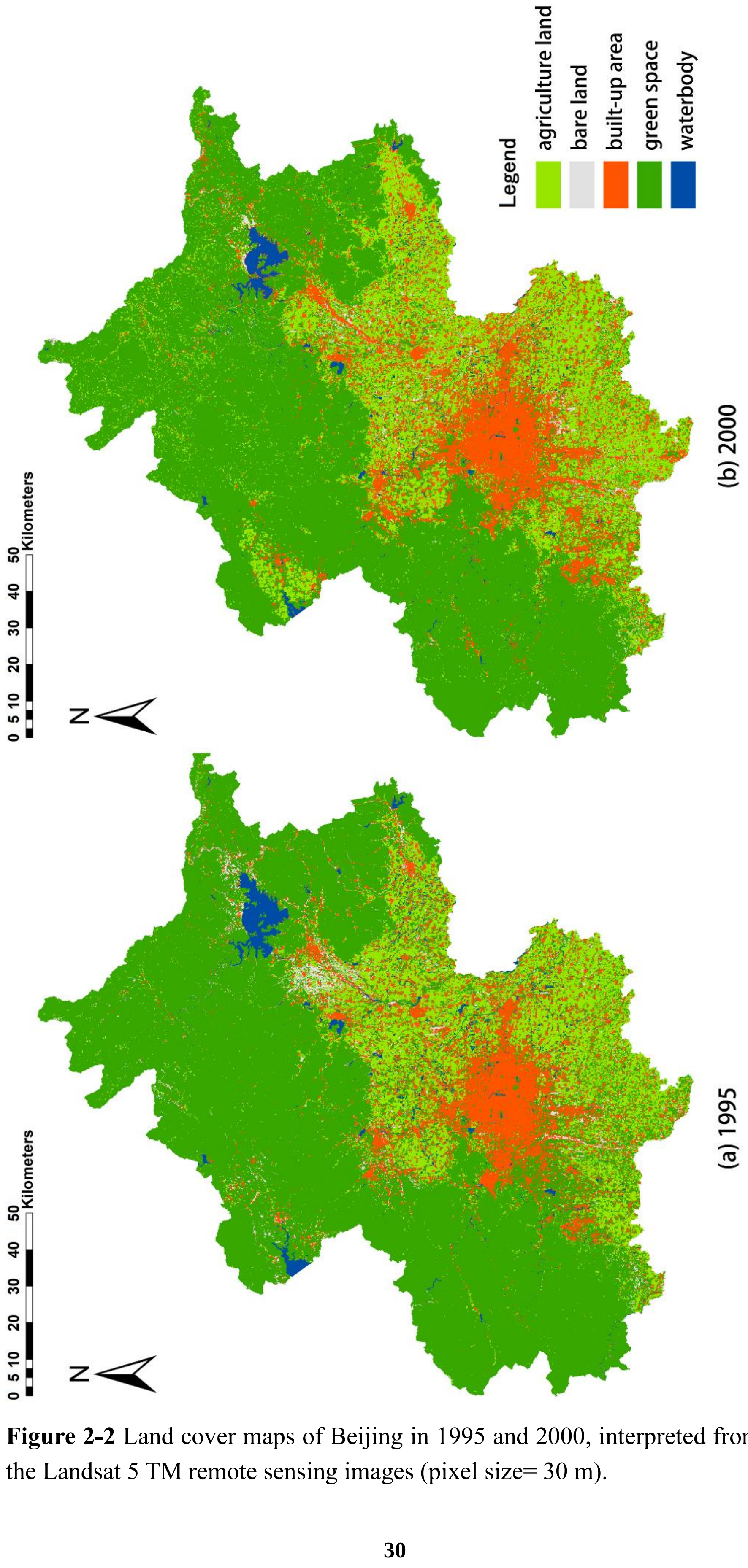


Figure 2-2 Land cover maps of Beijing in 1995 and 2000, interpreted from the Landsat 5 TM remote sensing images (pixel size= 30 m).

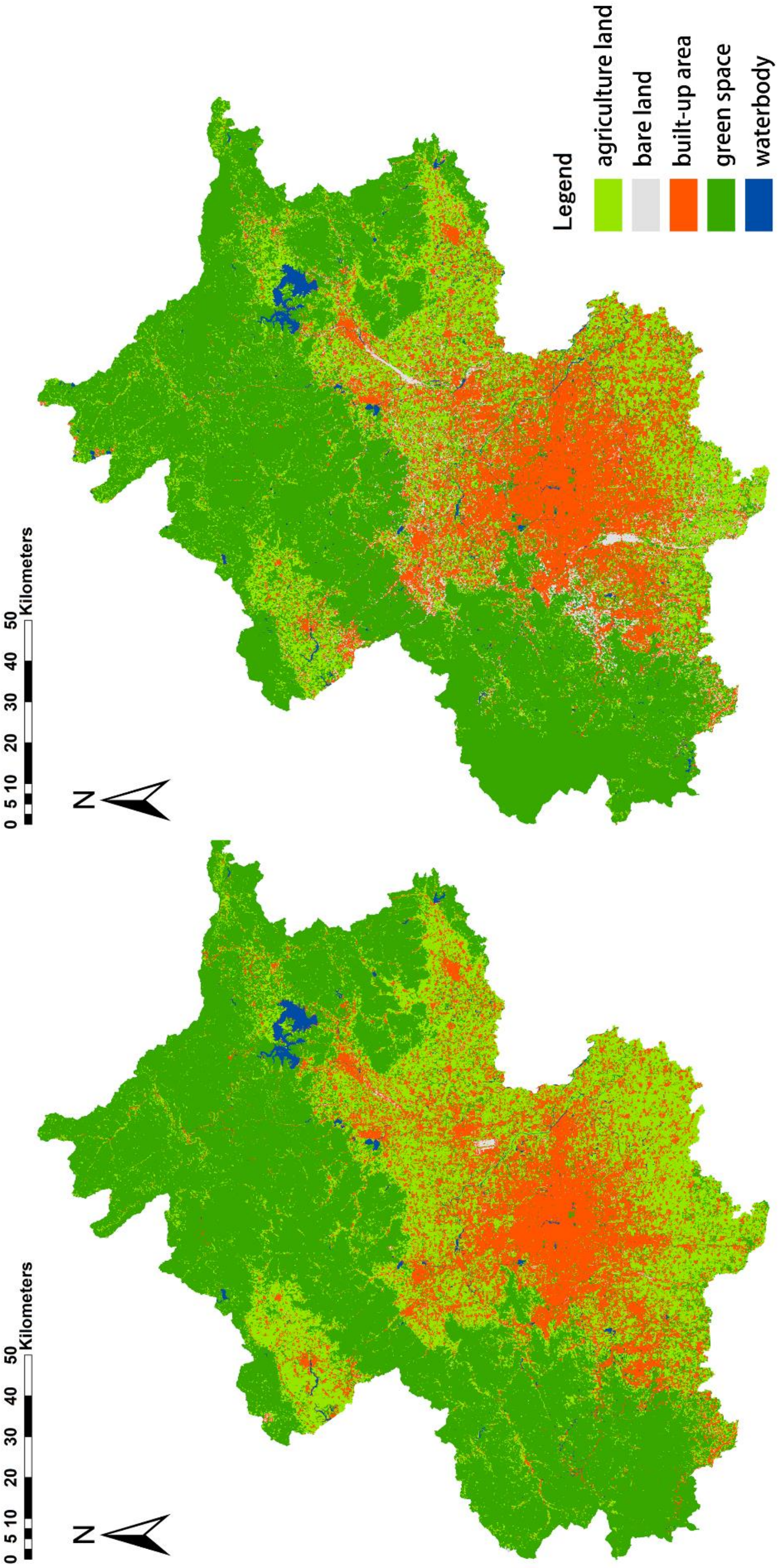


Figure 2-3 Land cover maps of Beijing in 2005 and 2010, interpreted from the Landsat 7 ETM remote sensing images (pixel size= 30m).

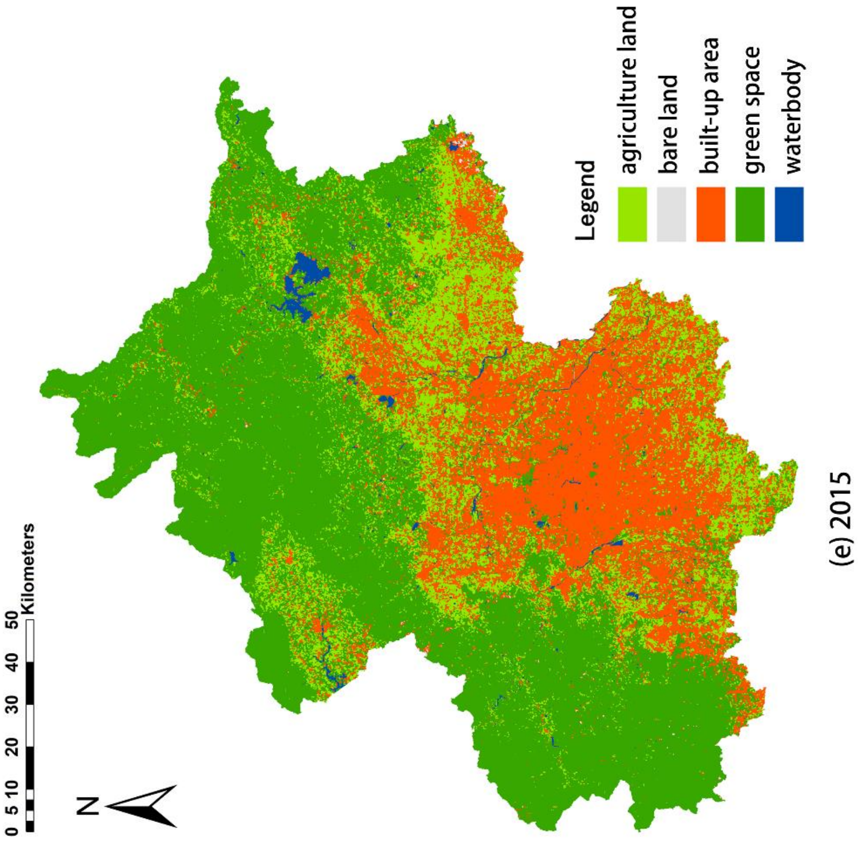


Figure 2-4 The land cover of Beijing in 2015, interpreted from the Landsat 7 ETM remote sensing images (pixel size= 30m).

Table 2-1 Information and descriptions of remote sensing data of Beijing (1995-2015)

Date	Type	Number of Bands	Resolution
1995/09/06	Landsat 5 TM	7	Band 1-5,7: 30 m; Band 6: 120 m
2000/09/13	Landsat 5 TM	7	Band 1-5,7: 30 m; Band 6: 120 m
2005/09/03	Landsat 7 ETM	8	Band 1-7: 30 m; Band 8: 15 m
2010/08/16	Landsat 7 ETM	8	Band 1-7: 30 m; Band 8: 15 m
2015/09/15	Landsat 7 ETM	8	Band 1-7: 30 m; Band 8: 15 m

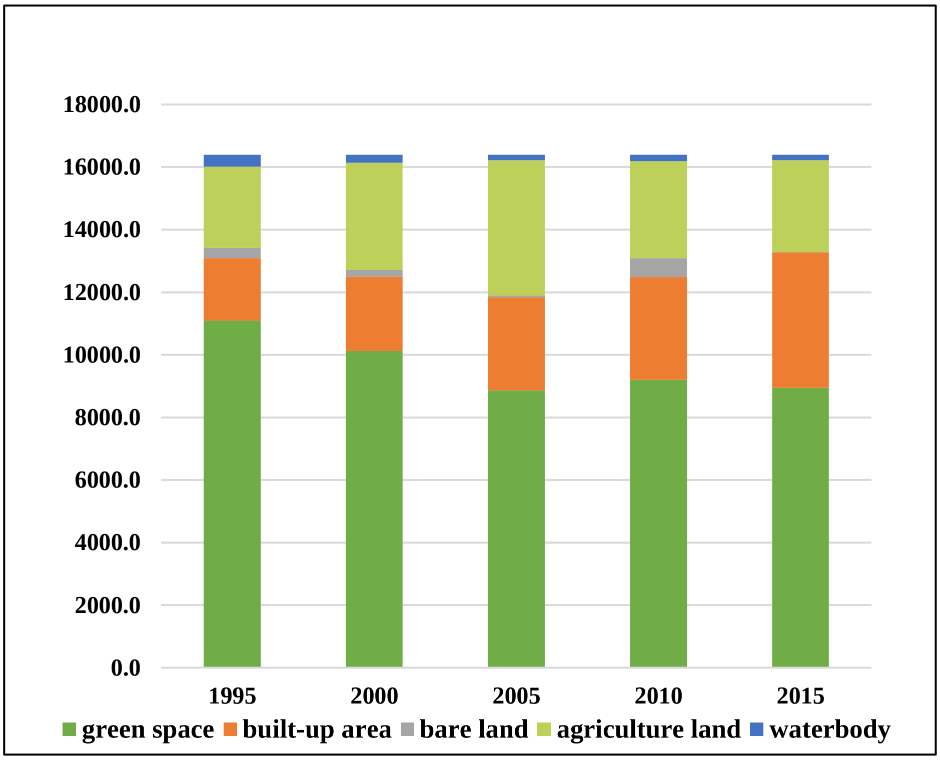


Figure 2-5 Land cover changes of Beijing (1995-2015) (km^2).

2.3 Biodiversity conservation in Beijing

2.3.1 Current situation

Biodiversity is the general term for all species, intraspecific genetic variation, and their living environment, including different kinds of vertebrates, vegetation, and microorganisms as well as the genes they possess, the ecosystem composed of them, and their living environment. This section presents vegetation and animal diversity as examples to introduce the status of biodiversity in Beijing.

A vegetation survey, conducted from January 2008 to December 2010, was the main part of a project called "Investigation and Evaluation of Main Plant Species Resources in Beijing". It was chaired by the Beijing Landscape and Greening Bureau and carried out an investigation and cataloging of vegetation in Beijing. The investigation and statistics showed that there are 140 families, 654 genera, and 1790 species of wild vascular plants in Beijing. Among them, there are approximately 803 species of garden plants belonging to 407 genera and 113 families in central

Beijing. There are 507 species in public parks, 294 species in roadside green spaces, 226 species in green spaces attached to residential areas, and 288 species in green buffers.

In terms of vertebrate diversity, the complex physical and geographical conditions and diverse vegetation types in mountainous areas of Beijing provide habitats for the survival of various wildlife (Cui et al. 2008). There are 8 species of amphibians in 1 order and 5 families in Beijing, 16 species of reptiles in 3 orders and 6 families, 61 species of wild mammals in 19 families, 84 species of wild fish in 9 orders and 15 families, and 156 species of wild birds in 16 orders and 48 families. The abundant topographic changes, land-use types, and complex natural environment in Beijing have formed superior natural ecological conditions, providing a suitable climate and breeding environment for birds as well as providing green space for resting and supplementing energy for migrating birds. Beijing's geographical location determines that the number of winter and resident bird species is less than that of summer and migrant birds. In addition, urbanization makes the habitat more complex than natural areas, which affects the distribution of birds and leads to the diversity of bird distribution in Beijing (Wu 2010).

Beijing formulated *Measures for the Implementation of the Wildlife Protection Law of the People's Republic of China* in 1989 and *Measures for the Administration of the Issuance of Wildlife Domestication and Reproduction Licenses* in 1991. In addition to legal management, investigations of plant diversity have been conducted since the 1960s, which compensates for the lack of information on plant diversity and its change along with environmental changes (Wu 2010). However, investigations of overall vertebrate diversity in Beijing are still lacking, despite some studies and surveys conducted in parts of Beijing, especially in mountainous and riparian areas (Bao et al. 2005, 2007).

2.3.2 Threats to biodiversity

Habitat destruction and fragmentation are the main threats to biodiversity (Tannier et al. 2012). With the expansion of the city, land cover and the distribution of green space in central Beijing have changed significantly. Consequently, this triggered the fragmentation of green spaces: the spread of urban space to surrounding natural mountains and farmland areas, the expansion of villages and towns in urban shallow mountains and mountain areas, the replacement of a large number of biological habitats by artificial surfaces such as buildings and roads, and the shrinking and fragmentation of habitat areas. A series of chain reactions have led to the gradual simplification of the original green space pattern, significantly affecting the habitat environment and migration routes of a considerable number of species, thus affecting the level of biodiversity in the central area and in the whole city.

Through afforestation of barren hills and conversion of farmland to forestry, the forest area of Beijing has increased. However, there are still some problems, such as low overall quality of forest resources, simple stand structure, low species diversity, and low soil and water conservation capacity (Wang et al. 2018). There are many soil erosion areas in Beijing, especially in the periphery of the city. In Mentougou District, for example, coal, lime, and sand mining became the leading industry in order to meet the development needs of the capital's modernization. However, many bare hills and abandoned mines were formed by energy exploitation, causing significant damage to the ecological environment (Ren et al. 2012).

Wastewater, exhaust gas, and waste have become significant pollution sources in Beijing's urban environment (Liu et al. 2002). Pollutants directly affect the survival and reproduction of various organisms. Moreover, pollutants indirectly pose risks to organisms by causing soil acidification and eutrophication. In Beijing, the growth rate of many tree species has slowed down because of the aggravation of pollution and the decline of groundwater levels (Wang 2006). Even native tree species, such as *Populus nigra*, which originally grew well, has been gradually eliminated from the main tree species of urban landscape greening (Zheng and Zhang 2011).

To some extent, exotic species can enrich the composition of local ecosystems, such as economic plants producing economic benefits. However, because of the lack of natural enemies and ecological constraints, some species will grow rapidly and spread to disastrous levels, endangering local species. In Beijing, there are 96 invasive alien plant species, most of which were introduced with landscape gardening projects (Wu 2010).

2.4 Challenges of urban green space planning in Beijing

Urban green space planning is regarded as an important section of urban planning as it can can profoundly affect how and where cities develop (Marzluff et al. 2012). Some cases in Europe and America are well-known, such as the green belt in the Greater London Plan, the Regional Parks of Berlin-Brandenburg, the Green Space Network in the Metropolitan Region planning of Madrid, and the Regional Green Plan of Paris in 1995 (Xu et al. 2011). Challenges of urban ecological conservation brought by urbanization are particularly serious in Beijing, which has been going through rapid development for decades, as mentioned above. Ecological conservation has been given increasing attention in China. For instance, in March 2018, the 13th National People's Congress passed an amendment to the Constitution of the People's Republic of China writing Ecological Civilization (a guild of ecological conservation) into the Constitution (China Eco Development Association: http://www.eco.gov.cn/policy/judicial_interpretation/1598.html).

However, especially in the capital city of Beijing, challenges and problems remain in how to embody ecological conservation in urban green space planning. The first challenge is how to input knowledge of ecology and biology into the current urban green space planning of Beijing. Although a lot of work has been done by ecologists and biologists in the conservation of urban ecosystems, there has been limited application of this in urban green space planning (Xu et al. 2011). For example, the evaluation of urban green spaces from the perspective of biodiversity conservation remains focused on the calculation of landscape patterns rather than being applied in urban ecological planning. The second challenge is that most decisions regarding residential and commercial development are based on the local government (Haaland and van Den Bosch 2015). These decisions are diffused in time and space and limit the impact of plans regarding ecological conservation at a regional scale. Third, although strategies such as setting multi-hierarchy protection areas were proposed in both the master plan and urban green space planning, they are regarded as conceptual guidance rather than a legal plan. Both academic knowledge and legal support are needed to create more comprehensive and effective urban green spaces in Beijing. In this case, this study aims to optimize the urban green space planning of Beijing by proposing a framework for evaluating urban green space from the perspective of species.

Chapter 3 Evaluation of urban green space based on landscape connectivity

3.1 Introduction

3.1.1 Social background

Green space broadly refers to a natural or semi-natural open space with considerable vegetation coverage (Kabisch and Haase, 2013), including woodland (forest), grassland, farmland, etc. (Xu et al. 2016). In the urban context, green space is inevitably turned to the built-up area for development, especially in rapidly developing cities. Due to the public ownership of land (a land law) in China, the land belongs to the central and local governments, land users only have the usage right of the land, instead of the land itself or any resources in or below the land. Master plan and green space planning dominated by the local government are regarded as a guide of green space management. In this context, planning is the most direct way to control urban expansion and mitigate landscape fragmentation.

However, there is a lack of theoretical research in the current green space planning, especially for its ecological function. The current green space planning includes the location, shape and function of each green space (Jim and Chen 2003). The area, number and accessibility are paid more attention, rather than its role in biodiversity conservation. Although it is important for urban green space to satisfy the human demand of recreation, the potential ecological function of it should be involved in urban green space planning, too. In the context of the loss of biodiversity with urbanization, how to mitigate the damage to the natural environment through a more efficient green space planning?

3.1.2 Academic background

Due to the rich biological resources in urban development areas, the contradiction between the green space conservation and urban construction has become one of the most important problems in densely populated areas (Miller et al. 1998). Landscape fragmentation, one of the consequences of urban expansion, is defined as habitat loss through the spread of artificial surfaces and the split of remaining habitat areas (Tannier et al. 2012). Landscape fragmentation will limit the movement of organisms between habitats, which can be a huge threat to biodiversity; especially in small isolated patches, the gene exchange will be greatly reduced (Harris, 1984; Cook, 2002; van Langevelde, 2015).

Landscape connectivity, the opposite of landscape fragmentation, "describes the facilitating or impeding effect of the landscape on the dispersal of species among habitats" (Taylor et al. 1993; Forman and Collinge 1997). It is used to evaluate ecological service function of a certain landscape by quantifying landscape pattern from a macro point of view. In recent decades, an interdisciplinary field called landscape ecology, has proposed new methods to understand how landscape pattern influences ecological processes (Turner,1989; Forman, 1995) For instance, landscape pattern may affect biodiversity (Pătru-Stupariu et al. 2017; Wilson et al. 2017) and warmer microclimate-heat island effect (Buyantuyev and Wu, 2010; Connors et al. 2013; Guo et al. 2015). In recent years, increasing numbers of ecologists, biologists and other scientists, planners and engineers have taken part in the interdisciplinary research about the landscape ecology and urban ecosystems (Grimm et al. 2008).

Landscape metrics and models focused on simulating ecological process become trends in landscape ecology (Taylor, 2006; Lookingbill et al. 2010; Grunwald, 2016). In terms of the dispersal probability among habitats, the connectivity between habitats can be characterized by a probability model, for example, the inter-patch Euclidean distance (Urban and Keitt, 2001; Adriaensen et al. 2003). Numerous landscape metrics and models were therefore developed to quantify landscape configuration and further evaluate the ecosystem services or forest management. In the meanwhile, an increasing number of cases applied landscape connectivity metrics or models in green space planning (Feranec et al. 2010; Wickham et al. 2010; Saura et al. 2011; Cushman et al. 2013).

According to the existing research about species response to landscape connectivity, it is generally accepted that landscape connectivity plays an important role in biodiversity, though it is case by case among all species. However, the arguments between biology and biogeography still exists (Baguette et al. 2013). There is a growing skepticism that it is unreliable to identify ecological corridors without including details such as animal behavior and intraspecific differences (Gurrutxaga et al. 2011; Maiorano et al. 2017). For example, species with lower dispersal ability tends to be less sensitive to landscape connectivity; and the barrier effect for dispersal of a certain land cover in urbanized areas is also species-specific (Minor and Lookingbill, 2010). In this study, seeking to make a more comprehensive assessment of landscape connectivity, the core habitats and corridors will be identified according to the habitat type and dispersal distance of the focal species. We will address the following methodological problems:

a) How to recognize habitat and corridor in existing research from a perspective of species? In this study, landscape is no longer a homogeneous area. Instead, it is defined according to habitat types of species. Besides, the distance threshold in the measurement of landsape connectivity depends on the dispersal distance of species.

b) How can we get a multi-species result? We provided a paradigm for ecological network identification of multi-species, such as birds, mammals, plants, a community or all species. Here, two focal species were selected to model the biodiversity of the whole. The final ecological network can be obtained by overlaying the scenarios of each species involved.

3.2 Study area and data

3.2.1 Study area: Haidian District

Haidian District (39°53′-40°09′N) is located in the northwest of Beijing (**Figure 3-1**). There are ten rivers (the total length of 119.8 km) running across the whole area, the water area accounts for 41.28% of the entire water area of Beijing, ranked first among 16 districts. Haidian District is surrounded by mountains and forests in the west. It is roughly separated into two parts by mountains: the impact built-up area in the south and the suburban area in the north.

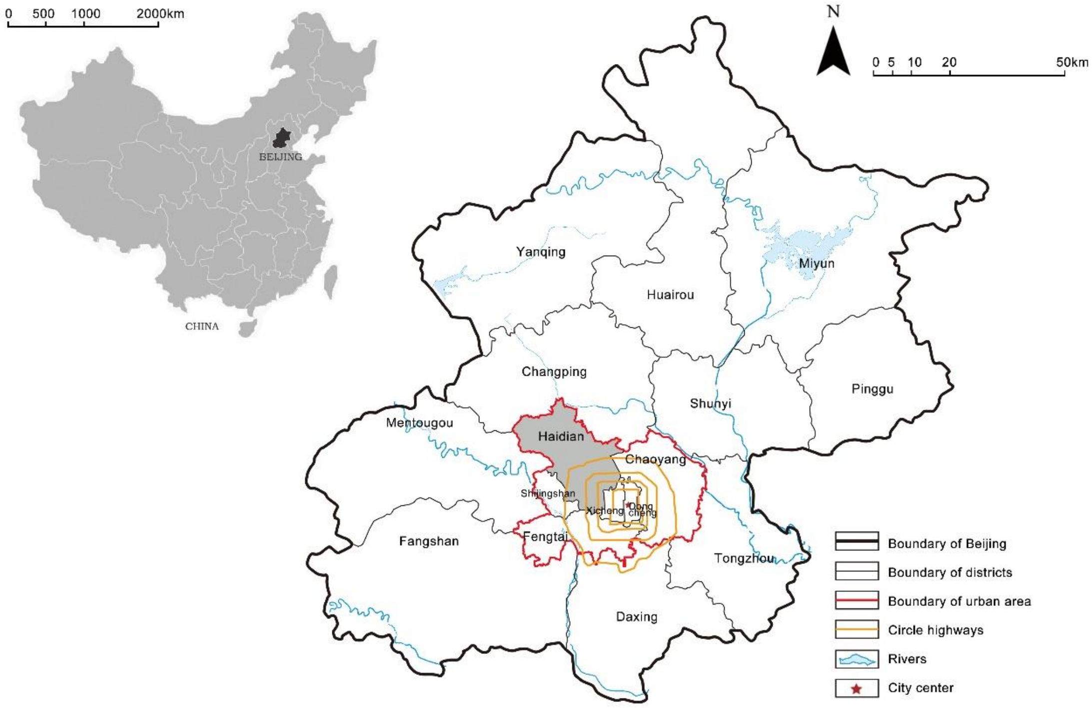


Figure 3-1 Location of the Haidian District

In this study, landscape connectivity analysis is based on focal species, which is supposed to have similar requirements of habitats with many other species, as described in *Section 1.4.2* (Marcot et al. 2007). The selection of focal species is adapted from a similar study in Beijing (Hu et al. 2010). In Beijing, large mammals are rare and mostly inhabited in mountain areas with high altitude, also, small mammals such as chipmunk are not sensitive to habitat changes resulting from urban expansion. In this context, a middle size mammal called raccoon dog (*Nyctereutes procyonoides*) is chosen as focal species, which is also belong to the first-grade protection animal of Beijing. We also select ring-necked pheasant (*Phasianus colchicus*) as one of the focal species. Dispersal distance of species is a critical process determining distance threshold. The maximal dispersal distance of raccoon dog is 2 km according to (Tokue et al. 2012); Leif (2005) radio-marked 95 male pheasants over five years (1997-2001) and get an average dispersal of 3.2± 0.3 km. We assign the dispersal distance of ring-necked pheasant as 3.2 km based on Leif's research. The focal species selected have common characteristics: 1) they rely on green space; 2) they are threatened by urban expansion; 3) their dispersal ability is ordinary.

3.2.2 Data source

We use the high-resolution remote sensing image (RS-image) to extract land cover information. Gaofen-2 (GF-2) satellite, successfully launched on August 19th, 2014. We used a GF-2 image (resolution=0.8m) of Haidian District, captured on 12 September 2015, acquired with a solar azimuth angle of 37° and an axial inclination of 94°. There are four multispectral bands (red, green, blue and near infrared) with resolution 3.2m and a panchromatic band with resolution 0.8m. They are fused into a 4-band pan-sharpened multispectral image.

3.3 Method

3.3.1 Data processing and landscape fragmentation analysis

Image processing was performed using ENVI and eCognition. Before classification, images must be segmented. The scale parameter refers to the threshold of the heterogeneity variation allowed in the segmentation process (Ming et al. 2015). Scale parameter will affect the accuracy and efficiency in the extraction process. Multi-scale segmentation was used to fix this problem. It is the foundation procedure of object-based image analysis (OBIA) to convert discrete pixels of RS-images into homogeneous image object (Kettig and Landgrebe, 1976; Vieira et al. 2012). Depending on the required land-cover categories (green space, agriculture land, built-up area, transportation area, and water), the segmentation scale parameter and the hierarchical relationship were identified according to their characteristics after several attempts to obtain a satisfactory result.

Difficulties in pixel-based classification caused by increasing satellite resolution led to the development of OBIA (Blaschke, 2010). By identifying spectral and spatial information (NDVI, geometry, brightness, texture, neighborhood attributes), adjacent pixels are grouped into multi-pixel objects (Aplin et al. 1999; Congalton and Green, 2008). We adopted the K-nearest neighbor method and obtained the land-cover categories by creating the following spectral characteristics: normalized difference vegetation index, standard deviation, maximum difference, brightness, length/width, roundness, and aspect ratio.

Landscape metrics, for example, L-Z complexity method (Li et al. 2009) and mean patch shape fragmentation index (Jiang et al. 2014) has been developed to quantify landscape fragmentation. Landscape fragmentation process can be classified into perforation, subdivision, shrinkage, and attribution, which can also be measured (Li and Yang, 2015). According to the definition of landscape fragmentation, fragmentation will bring two results: one is the decrease in patch area, and the other is the increase in patch number. In other words, the mean patch area will decrease. Therefore, we use the mean patch area to quantify the fragmentation. The RS-image is clipped into grids (size = 1 km × 1 km) using *Fishnet* Tool in ArcGIS. The area and number of patches in each grid can be summarized, then the mean patch area can be calculated to indicate its landscape fragmentation.

3.3.2 Habitats identification

Morphological Spatial Pattern Analysis (MSPA) and landscape connectivity are combined into a two-step methodology to identify habitats. First, we reclassified the landscape into seven categories based on morphological characteristics in GuidosToolbox (http://forest.jrc.ec.europa.eu/download/software/guidos/): core, edge, perforation, bridge, loop, branch, and islet (Soille and Vogt, 2009). We extracted core area for step2, which is regarded as the most important area for landscape connectivity. Unlike traditional methods, which focus on the area or importance of a single patch without taking the integrated landscape connectivity into account, this method provides the four- or eight-neighbor rule since the connectivity analysis is conducted on a raster grid. It allows automatic classification based on geometric concepts at a pixel level.

According to the habitat type of raccoon dog and ring-necked pheasant, in this case, we overlay the woodland as potential habitat for raccoon dog (waterbody is also involved because some raccoon dogs prefer riparian areas), meanwhile the woodland and agriculture land as the ring-necked pheasant's potential habitat for the following analysis. The potential habitats were identified referred to Gao et al. (1987), MacKinnon et al. (2000), Smith and Xie (2009). After reclassifying the land cover into the foreground (potential habitat) and background (all other land cover types), we obtained a potential habitat and non-potential habitat binary map. Due to the upper limit of this GuidosToolbox of 10000 × 10000 pixels, the pixel size of 4 m × 4 m is the upper limitation. We set the edge width as 5 pixels and extracted "core" area under the eight-neighbor rule to identify ecological habitats in the following analyses.

Probability of Connectivity (PC) can not only quantitatively describe the landscape connectivity but can also identify patches with important connectivity. PC is calculated by the following formula (0＜PC＜1):

$$\mathrm{PC} = \frac{\sum_{i=1}^{n}\sum_{j=1, i\neq j}^{n} P_{ij}^{*} a_i a_j}{AL^2}$$

where a_i and a_j refer to areas of habitats i and j; the connection strength between any pair of patches is marked by p_{ij}, which describes the ease of dispersal between patch i and j; AL is the area of Haidian District, including all kinds of land cover.

The delta of PC (dPC) can calculate the contribution of each patch to the overall connectivity of the ecological network. It can identify patches that crucial to PC. The calculation of dPC is according to the following formula:

$$\mathrm{dPC}\ (\%) = \frac{\mathrm{PC} - \mathrm{PC}'}{\mathrm{PC}} \times 100\%$$

where PC is the overall landscape connectivity, and PC' is the overall landscape connectivity after removing a patch from the original landscape. The change of the overall landscape connectivity is regarded as the importance value of the removed patch.

Further details of PC and dPC can be found in previous studies (Saura and Pascual-Hortal, 2007). We adopted Conefor Sensinode 2.6 software (http://www.conefor.org/) to calculate the dPC values of each green space. As the dispersal ability of different species varies, we assign the dispersal distance as 2.0 km and 3.2 km respectively for raccoon dog and ring-necked pheasant. The probability of dispersal was set as 0.5 (default). The calculation process refer to Saura and Pascual-Hortal (2007), Saura and Torne (2009), Saura et al. (2011). Finally, the top fifteen patches with the highest dPC values were chosen as habitats.

3.3.3 Least-cost path

The least-cost path is often used to optimize a grid module (Kong et al. 2010), making it possible to calculate the minimum cumulative link (corridors) between the target patch and the nearest source patch (habitat). The resistance value of a grid describes its facilitating or impeding influences on dispersal of species. The resistance value is attached to each land cover unit to calculate the connectivity between two habitats (Bunn et al. 2000). We calculate the path of the least resistance for the organism to migrate along and obtained the potential corridors between source patches using the *Cost Path* Tool in ArcGIS. The different resistance values of each land cover class was imported to generate a resistance surface. Then the tool calculated paths between each two patches with the minimum sum of resistance value. So the assignment of resistance value of each land cover can directly affect the result of identification of corridors. **Table 3-1** indicates the assigned resistance values with reference to (Kong et al. 2010).

Table 3-1 Landscape resistance value

Land cover class		Resistance value	
		Ring-necked pheasant (*Phasianus colchicus*)	Raccoon dog (*Nyctereutes procyonoides*)
Core	Habitat	1	1
	Important core	5	5
	Ordinary core	10	10
Woodland		20	20
Agriculture land		20	50
Waterbody		100	20
Road		500	500
Built-up area		1000	1000

Table 3-2 Land cover of the Haidian District

Class	Area (ha)	Proportion of total
Built-up area	19543	41.0%
Road	2257	4.7%
Woodland	16258	34.1%
Agriculture land	8747	18.4%
Waterbody	858	1.8%
Total	47663	100%

3.4 Results

3.4.1 Land cover and landscape fragmentation

We obtained a land-cover map using OBIA, showing in **Figure 3-2** (**a**). In the suburban area, the land cover map was a huge mosaic mixed with woodland, agriculture land and built-up areas. Here, disordered urbanization has resulted in severe landscape fragmentation. In contrast, the urban area is highly constructed; green space was mainly composed of parks and roadside green space. In both suburban and urban areas, roads and highways fragmented the green space. **Table 3-2** shows the land cover classification result: woodland occupies 34.1% of the total area and the proportion of the built-up area is 41%. We divided the land-use categories result using the fishnet tool in GIS and identified the center point of each grid (n = 429) as a sample of visual interpretation. We tested the classification accuracy through a confusion matrix. The overall accuracy was 85.5%, and the kappa coefficient was 79.23.

We roughly simulate the landscape fragmentation using mean patch area, landscape only includes woodland as a typical example. **Figure 3-2 (b)** shows that fragmentation is relatively severe in both urban area and suburb area. The legend on the left side of **Figure 3-2 (b)** shows the different levels of fragmentation, and the accounts for the overall proportion of each level. The value of 50-100 is 11% of the whole, on the contrary, the value of 0-6 is 65% of the whole, which means 65% of the study area has a serious landscape fragmentation.

3.4.2 The result of habitats and corridors

We obtained the MSPA-based landscape classification map (**Figure 3-3**). The amount of core patches is favorable, but they are relatively unevenly distributed. For ring-necked pheasant in **Figure 3-3 (a)**, core patches are distributed in the mountain area and suburban area. For raccoon dog in **Figure 3-3 (b)**, core patches that contribute to the overall connectivity are mostly concentrated in the mountainous region and several green spaces around.

The dPC value of each core patch with different dispersal distance considered (d = 2 km, 3.2 km) was calculated (**Table 3-3**). Then we obtained the importance rank map of the top fifteen cores showed in Figure 3-4. The size of the green space is often regarded as the most essential characteristic in green space planning. However, the size of a patch is not proportional to its role in maintaining PC. In **Figure 3-4 (a)**, although relatively small, Patch 3 and Patch 4 are more important than Patch 13 for landscape connectivity; also, in **Figure 3-4 (b)**, compared with Patch 9 and Patch 14, Patch 4 and Patch 5 are much smaller in size, but have higher dPC value. Besides, the geometry location of a patch is also unrelated to its dPC value.

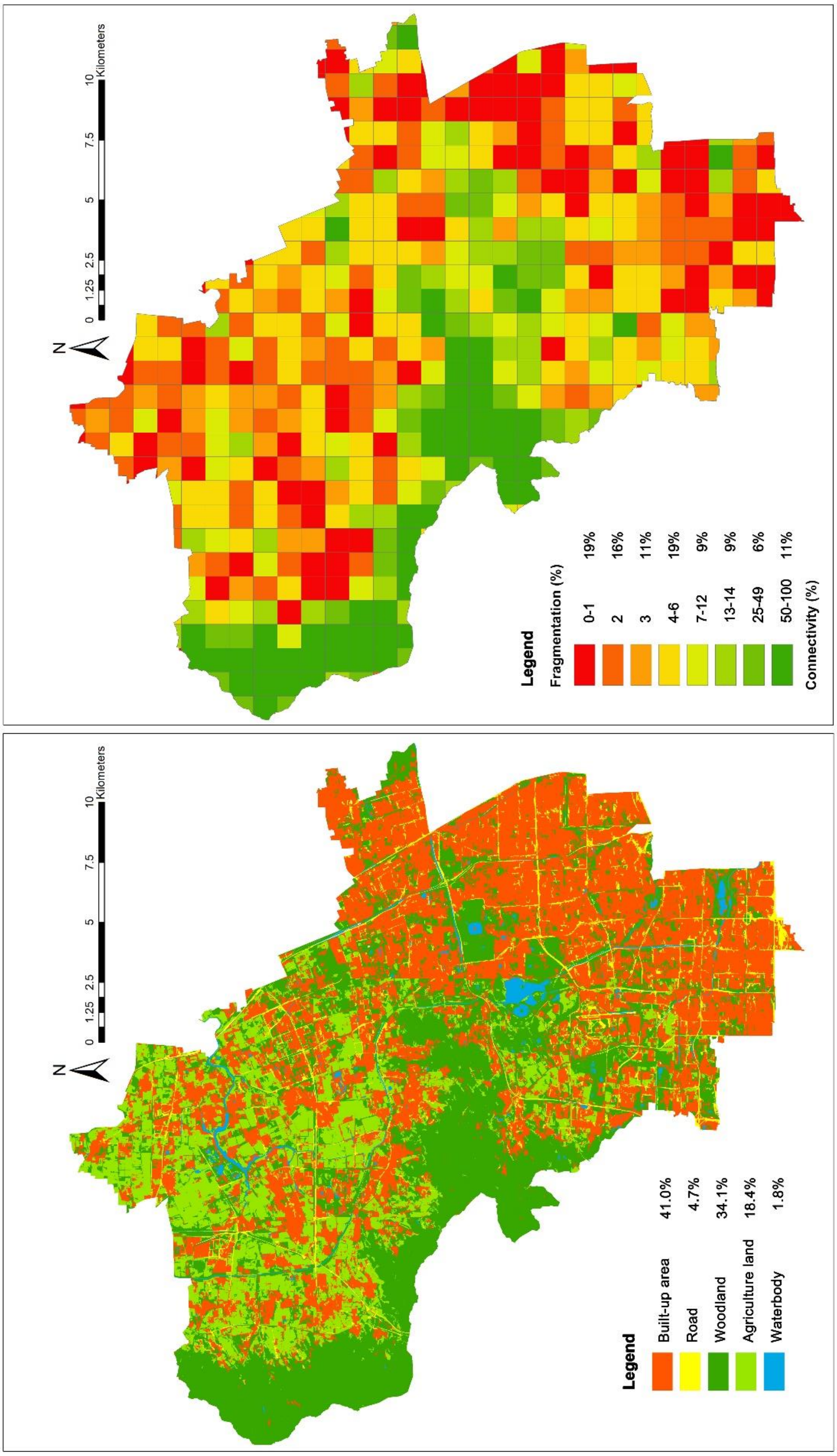


Figure 3-2 (a) Land cover classification of Haidian in 2015 **(b)** landscape fragmentation analysis of Haidian in 2015.

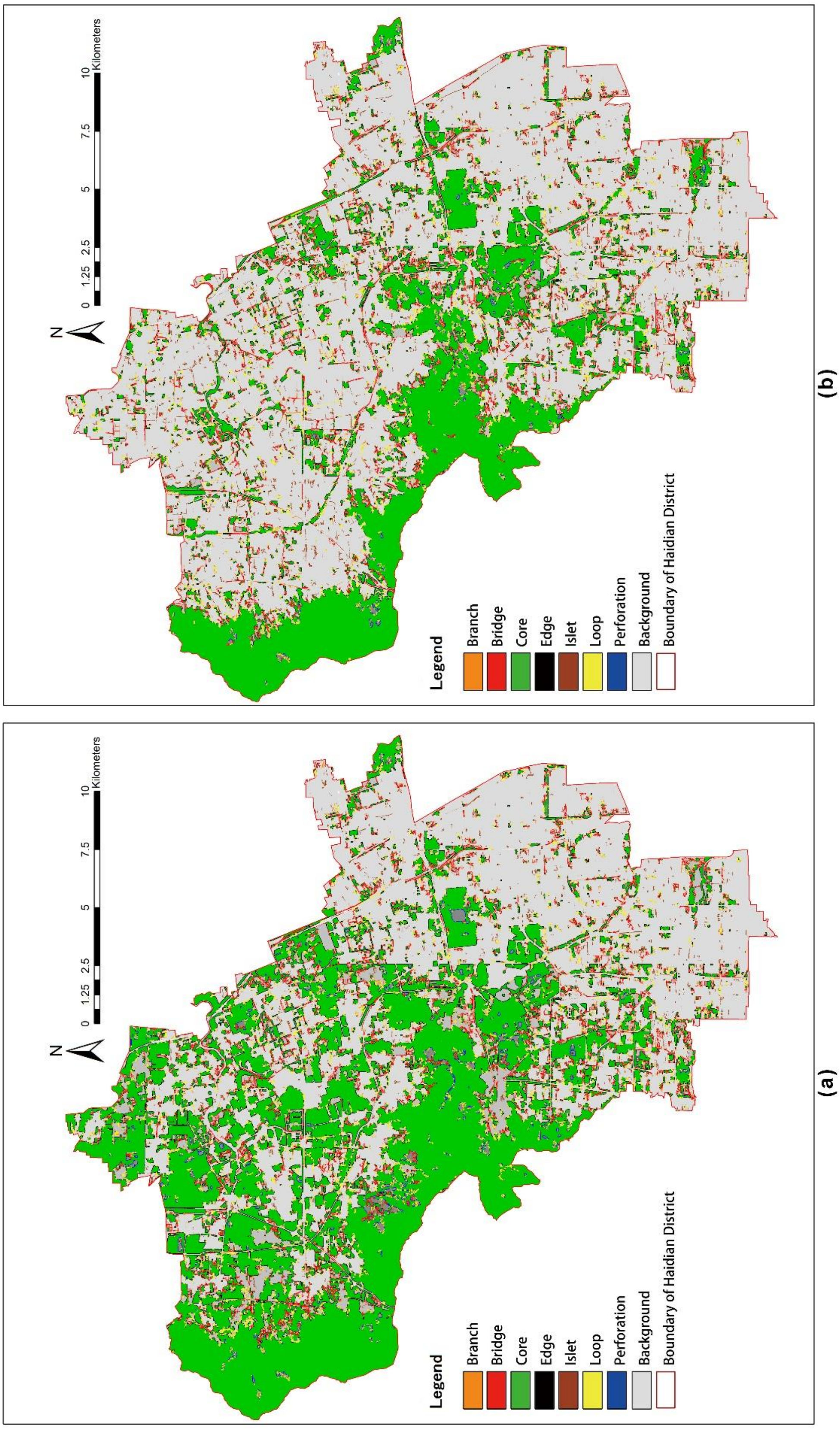


Figure 3-3 MSPA (morphological spatial pattern analysis) classification of Haidian according to different habitat type of focal species. **(a)** the ring-necked pheasant **(b)** the raccoon dog.

Table 3-3 Top15 Important core patches: Patch rank/No., dPC value, area

	Ring-necked pheasant *Phasianus colchicus*		Raccoon dog *Nyctereutes procyonoides*	
Patch Rank/No.[2]	dPC[1]	Area(ha)	dPC	Area(ha)
1	30.59	4153.8	22.11	2350.0
2	12.26	518.9	9.56	503.8
3	7.51	144.7	9.54	480.8
4	6.76	62.9	3.88	57.2
5	4.76	47.7	2.42	7.8
6	4.71	143.7	2.14	5.2
7	4.24	9.2	2.08	19.0
8	3.25	96.2	1.94	18.0
9	3.13	454.0	1.93	2188.0
10	3.10	260.0	1.87	6.0
11	2.84	207.5	1.73	17.3
12	2.62	3.6	1.73	43.1
13	2.34	86.1	1.72	11.3
14	2.20	1.8	1.65	285.7
15	2.00	8.0	1.51	1.1

[1]Note: dPC is an indicator of importance value for each patch on maintaining the overall landscape connectivity. [2]Note: The importance value is species specific. we assign the No. of a patch according with its dPC value. i.e., Patch 1 is the most important patch.

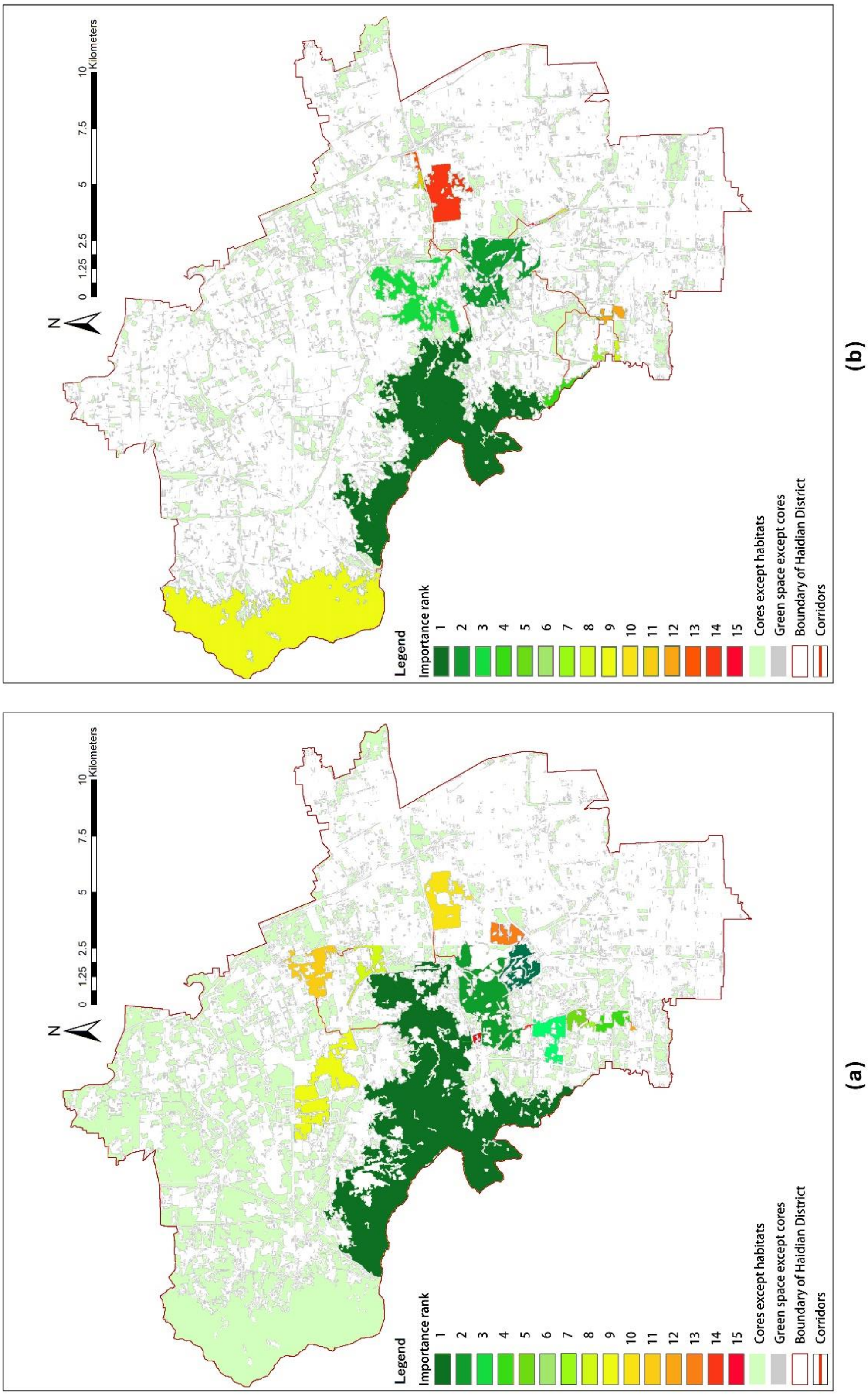


Figure 3-4 Importance rank of habitats and potential corridors. (**a**) the ring-necked pheasant (*Phasianus colchicus*) (**b**) the raccoon dog (*Nyctereutes procyonoides*).

Mountain region, in terms of its high importance value and large area, it has a dominant advantage of other green spaces in maintaining the whole landscape connectivity. When species need to immigrate from a patch far away from the mountain area, it needs to overcome the resistance from unsuitable land covers. In this case, the importance of patches far away from the mountain area is less than patches near the mountain area. In **Figure 3-4 (a)**, since agricultural land is also included in habitats, landscape connectivity seems to be favorable in the northern part of Haidan, the suburb areas.

Figure 3-5 shows result of overlaying **Figure 3-4 (a)** and **(b)**. The green space refers to all core area, includes woodland, agriculture land and waterbody. We didn't assign a specific width to the corridors, instead, they can only represent the possible dispersal path of focal species. According to the resistance value, these corridors are mainly woodland or agriculture land, mixed with other land covers. The calculation of corridors is based on the current land cover and landscape pattern. Our analysis revealed: a) there is a lack of woodland for forest species in the central urban area and suburb area; b) green spaces in the suburban and urban area are poorly connected; c) habitats are distributed in the mountain area and its surroundings. Due to the high resistance value of built-up areas, maintaining the organism migration is supposed to be more difficult in suburban and urban areas.

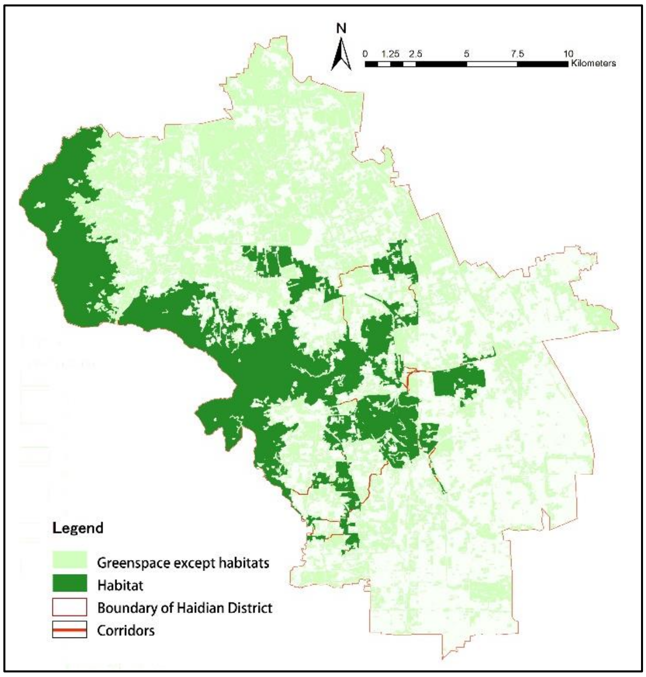


Figure 3-5 Ecological network of Haidian, obtained by overlaying habitats and corridors for ring-necked pheasant and raccoon dog.

3.5 Discussion

3.5.1 Involving biological characteristics in landscape connectivity

It is believed that the viability of a species depends on its ability of crossing unsuitable habitat and dispersal from one patch to another (Tannier and Girardet, 2012), which is also the start point of landscape ecology. This study failed to use empirical species census data, instead, it took biological characteristics of species into consideration by extracting habitat types and setting dispersal distances for each species respectively in the calculation of landscape connectivity. This study is an optimization of landscape connectivity measurement by involving biological characteristics of species.

Actually, it is still a tricky problem to test the correlation between landscape connectivity and biodiversity, considering that species survey usually takes a couple of years to get a considerable number of samples. According to a review (Humphrey et al. 2015), among the 94 studies about species and ecological network (1990-2013), only 34 studies discussed the connection or isolation between habitats as a factor affecting biodiversity (e.g. species richness, occurrence, abundance etc.). In addition, 32.4% of these 34 studies show no significant correlation, 67.4% of them refer to significant respond of species to landscape connectivity. But these studies are conducted under various conditions with various ways of measuring landscape connectivity and biodiversity. Seldom of them were conducted at a large scale, e.g. a region or a city. Therefore, the following chapters will use empirical species data to examine the relationship between landscape connectivity and biodiversity.

3.5.2 Contributions to the literature

The ecological network planning in the urbanized area should not only consider the connection between habitats but also in accordance with the characteristics of species. The existing studies on landscape connectivity of Beijing or other cities in China prefer to make an overall evaluation of landscape pattern using landscape metrics, i.e. FRAGSTATS (Zhang et al. 2004; Zhang and Wang, 2005). Besides, several studies using landscape connectivity to guide the wildlife protection area planning (Zhao et al. 2014; Liu et al. 2016). Compared to other research that focuses explicitly on certain species (Gurrutxaga et al. 2011; Matsuba et al. 2016; Maiorano et al. 2017), we propose a paradigm showed in **Figure 3-6** to evaluate and optimize the landscape pattern by involving biological characteristics of focal species. In this case, habitats and corridors are identified under the consideration of the habitat type and dispersal distance of focal species. Analogously, we can also identify a potential ecological network for any species or community.

We believe that landscape connectivity promotion, taking landscape pattern as the starting point, can be easily applied and promoted in studies or projects such as urban planning and biodiversity conservation at different spatiotemporal scales. The current green space planning in China does not make specific and reasonable ecological network planning according to different species. Urban planners can regard our green space importance evaluation as a reference. They can easily figure out the importance of a patch for different species. Especially when there is a demand for a certain amount of green space should be turned into the built-up area due to urbanization, our study can help urban planners make a wise choice to balance the urban construction and biodiversity conservation. Biologists can apply our method in species conservation, especially for endangered species, and collect information such as breeding and feeding habits, dispersal ability, habitat requirement, home range size and so on.

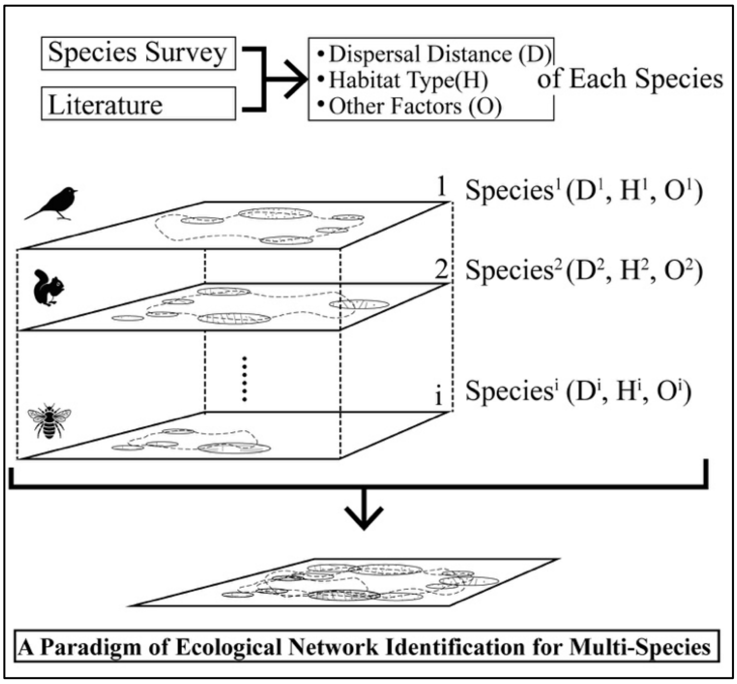


Figure 3-6 A paradigm of ecological network identification of multi-species. Dispersal distance, habitat type and other factors (altitude, slope, home range size, vegetation configuration, adapt to urban or not, residence type etc.) can be obtained from species survey and literature review. The simulated scenario of each species will be overlaid to get a final ecological network.

3.5.3 Limitations of results

Intensive studies have been conducted by scholars towards the identification of the preferable approach to quantifying and incorporating connectivity into green space planning (McGarigal et al. 2002; Marulli and Mallarach, 2005; Pascual-Hortal and Saura, 2007, 2008). Pascual-Hortal and Saura (2007) proposed that dPC performs best among other metrics they have put forward for identifying the importance value of a patch. We adopted the MSPA and dPC to identify key green space as habitats.

However, we acknowledge the limitation of this study in the consideration of resistance value and the dispersal distance. For resistance value, we extracted the woodland with no distinction between original forest and plantations. Although we identify the resistance value according to other Chinese scholar’s research (Kong et al. 2010), it is subjective to some extent. There are two ways to identify dispersal ability: 1) scenario analysis conducted to figure out its effect to landscape connectivity (Gurrutxaga et al. 2011; Freeman and Ray, 2001; Freeman and Bell, 2011; Devi et al. 2013); 2) surveys of specific species (Closset-Kopp et al. 2016; Moqanaki and Cushman, 2017). Although we assign the dispersal distance for focal species according to published research papers, the dispersal distance of most species is under discussion.

In addition, there are some inconsistencies on the habitat of raccoon dog (*Nyctereutes procyonoides)* between literatures from China and Japan. Saito and Koike (2013, 2015) surveyed the distribution of raccoon dog in the Tama Hills and on the Boso Peninsula of Tokyo metropolitan area of Japan. They found that raccoon dog had a bell-shaped distribution along the urban-rural-forest landscape gradient and dominated the intermediate rural landscape, including agricultural areas. However, agricultural areas were not included in the habitat of raccoon dog in this study, which were referred to two Chinese books: Gao et al. (1987), Smith and Xie (2009). At present, studies on the habitat selection of raccoon dog is limited in China, but future studies may prove the distribution of raccoon dog in agricultural areas in China. At that time, we should reconsider the selection of focal species and update the results.

Further, different land types may be used by a given species for different purposes or to different degrees (Mörtberg, 2001), the specific needs and behaviors of certain species should be considered in the future work. Land cover configuration, distance to water resources, distance to the residential area and vegetation configuration may be factors affecting habitat selection.

3.5.4 Summary of Chapter 3

In this Chapter, we analyzed the potential network configurations by combining the method of OBIA, MSPA, PC, and the least-cost path. Landscape metrics and models have proven their efficiency in wildlife conservation planning and green space management (Mörtberg, 2001;

Moqanaki and Cushman, 2017). We went deeper into the details of landscape models in habitat identification and distance threshold. Two focal species were adopted to get a species-specific result. the habitat type and dispersal distance of each species were obtained from the existing research. Results obtained from the landscape analysis show that: involving biological characteristics of focal species in the calculation of landscape connectivity provides new perspective of urban green space evaluation.

A multi-species paradigm is put forward to solve the deficiencies in the existing research, which can be adopted to conservation planning and green space management. Taking the conservation of endangered birds as an example, factors affecting the diversity of the endangered birds should be investigated through bird surveys and literature review. After figuring out species home range, dispersal distance, demand for habitats etc., a comprehensive result can be obtained by overlaying all scenarios for all species.

Chapter 4 Bird communities in urban riparian areas: response to the local- and landscape-scale environmental variables

4.1 Introduction

4.1.1 Bird: an indicator of biodiversity

Bird is often selected as an indicator of biodiversity because their autecology is well studied, and they are relatively easy to observe and identify (Liang et al. 2018). As highly mobile species, birds are relatively environment-sensitive, and they are particularly responsive to environmental changes at different scales (Cunningham et al. 2014; Burgess and Maron, 2016; Guttery et al. 2017). Studies using bird as an indicator of biodiversity have become an important part of urban ecology. Zhang and Huang (2018) reviewed the studi es on bird communities in urban areas of China from 1950 to 2015. It showed that the publications on urban ornithology has been increasing since 1980s, from an annual average of 0.7 articles in 1981-1990 to 11.6 articles in 2011-2015. The study areas cover 57 cities located in 29 provinces, including Beijing, Shanghai, Guangzhou with the most attention. Results show that environmental variables at different scales had a variety of effects on bird species; however, long-term study on urban avian is rare.

4.1.2 Urban green space and biodiversity

Urban green space not only provides recreational places for residents, but also makes great contribution to support the functions of ecosystems, e.g. mitigating heat island effects (Sun and Chen, 2017), isolating the noise (Margaritis and Kang, 2017), conserving the biodiversity (Garddard et al. 2010). Although the heterogeneity of urban green space is higher than that in natural environment, the complex landscape type in urban green spaces may even be the shelter of endangered species. Ye (2016) compared the bird diversity in six urban parks and urban areas outside the parks in Shanghai. Although the bird diversity in urban areas and parks is almost the same, the number of bird species in urban areas is significantly lower than that in parks.

Understanding the ecological mechanisms supporting biodiversity is essential for understanding ecosystem functioning (Humphrey et al. 2015). Urbanization, especially regarding megacities, has been regarded as a primary cause of habitat destruction and biodiversity loss (Ortega-Álvarez and MacGregor-Fors, 2009; Haedo et al. 2017). The influx of population and the deforestation of trees in the process of urban construction leads to the destruction of natural habitats. Meanwhile, land cover changes and landscape fragmentation indirectly affect biodiversity in these habitats. Fortunately, urban green space, including urban parks, croplands, and water bodies, provides important habitats for wildlife (Jokimäki et al. 2018). For example, Oliveira Hagen et al. (2017) compared the bird diversity of 25 urban areas and non-urban areas (covered by semi natural habitats and agricultural areas with simple vegetation structure in suburbs) over the world. They found that the bird diversity in the city was even higher than those in these semi natural habitats, which they believed was due to the high habitat diversity within the city.

4.1.3 Impacts of urbanization on bird communities

At local level, one of the effects of urbanization is vegetation configuration, which is often selected by the designer, especially in the newly built urban green space. For the sake of beauty, various vegetation will be used in urban green space; but for the sake of climate and economy, local plants are often widely used. The monopoly of local plants will homogenize the vegetation structure. In addition, compared with natural green space, birds in urban green space tend to suffer more disturbance from traffic noise and human activities. For example, omnivorous birds have more chance to find food due to the domestic garbage and human feeding, e.g. *Pica pica* and *Passer montanus*. Under the influence of urbanization, some sensitive birds are filtered, while urban adaptive birds are remained. Sandström et al. (2015) shows that from the center of the city to the surrounding areas, the species and individuals of woodpeckers, burrowing birds and forest birds are on the rise, while the urban adaptive species are on the contrary. In addition, some studies show that the urbanization environment is generally not suitable for the survival of local birds:

this means that with the change of urban environment, local birds gradually move out, while foreign birds gradually move in. In 1954-1978 and 2003-2013, bird data in Beijing showed that the proportion of exotic species increased from 22% to 46%. Therefore, urbanization will change bird species composition.

At the regional level, urbanization will change the land cover type and landscape pattern, which may increase the landscape heterogeneity. A growing number of studies have shown that the richness and diversity of bird communities in cities will significantly decrease with the increase of urbanization (Galbreath et al. 2014). The similarity between bird communities, called biological homogenization, will increase in urban areas. Just as mega cities in the world are becoming more and more similar, the similarity of urban environment will lead to the global biological homogeneity. When the original habitat diversity is replaced by similar buildings and roads, leading to similar microclimate, bird diversity will inevitably decrease. Sol et al. (2017) pointed out that the highly urbanized environment supported an average of 450 million years of evolution less than the natural environment after analyzing the bird functional diversity data from five continents.

4.1.4 Research gaps, aims and objectives

Rivers play a significant role in biogeochemical cycles and in the provision of water for domestic, agricultural, recreational, navigational, and industrial purposes (Ramkumar et al. 2015; Suri et al. 2017). Particularly, well-managed urban riparian areas are considered to be important habitats for birds to escape human disturbance when living in urban areas, and act as transient habitats for migratory birds (Pennington et al. 2008; Tiwary and Urfi, 2016; Threlfall et al. 2016). There is a growing number of studies regarding biodiversity in urban green spaces, mostly in urban parks (Huang et al. 2015; Yang et al. 2015; Sasaki et al. 2016; Estevo et al. 2017; Morelli et al. 2017, 2018; Steel et al. 2017; Canedoli et al. 2018). Only a few studies have looked into bird diversity in urban riparian areas, two of them examined how rivers and their catchments impact the functional composition of urban bird communities (Suri et al. 2017; Banville et al. 2017). Therefore, this study selected urban riparian area as a .case study clarify influential environmental variables affecting bird communities living there.

Although a number of studies have looked into the response of birds to environmental changes, most of them have focused on a particular issue in a particular habitat, or on the influence of a particular environmental variable (Barton et al. 2014; Mammides et al. 2015; Chambers et al. 2016; Farmann et al. 2018; Salgueiro et al. 2018). “Local” and “regional” are defined as simple hierarchical classifications of environmental variables (Galitsky and Lawler, 2015). Since the 1980s, many studies have been conducted on the species–habitat relationship at a local level, i.e., the responses of a single or a group of species to local habitat characteristics (e.g., number, height, and crown radius of trees) (Taylor et al. 2016; O'Neill et al. 1986), including human disturbance represented by the traffic volume (Rukke and Midtgaard, 1998; Dorresteijn et al. 2015; Reijnen et al. 1995). In the last decade, with the development of remote sensing technology, information on land features has become easily accessible, and studies have started to focus on the habitat–species relationship at a regional level, e.g., land cover and landscape connectivity (Humphrey et al. 2015; Reijnen et al. 1995; Forman et al. 2002). However, few studies have assessed both local- and regional level variables at the same time (Humphrey et al. 2015; Sahraoui et al. 2017; Xu et al. 2018).

In addition, biological characteristics of species should be included in analyses, because the influence of environmental variables on biodiversity in a certain area also depends on biological traits of the species inhabiting that area. Since specific traits of bird species, such as habitat type, may affect the response of which to environmental variables, a few studies used a method to avoid the bias: they clustered species into groups according to the biological trait, and then defined a virtual species for each group (Suri et al. 2017; Foltête et al. 2012). There are growing numbers of studies that simulate bird migration by setting a distance threshold, and then model biodiversity in each habitat patch (Forman and Collinge 1997; Mimet et al. 2016; Xu et al. 2018). It is often selected as an efficient metric for modeling ecological networks and maintaining biodiversity by using focal species, although usually empirical evidence is insufficient to verify the focal species (Lambeck 1997; Silvano et al. 2017). However, studies considering the dispersal ability of the whole bird communities are still lacking, because information about their dispersal ability based on empirical evidence is limited in existing research (Loman et al. 2018).

Comprehensive studies that include the biological characteristics of multiple species and environmental variables at both the local and landscape scales are lacking, which limits their utility for biodiversity conservation (Miguet et al. 2016). Given the limitations of existing species-habitat relationship studies, this study aimed to clarify and compare the response of bird communities to both local and regional level environmental variables. As shown in **Figure 4-1**, we explored environmental variables from four aspects: a) vegetation composition, b) human disturbance, c) land cover, and d) landscape connectivity. We sought to:

1) Clarify the avian biodiversity in urban riparian areas of the Tsing River.
2) Identify the influence of environmental variables on avian biodiversity.
3) Obtain a species-specific understanding of 2) by including biological characteristics.

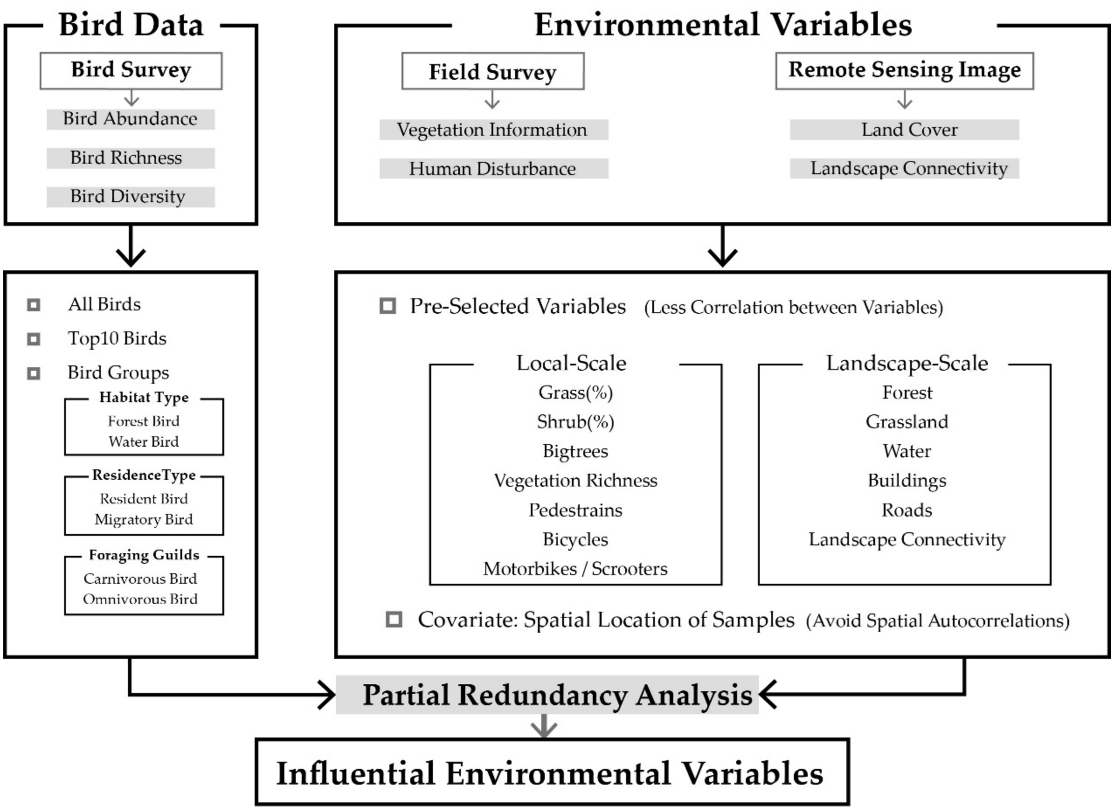


Figure 4-1 The research framework of Chapter 4.

4.2 Methods

4.2.1 Study area

The Tsing River is a small urban river in the northwest of central Beijing (39°38′–41°05′N), China (**Figure 4-2**). It is about 23.7 km long and 100 m wide. The catchment is 375 km2 and was modeled with Digital Elevation Model (DEM) data (resolution = 30 m) by hydrological simulation in ArcGIS. The water originates in the Western Mountains and Jade Spring Hills in the west. An important function of the Tsing River is its drainage of storm waters, but is also an important habitat for wildlife in its riparian area. The Tsing River flows along the boundary between urban and suburban areas, where the mixture of land cover has resulted from continuous urban expansion and construction over the past decade. In the upper reaches of the Tsing River, there are large green spaces and residential areas around the riparian green spaces. In the middle part, there are highly urbanized areas with residential and commercial constructions, and the riparian green spaces are maintained as urban parks to meet the recreational demand from residents. In the lower reaches, the waterfront remains relatively close to its natural condition with fewer artificial structures and lower levels of human activity and vegetation management (**Figure 4-3**).

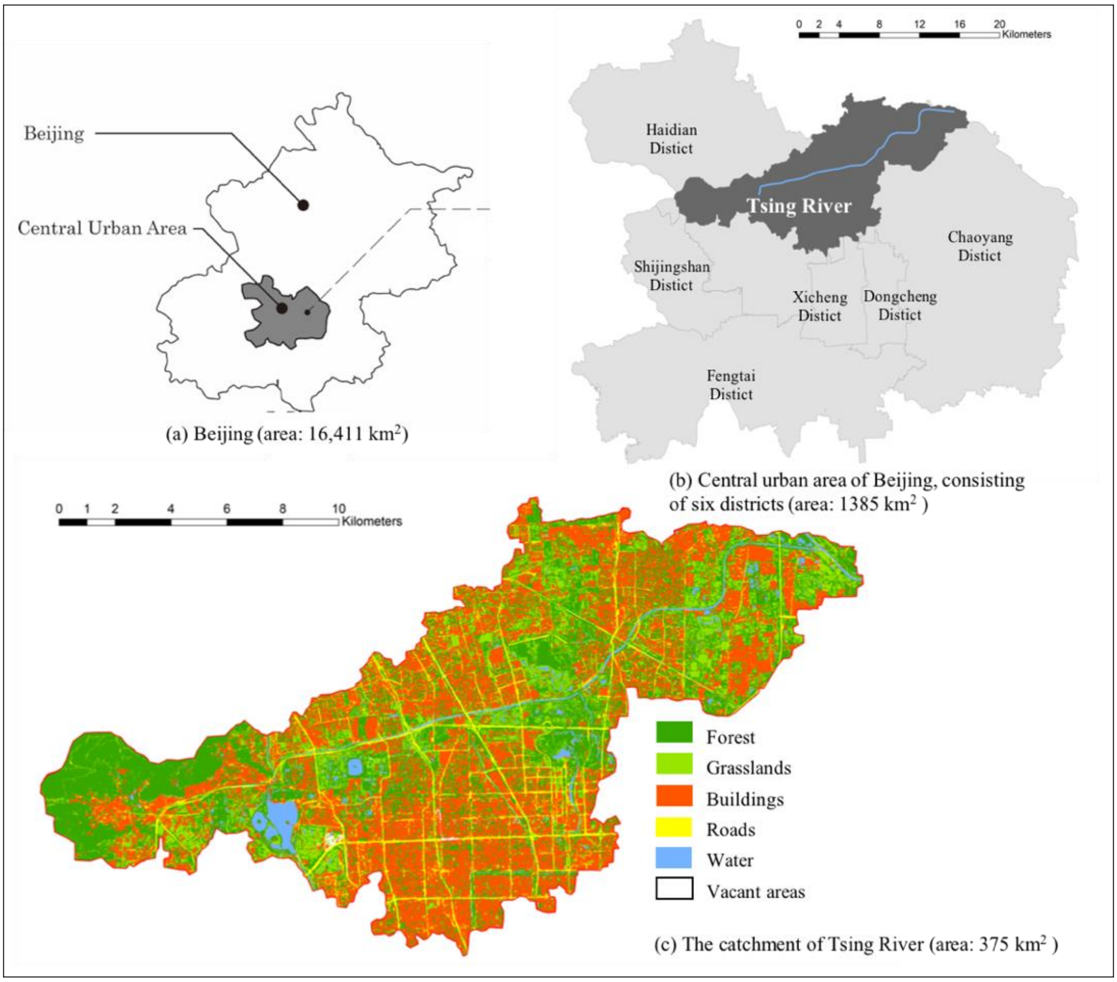


Figure 4-2 The location and the catchment of the Tsing River. Land covers were interpreted from Gaofen-2 remote sensing image.

Figure 4-3 Images of the Tsing River and waterfronts: (a) in the upper reaches, although there are walkways along the river, there is little human activity and some sections are inaccessible;

(b) in the middle reaches, recreational functions are taken into account in plant allocation and landscape design because of the large number of residential and commercial areas around it; (c) in the lower reaches, human activities and vegetation management are less frequent, and there are dense trees on both sides of natural riverbanks.

4.2.2 Bird surveys and biodiversity metrics

Bird surveys were conducted once a month from dawn–10:00 and from 16:00–dusk between May 2016 and Apr 2017. They were carried out by professional ornithologists along 1-km randomly allocated line transects (n = 18). Plots on Figure 4-4 were generated by a buffer of 500 m on each side of line transects, representing the boundary of collecting regional-level environmental variables. Line transects (1-18) were coded from upper reach (west) to lower reach (east). Observers walked along the line transect at a speed of 1 km/h while recording birds and identifying species based on a book: *A field guide to the birds of China* (MacKinnon et al. 2000). To reduce bias, birds flying over or beyond 100-m distance were not counted. We calculated three metrics of biodiversity for each transect: a) bird abundance (BA); b) bird richness (BR); c) bird diversity (BD).

We selected environmental variables according to existing research (Huang et al. 2015; Tannier et al. 2012; Miguet et al. 2016; Shih, 2018). At the local scale, we estimated 8 environmental variables: 1) ground-level: coverage of grass (%); 2) shrub-level: coverage of shrub (%); 3) tree-level: number of big trees with height > 10 m or crown > 6 m; 4) vegetation richness: the number of vegetation species; 5) human disturbance: pedestrians, bicycles, motorbikes/scooters, and cars. Local-scale variables were investigated from field surveys in 54 sample sites (size: 10 m × 10 m), conducted between Sept 19, and Sept 25, 2018. An example section of the Tsing River is shown in **Figure 4-4**. We selected 54 sample sites (3 sample sites on each line transect) to collect the information pertaining to vegetation structure. As to human disturbance, observers recorded the number of human disturbance items passing each sample site in 5 min during rush hour. To reduce bias, we calculated the average value of the three sites along each line transect.

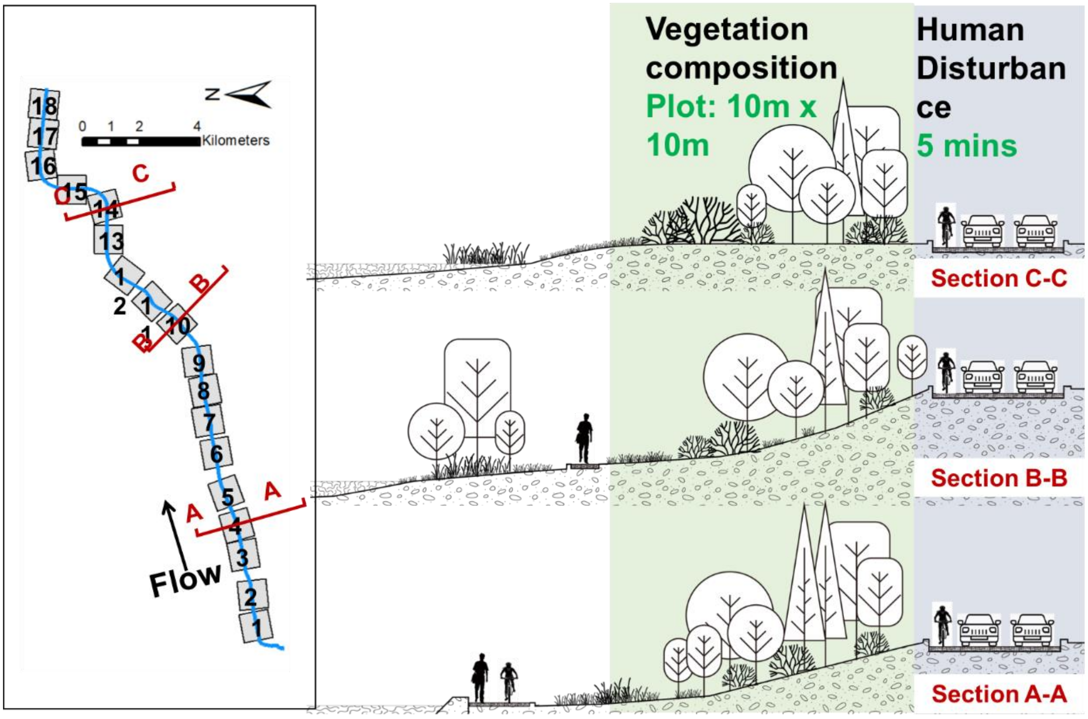


Figure 4-4 Illustration of the river section. Tsing River flows from west to east. Local environmental variables were investigated from field surveys in 54 sample sites (size: 10 m × 10 m). The illustration shows three typical sections with the location marked on the left. The section A-A, B-B and C-C are located in the upper reaches, middle reaches and lower reaches respectively.

At the landscape scale, we selected land cover and landscape connectivity as variables: forest (F), wooded areas (WA), grasslands (G), waterbodies (WB), water courses (WC), buildings (B), roads (R), and vacant areas (VA). Land cover classification was extracted from Gaofen-2 remote sensing imagery at a resolution of 0.8 m, which were captured on Sept 12, 2015 with a solar azimuth angle of 37° and an axial inclination of 94°. Since we can measure land cover and landscape connectivity at any spatial scale, it is necessary to determine the scale at which different variables more accurately reflect biodiversity. As Alexandrino at al. (2016) mentioned, variables are measured in different sized area, which may lead to a “scale of effect”. To test the “scale of effect,” all regional level variables were measured in two spatial areas: the observation area (with a buffer of 100 m) and the surrounding area (with a buffer of 500 m). As shown in **Figure 4-2**, we roughly interpreted the land cover of the river catchment using an object-based image analysis (Kettig and Landgrebe, 1976; Vieira et al. 2012). To determine the height of buildings, road hierarchy, and high-density planted woodland and low-density shrub areas, we reclassified the land covers, and verified the classification results by field surveys and Baidu Street View Map (https://map.baidu.com/) (**Figure 4-5**).

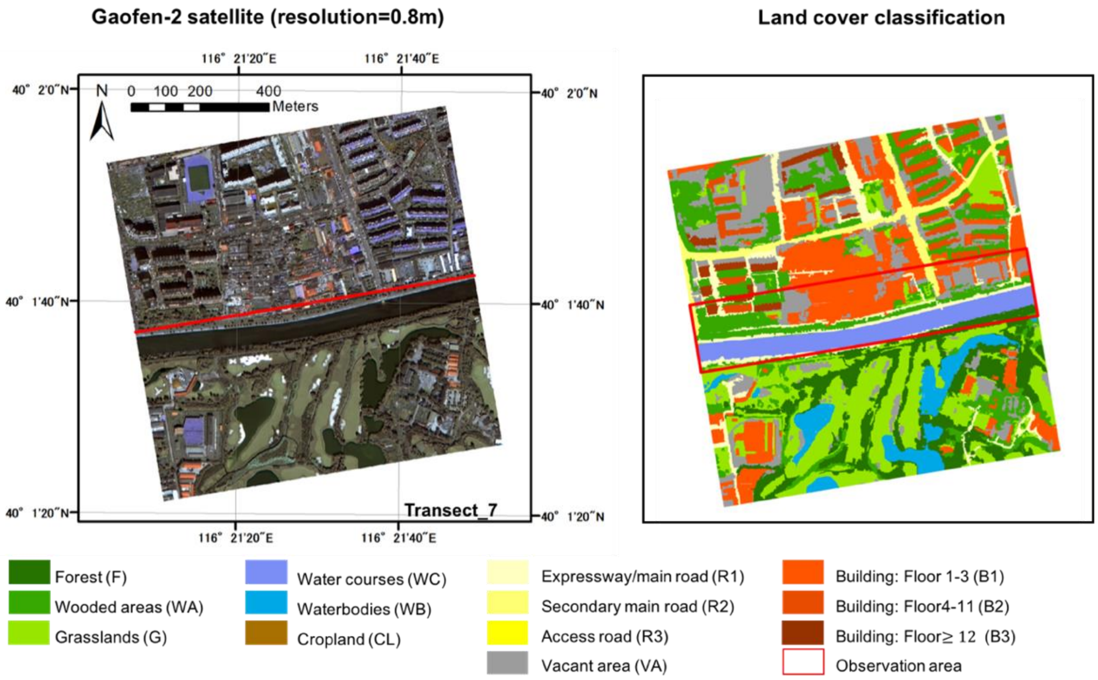


Figure 4-5 An example of land cover interpretation. Objective-based image analysis can classify forest, wooded areas, grassland, water, cropland, road, building, and vacant area. The reclassification of building and road were based on field surveys and Baidu Street View Map Baidu street map (https://map.baidu.com/). The complete land cover of 18 transects is showed in Figure A1.

Green spaces were further extracted from the land cover classification to calculate the delta of PC (dPC) of each patch using the Conefor Sensinode 2.6 software (http://www.conefor.org/). The linkage thresholds were defined by the dispersal distance of each species. To map the landscape connectivity of a patch to a line transect, we slightly modified the existing equation of dPC (Saura and Pascual-Hortal, 2007; Saura and Torne, 2009) and obtained the expression of landscape connectivity of a transect (dPC_t) as follows:

$$\mathrm{dPC_t} = \sum_{i=1}^{n} \frac{\mathrm{A_i}}{\mathrm{A_i}'} \mathrm{dPC_i}$$

where there are n patches overlapping with the line transect, the ith part belongs to a patch i, the area of the ith part is A_i, the area of patch i is A_i', and the landscape connectivity of patch i is dPC_i.

4.2.3 Preselection of variables and statistical analyses

Connections between various environmental variables are complex in the ecosystem. In the field survey, we tried to collect as much environmental variables as possible. However, those variables could be highly correlated with each other. In this case, variables with a correlation coefficient larger than 0.7 was deleted from the database according to the result of Pearson Correlation analysis (IBM SPSS Version 22). Using stepwise regression analysis, if the newly added variables can not improve the degree pof freedom and make other variables not significant, it can be eliminated.

Besides, we tested the hypothesis that birds may respond to the height of building and the hierarchy of roads. The result of correlation analysis failed to support our hypothesis. We thereby merged buildings with different height, and roads in different hierarchy into B and R respectively. In addition, we assigned the spatial scale of landscape variables, after examining the difference when they were measured in different spatial scale. Except forest land (F), the landscape-scale variables showed a better correlation with biodiversity indices when they were measured within a buffer of 500 m. Finally, we selected 7 local-scale variables: grass (%), shrub (%), big trees, vegetation richness, pedestrians, bicycles, motorbikes/ scooters; and 6 landscape-scale variables: forest land (F), grass land (G), water (W), buildings (B), roads (R), landscape connectivity. The raw data of all environmental variables and were log-transformed, and the violation of the assumption of normality of those variables was detected using the Kolmogorov-Smirnov (K—S) test to meet the requirements of the following analyses.

The redundancy analysis (RDA), one of the multivariate statistical methods, was applied to identify the most influential variables for avian biodiversity (CANACO 5.0) (Lepš and Šmilauer, 2003). The stepwise selection procedure of RDA can simplify the subset of explanatory variables (i.e. environmental variables) by a constrained ordination model. We identified the contribution of each variable to explaining the variance and tested the significance of the contribution using a partial Monte Carlo permutation test. False discovery rate estimates, one of the significance adjustment methods, was applied to avoid Type I errors. Since the flow of the Tsing River also reflects the trend of urbanization, the spatial autocorrelation between samples was excluded by adding a covariate named "location": the upper (transect 1-6), middle (transect 7-12) and lower (transect 13-18) reaches of the river were assigned values of 1, 2 and 3 respectively. We conducted partial RDA on all birds and the top 10 birds. To further discuss the response of birds with different biological characteristics, we grouped them based on the habitat type, residence type and foraging guilds: forest bird (inhabiting forest or wooded areas), water bird, resident bird, migratory bird, carnivorous bird, omnivorous bird. Then we repeated the RDA on the six groups respectively. **Table A3** contains all information on biological characteristics, collected from MacKinnon et al. (2000).

4.3 Results

4.3.1 Observed birds and the selected variables

The land cover of each transect is shown in **Table A1**. The vegetation composition and human disturbance investigated from field surveys are shown in **Table A2.** A total of 85 bird species and 15,632 individual birds were observed along Tsing River, as shown in the species list (**Table A3**). Among all species (n=85), forest birds and water birds occupied 31.8% and 18.9%; resident birds occupied 29.4%, the rest were regarded as migratory birds; carnivorous birds and omnivorous birds occupied 40% and 57.6%, respectively. Bird census data is shown in **Table A4**.

4.3.2 Overall responses of birds

Table 4-1 shows the result of RDA, with an acceptable significance calculated by the permutation test (p=0.022). The overall influential variables for all birds are grass (48.5%), pedestrians (11.7%), bicycles (10%), and F (5.5%). In comparison, the top 10 birds responded to more variables: F (22.7%), coverage of grass (18.2%), coverage of shrub (11.3%), bicycles (8%), vegetation richness (6.2%), and motorbikes (7.3%).

Figure 4-6 illustrates the correlation between all response variables and explanatory variables, from which we can get more details of a single biodiversity metric. **Figure 4-6 (a)**

shows that bird abundance and bird richness are more related to the coverage of grass (explained variance= 48.5%, p= 0.034), compared with bird diversity. **Figure 4-6 (b)** suggests that the coverage of grass has a significant correlation with the abundance of *Cyanopica cyana (CyanCyan*) and *Pycnonotus sinensis (PycnSine)*, although it is the second important variable for the top 10 birds (explained variance= 18.2%, p= 0.054); the most influential variable F (explained variance= 22.5%, p= 0.034) has a more significant correlation with *Anas platyrhynchos (AnasPlat)* than other species among the top 10 birds.

4.3.3 Responses of different groups of birds

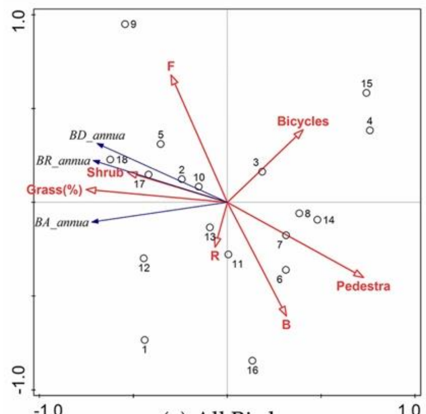
Similar as the RDA for all birds and top 10 birds, with regard to each bird group, **Table 4-1** shows influential environmental variables and the contribution of explained variance of each variables. The coverage of grass is the crucial variables for resident birds (45.4%), forest birds (49.1%), water birds (26.9%), and omnivorous birds (54.1%). Meanwhile, the number of pedestrians passing ranks top for migratory birds (30.7%) and carnivorous birds (39.7%). Among all variables, the coverage of grass and the disturbance of pedestrians can be the most influential variables affecting the avian biodiversity.

Specifically, for resident birds and migratory birds, although both of them negatively correlated with human disturbance, migratory birds (explained variance of pedestrians= 30.7%, p= 0.034) are more sensitive than resident birds (explained variance of bicycles= 13.9%, p= 0.038).

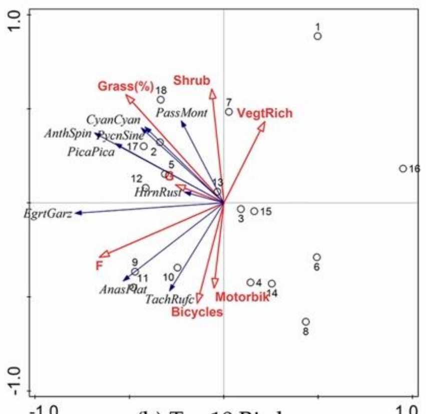
Table 4-1 Redundancy analysis to identify influential environmental variables.

	Name	Explains %	pseudo-F	P
All birds	**Grass (%)**	**48.5**	**14.1**	**0.034**
	Pedestrians	11.7	4.1	0.074
	Bicycles	10.0	6.2	0.098
	F	5.5	4.6	0.080
Top10 birds	**F**	**22.7**	**4.4**	**0.034**
	Grass (%)	18.2	4.3	0.054
	Shrub	**11.3**	**3.1**	**0.022**
	Bicycles	**8.0**	**2.4**	**0.038**
	Veg. Richness	6.2	2.0	0.066
	Motorbikes/scooters	7.3	2.8	0.052
Resident birds	**Grass (%)**	**45.4**	**12.5**	**0.034**
	Bicycles	**13.9**	**7.9**	**0.038**
	B	**11.2**	**3.6**	**0.020**
	W	6.6	5.0	0.052
Migratory birds	**Pedestrians**	**30.7**	**6.7**	**0.034**
	W	18.3	5.0	0.064
	Connect.5km	**10.7**	**4.4**	**0.044**
	Big trees	8.3	5.5	0.070
	Bicycles	3.1	2.3	0.074
Forest birds	**Grass (%)**	**49.1**	**14.5**	**0.034**
	Bicycles	**12.9**	**4.8**	**0.020**
	Shrub	9.6	4.4	0.052
	R	4.5	2.5	0.082
	B	6.8	5.4	0.052
Water birds	Grass (%)	26.9	5.5	0.078
	F	6.1	2.1	0.068
	Pedestrians	9.8	4.7	0.074
Carnivorous birds	**Pedestrians**	**39.7**	**9.9**	**0.034**
	Bicycles	6.2	2.5	0.096
Omnivorous birds	**Grass (%)**	**54.1**	**17.7**	**0.034**
	B	**11.4**	**4.6**	**0.050**
	Bicycles	**9.9**	**6.9**	**0.038**

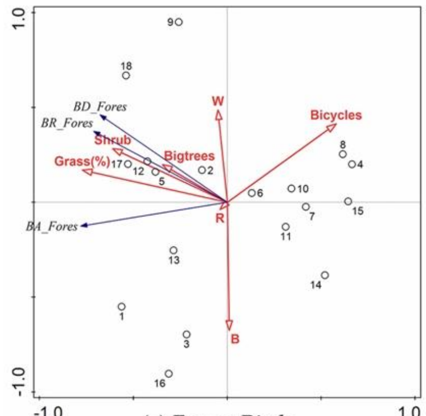
Note: F= forest land, G= grass land, W= water, B= buildings, R= roads, Connect.5km= landscape connectivity with a distance threshold of 5 km.



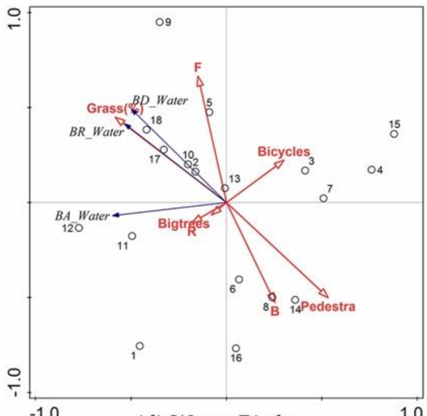
(a) All Birds



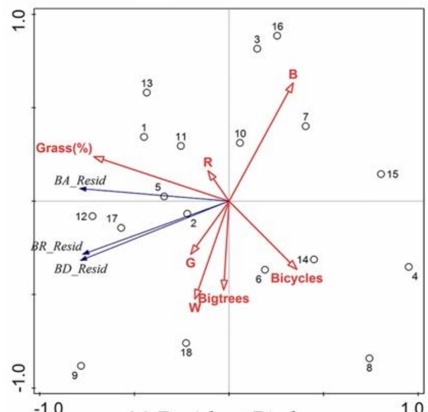
(b) Top10 Birds



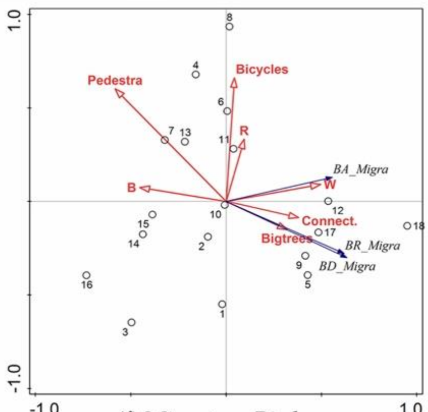
(c) Forest Birds



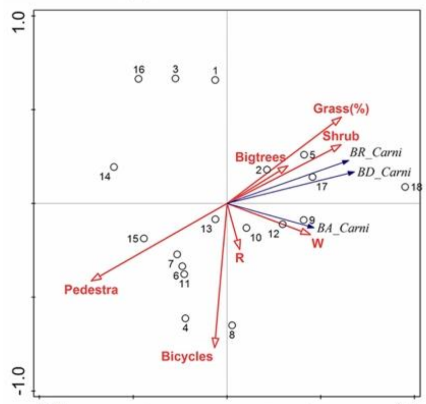
(d) Water Birds



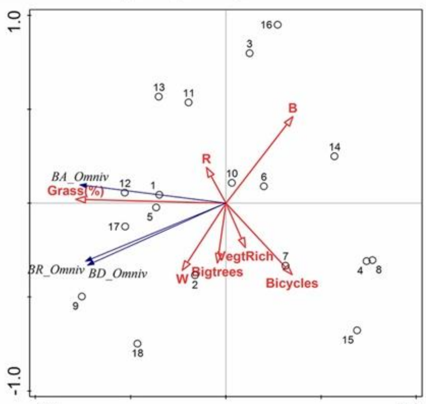
(e) Resident Birds



(f) Migratory Birds



(g) Carnivorous Birds



(h) Omnivorous Birds

Figure 4-6 Redundancy analysis.
Triplots show correlation between environmental variables and avian biodiversity, including: (a) all birds; (b) the top 10 birds; (c) forest birds; (d) water birds; (e) resident birds; (f) migratory birds; (g)carnivorous birds; (h) omnivorous birds. Seven local-scale variables and six landscape-scale variables were involved in the analysis: Grass= the coverage of grass; Shrub= the coverage of shrub; VegtRich= the vegetation richness; Bigtrees= the number of big trees; Pedestra= the number of pedestrians; Bicycles= the number of bicycles; Motorbik= the number of motorbikes; F= forest land; W= water area; G= grassland; B= building; R= road; Connect_= landscape connectivity. Three biodiversity indices: BA= bird abundance; BR= bird richness; BD= bird diversity. The top 10 birds: *Passer montanus (PassMont), Anas platyrhynchos (AnasPlat), Cyanopica cyana (CyanCyan), Pica pica (PicaPica), Tachybaptus ruficollis (TachRufc), Hirundo rustica (HirnRust), Corvus dauuricus (CorvDauu), Anthus spinoletta (AnthSpin), Pycnonotus sinensis (PycnSine), Egretta garzetta (EgrtGarz)*.

4.4 Discussion

4.4.1 What can be concluded from the responses of migratory birds?

As shown in **Table 4-1**, the responses of migratory birds are slightly different with others, but quite informative. The migratory bird is the only group that responded to landscape connectivity (10.7%, p=0.044). For birds who need to migrate, what factors are most attractive for them? In riparian areas of Tsing River, most migratory birds are water bird or forest bird. They usually migrate in summer, which are called summer birds. They fly for a long time, trying to find a warm place to breed their next generation. Therefore, it is not difficult to understand why the area of water and the number of big trees present an effect on biodiversity of migratory birds. Basically, migratory birds need to find water sources and food for living; second, big trees are usually regarded as a safe place to build a nest.

4.4.2 The dispersal distance of birds and landscape connectivity

Landscape connectivity is a metric to simulate the connection between habitats in the eye of species. In the existing research, several studies assessed landscape connectivity for multiple species by defining species groups and selecting focal species or virtual species for each group (Hostetler. 1999; Moudrý et al. 2017; Bernard et al. 2018). However, seldom studies have included dispersal distances of birds in landscape connectivity assessment. In this case, we proposed an approach of defining the distance threshold of landscape connectivity.

First, the dispersal distance of each bird species should be identified. Two studies (Foltête et al. 2012; Mimet et al. 2016) modeled the dispersal distance of birds by the taxonomy, with which the dispersal distance can be calculated based on foraging guild and body mass M (in kg); the equation used to calculate this was: pM0.63 (p = 13.1). However, when we verified this equation using observed dispersal distance data collected from the literature, it failed to predict the dispersal distance. We proposed a method synthesizing the result from the literature regarding the dispersal distance of birds and optimized the existing calculation formula. 1) group all birds according to their family; 2) calculate the parameter p of each family according to the observed data from the literature; 3) figure the dispersal distances of the rest species in the family using the parameter p calculated in the last step. The complete results of dispersal distance and references are shown in **Table A5**.

Second, to determine the optimal threshold of landscape connectivity, when landscape connectivity can best represent the actual bird diversity, we group birds based on their dispersal distance: 0-1 km, 0-2 km, 0-3 km, 0-4 km, 0-5 km. Accordingly, we set different distance threshold of landscape connectivity (1 km, 2 km, 3 km, 4 km, 5 km) for scenario analysis, and compare the correlation between landscape connectivity and biodiversity of birds (**Table A6**). The most significant correlated scenario implies that the dPC metric can better simulate the real dispersal distance of species under the setting of that threshold. When the distance threshold is within the range from 2 km to 5 km, the correlation between landscape connectivity and biodiversity of birds is similar and significant. So it is inferred that, in this case, the distance threshold should be set between 2 km to 5 km, to ensure landscape connectivity can represent the biodiversity of birds. This approach is repeatable, applicable, and could be useful for determining the linkage threshold in the assessment of landscape connectivity and guide city planners in the conservation of multiple species.

4.4.3 Significant variables and implication for bird conservation

Avian biodiversity is affected by combinations of environmental factors (Tannier et al. 2012). We discussed the impact of environmental variables on bird communities in urban riparian areas, our findings provide evidence for bird conservation in urban riparian areas. If we compare the explanation of all environmental variables, it is obvious that local-level variables explain much more than regional-level variables. In other words, in general, bird species tend to have a more significant response to environmental changes happening in local habitats (e.g. human disturbance) than that happening in surroundings (e.g. land cover changes).

The coverage of grass on the ground had a most significant, positive influence on birds. It may due to the proportion of birds feeding on the grassland, which is much higher than that of other foraging zones. Besides, the positive influence of large trees and the coverage of shrubs on birds agrees with the results of Huang et al. (2015).

In riparian areas of Tsing River, bird communities were more sensitive to the number pedestrians and bicycles than that of motorbikes and cars. Li et al. (2005) found that bicycles have more negative effects on birds than pedestrians in parks. Surprisingly, the top 10 birds responded less significantly to the disturbance of pedestrians than the whole bird communities, which may be due to the adaptation of common birds to urbanized environment with human disturbance.

At the regional level, among all land-cover types, the area of forest and water, playing as a green surrounding environment in an urbanized area, affect the biodiversity of birds positively. Conversely, the area of buildings and roads had a negative influence. It is undeniable that urbanization damages biodiversity (Huang et al. 2015; Buelow et al. 2017). The species list shows that water bird and forest bird occupy a lot in bird communities, which may explain the reason why the area of forest and water are positive variables. Meanwhile, it also due to the rough classification of land cover. Since land cover classification is not as specific as that in studies of ecology, which may conclude more information of vegetation. For example, forest land can be divided into coniferous forest and broad forest, or primary forest and secondary forest. Moreover, some bird species may rely on a certain vegetation type. Unfortunately, these details were not considered in this study due to data availability. If we can involve more detailed land cover types, the explanation of forest land would be decreased.

During riverbank maintenance, improving grassland coverage can not only prevent soil erosion, but also provide food (plant seeds and insects) for birds; planting more large trees is also an important strategy that can offer a wider range of options for nesting birds. In addition, increasing green spaces and reducing pavement and building density near riparian areas are efficient strategies, providing a greener environment for bird communities, which agrees with Pei et al. (2018) that increasing green spaces have positive influences on bird diversity in Beijing. Future research could not only include birds but also insects, reptiles, and mammals with relevant empirical evidence of their dispersal distance. A long-term species survey is also needed, regarding that the responses of birds to environmental changes at regional level may be hysteretic (Groffman et al. 2006).

4.5 Summary of Chapter 4

This study assessed the responses of bird species to the local- and regional- level variables in riparian areas of Tsing River, which flows through urban and suburban areas of Beijing. Among all environmental variables, we found that the coverage of grass has the most positive effect on biodiversity of birds, while the number of bicycles and pedestrians have the most negative effect on biodiversity of birds. In general, local-level variables are more influential for avian biodiversity than landscape-scale variables.

In addition, our findings agree with the previous studies that the biological traits of birds should be considered in discussion of species-habitat relationship; the fine scale of an environment variable should be clarified case by case. The optimal distance threshold of landscape connectivity was identified as 5 km, according to the dispersal distance of birds in the study area. Our findings provide new data for bird conservation in riparian areas of cities, which can be applied to vegetation management and landscape planning for biodiversity conservation.

Chapter 5 Response of bird communities to environmental changes based on long-term observation data in Beijing

This Chapter will be accessible until September 18, 2023.

Chapter 6 Conclusion and prospects

6.1 Conclusion

6.1.1 Summary

Compared with the widely applied landscape connectivity, this study proposed a new approach for evaluating urban green spaces by clarifying how variables of both local habitats (i.e., vegetation composition and human disturbance) and regional environment (i.e., land cover, landscape pattern) affect the biodiversity of bird communities. The biological characteristics of bird species, such as dispersal ability, habitat type, and foraging guilds, were analyzed. This study also tested the relative accuracy of landscape connectivity in indicating the biodiversity of bird communities using long-term bird observation data, thereby filling the current research gap. Key results are summarized in **Table 6-1**. In conclusion, this study provides an insight into the evaluation of the ecological efficiency of urban green spaces in maintaining the biodiversity of birds based on the response of bird species to environmental variables. The findings of this study can assist urban green space planning in the future.

All the key objectives stated in Chapter 1 were met with the case studies stated in Chapters 3, 4, and 5, in the order of gradually including species from easy to complex, regarding focal species, one-year bird census data, and long-term bird census data, respectively. In Chapter 3, the introduction of species was realized by modifying the parameters in the original model of landscape connectivity, which is based on the ecological characteristics of species, namely habitat type and dispersal ability. In Chapter 4, the responses of bird communities to various environmental variables in riparian areas of the Tsing River were applied to evaluate the ability of urban green spaces to support the biodiversity of birds.

Landscape connectivity has been widely adopted in previous studies as it enables a quick identification of the importance of each green space in relation to maintaining the connection of the whole ecological network, which is regarded as representing biodiversity (Baguette et al. 2013). In terms of the increasing skepticism that ecological processes should be involved in landscape ecology, case studies were conducted as a trial to integrate the information and knowledge of species with urban green space evaluation. In this study, landscape connectivity (dPC) was optimized by assigning distance threshold and green space based on the dispersal ability and habitat type of focal species. Our analyses demonstrated that landscape connectivity is influential for migratory birds; however, there are limitations to using landscape connectivity for representing the biodiversity of birds in small green spaces in highly urbanized areas.

In conclusion, this study tested two methods of evaluating urban green space: from perspectives of landscape connectivity and the responses of bird communities to environmental variables (i.e., vegetation composition, human disturbance, land cover, and landscape patterns). It demonstrated the necessity of including species census data in the evaluation of urban green spaces. Approaches using empirical bird data are repeatable and applicable to urban areas worldwide if there are accessible and reliable species data based on long-term monitoring.

Table 6-1 Summary of the results (except Chapter 5)

		CHAPTER3	CHAPTER4
Study Area (Scale)		Haidian District, Beijing (regional level)	Tsing River, Beijing (local and regional level)
Data Used		Remote Sensing Image (Gaofen-2)	1) Remote Sensing Image (Gaofen-2) 2) One-year empirical bird data (bird survey) 3) Environment data (field surveys)
Analysis		Landscape Connectivity Analysis	Redundancy Analysis
Result (Reliability Level*)	High	Results of urban green space evaluation from the perspective of the focal species (*Nyctereutes procyonoides, Phasianus colchicus)* were different, e.g. some agricultural lands are important (according to the value of landscape connectivity) for *Phasianus colchicus,* but not important for *Nyctereutes procyonoides.*	The coverage of grass and the number of pedestrians/ bicycles passing by were more influential for bird communities than land covers and landscape connectivity in most bird groups divided by habitat type, residence type, and foraging guilds. For example, the coverage of grass explained 49.1% of the variation of forest birds and 45.4% of resident birds.
	Middle		Landscape connectivity only showed its effect on the variation of migratory birds (explained= 10.7%, $p< 0.05$) among all bird groups.
Key Conclusion		The distance threshold and potential habitats were assigned based on the dispersal ability and habitat types of the focal species respectively, which provides a new perspective of evaluation urban green space compared with previous landscape connectivity assessment using a default value such as 500 m as the distance threshold of connectivity.	It is necessary to consider the response of various bird groups to both local variables (e.g. the coverage of grass, the number of bicycles passing by) and regional variables (e.g. area of waterbody, landscape connectivity).
Further Improvement Needed (Limitations)		· More empirical data of the focal species in this study are needed. · If we can obtain more census data and examine the predicted biodiversity, the results will be more reliable.	· If we can make long-term surveys (e.g. 10 years) along Tsing River, the results will be more reliable.

**Notes*: the division of "high" and "middle" is base on the reliability of the result, affected by data source, samples, statistics, etc.

6.1.2 Advantages of multi-scale analysis

In addition to spatial pattern analysis and species introduction analysis, this study also evaluated green space from different scales. Scale is the key to understanding urban green spaces from the point of view of a species (Hostetler 1999). The ecological process is unique for each bird, meaning that the response of a group of birds to environmental changes is complex; thus, the "scale of effect" (Jackson and Fahrig, 2012) of each environmental change for a certain (group of) species may be different (Marzluff et al. 2012). For example, the land cover change from forest land to built-up area is a regional environmental variable, which can also be regarded as a synthetic result of vegetation deterioration and an increase in impervious surface in local habitats. In addition, bird species affected significantly by vegetation deterioration at the local level do not necessarily respond to the conversion of forest land to built-up areas at the regional level. Therefore, many empirical studies discuss the responses of species to environmental changes at multiple scales and make conclusions of the fine scale that environmental changes can best describe the variation in species (abundance, richness, diversity, etc.) (Miguet et al. 2016).

At the local level, the influence of various natural and social factors on species can be investigated more precisely. We found that the coverage of grass and the number of pedestrians or bicycles passing through the site were the most significant factors in the case study of the Tsing River. We then grouped bird species according to their biological characteristics, such as habitat preference, feeding guilds, and residential type. The results for each bird group suggest that the response of birds with different characteristics to environmental changes is different. For example, migratory birds and carnivorous birds are significantly negatively affected by the number of pedestrians, while other birds are not. However, more evidence is needed to test the universality of environmental factors at the local level. There is currently a research gap in exploring whether the impact of environmental variables on a group of species at a local level in a given site has significance for other sites.

At the regional level, insights into various aspects of urbanized areas, such as land cover, landscape pattern, population, and economic development can be investigated to clarify their far-reaching effects on the biodiversity of bird communities. This study focused on the effect of land cover and landscape patterns on birds. Using multiple scale analyses, the results showed that land cover changes had the most significant effect on bird richness and bird diversity when measured within a buffer of 500 m, while landscape pattern changes showed the most significance when measured within a buffer of 1000 m. Further studies on species-environment relationships at the regional level would provide us with a synthetic effect of urbanization on biodiversity and contribute to more effective land use management and urban planning for biodiversity conservation.

6.2 Research limitations

This study provides an insight for evaluating urban green space from the perspective of species, based on a comprehensive understanding of how environmental changes resulting from urbanization affect the biodiversity of bird communities. However, there are some limitations.

First, the characteristics of birds were included in the case study of the Tsing River as a reference to classify birds into groups, but we did not make further distinctions on the composition of bird communities in different seasons because of the short time period of the bird survey. A study by Jokimäki et al. (1998) showed that birds such as *P. domesticus* and *Columba Livia Domestica* are positively related to human population density and successfully adapted to urbanized environments, owing to their omnivorous feeding guild. In contrast, species living mainly in coniferous forest habitats are unable to use deciduous-dominated urban parks in winter, such as *Dendrocopus major* and *Parus Montanus*. In addition, the composition of bird communities may vary with habitat changes resulting from human disturbance. For example, avian feeding guilds may be affected by human disturbances such as selective logging (Gray et al. 2007), where granivore abundance increases significantly and insectivore and frugivore abundance decreases significantly. More precise information on both bird communities and environmental changes would lead to a deeper understanding of their relationship.

Second, there are limitations in citizen science data. Lack of biodiversity data based on long-term observation is a common problem worldwide, although several countries are currently conducting similar projects. Citizen science data is regarded as an important data source to collect wide-range species information. However, the utilization of citizen science data has a limitation in that the distribution of observation sites depends on data contributors; thus, observation sites may be unevenly distributed. In Beijing, because of the unbalanced development between the north and south of the city, the establishment of urban green spaces and bird observations are similarly uneven. There are more bird empirical data for the north of central Beijing than the south, weakening the reliability of results. To solve this problem, bird information from other data sources should be included in supplementary data; in addition, missing bird data can be complemented by additional bird surveys.

Third, although this paper reveals the response of bird communities to environmental change at multiple scales, the impact of other external environmental factors, such as population density at the regional level, is not considered. Liu et al. (2017) obtained bird census data in different seasons, including 140 parks in 17 cities, by extracting a bird species list from 33 published sources, and selected 96 parks with seasonal survey data for analysis. Their results showed that the correlation between population density and bird diversity was not significant. However, Geschke (2018) found an impact of population density on bird diversity and proposed an optimal land allocation model. These two examples show that the impact of population density on bird diversity has not yet been determined. Therefore, the effects of population density on species diversity and the methods of determining population density and their accuracy should be discussed for each individual research site. In this study, because of data limitations, we did not include population density in the discussion.

6.3 Future prospects

6.3.1 Species-human- environment relationship

The dualism of species-environment should be changed into the ternary relationship of human-species-environment. The importance of maintaining sufficient green space in a city to promote the interrelations between people and nature is well-known. However, most studies focus on the response of species to changes in the natural environment but neglect the role of humans in environmental changes.

From a physical point of view, human activities have an impact on the environment and biodiversity, such as traffic noise and interference on the microbial environment. In addition, Zhu et al. (2019) demonstrated that urban plant diversity in Haikou, China is affected by not only social, economic, and physical environmental factors, but also by green land management behaviors (such as weeding, fertilization, and watering times). Anderson et al. (2019) analyzed the science of the total environment, which showed that urban plant richness and vegetation height were mainly affected by social factors, while vegetation area and evenness were mainly affected by biophysical factors. In addition, there was an interaction between social factors and biophysics. At present, there are two hypotheses about social factors as an impact mechanism of urban vegetation diversity: 1) the land-use hypothesis, stating that areas with more human input have less vegetation coverage, while areas with less human input and management have more vegetation; and 2) the wealth effect hypothesis, stating that areas in a rich city have higher vegetation coverage and diversity.

From a psychological point of view, residents' perception, that is, people's subjective perception of green space and evaluation of the value of green space protection, affects urban planning and management and potentially affects the environment and biodiversity. Wang et al. (2019) showed that biodiversity has a strong positive impact on the perceived uniqueness (conservation value) of urban forest patches in Helsinki, Finland, with higher biodiversity patches perceived as more unique. Improving people's perception of urban forests would help enhance the connectivity between people and the environment and arouse people's awareness of environmental protection. Therefore, future research should focus on building a mechanism to clarify human impact on the environment to fully understand the impact of urbanization on biodiversity.

6.3.2 Biological conservation at multiple scales

Efficient management of green spaces should be based on the understanding of multiscale diversity preservation. For specific species or communities, we can determine the distribution of species or community composition through field investigation; at the same time, we can fully investigate the field environment and extract environmental factors. Alternatively, we can extract environmental changes, such as land cover, climate, and air pollutants, at the macro level through remote sensing images. We can then obtain long-term species change data from public scientific data or long-term survey data and determine environmental factors with greater influence through statistical analyses. Local impacting factors should be implemented in the short-term plan of urban planning and management, such as controlling the local flow of people to reduce human interference. Factors influencing landscape affect long-term urban planning and management decision-making. For example, the intensity of urban construction and development should be controlled within 500 meters outside historical forests, such as summer palaces. In urban ecological restoration and protection, green land cannot be simply regarded as an island isolated by the surrounding artificial surface types. In addition to protecting the natural patches themselves, the improvement of the quality of the surrounding artificial green land is also conducive to the restoration of the urban natural ecosystem.

6.3.3 Including multiple species in an urban green space evaluation

Evaluating and protecting urban biodiversity is still a challenge for many scientists. This is mainly due to: 1) priority of habitat data and species occurrence data (highly dependent on the quantity and quality of data) in urban areas, such as the use of birds as indicators in this study, and 2) complexity of urban environment and people's space caused by human factors (such as factors other than land cover and landscape pattern) in urban areas. The effects of heterogeneity on species distribution lack sufficient cognition.

We investigated multiple environmental variables and attempted to add more details on ecological processes. However, the definition of urban green space should vary by species, such as the classification of agricultural lands (Ichinose et al. 2007) and coastal forests (Buelow et al. 2017). There are still many uncertainties when we identify urban green spaces from the perspective of humans rather than specific species. Future research should improve the classification of urban green space to obtain a better understanding of urban environments from the point of view of species living in urban areas.

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Appendix

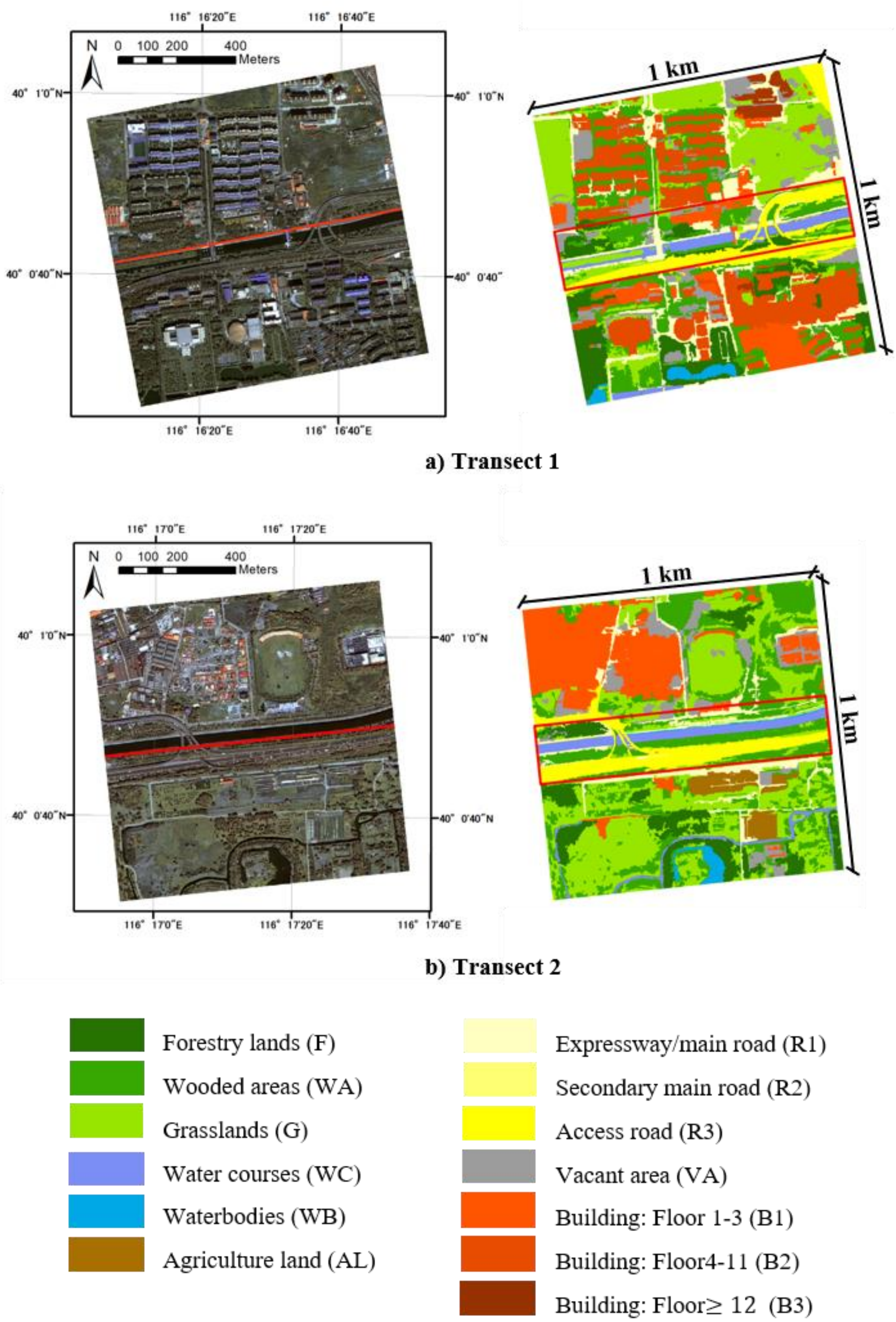


Figure A1- 1 The remote sensing image and the land cover classification based on object-based image analysis: transect 1 and 2

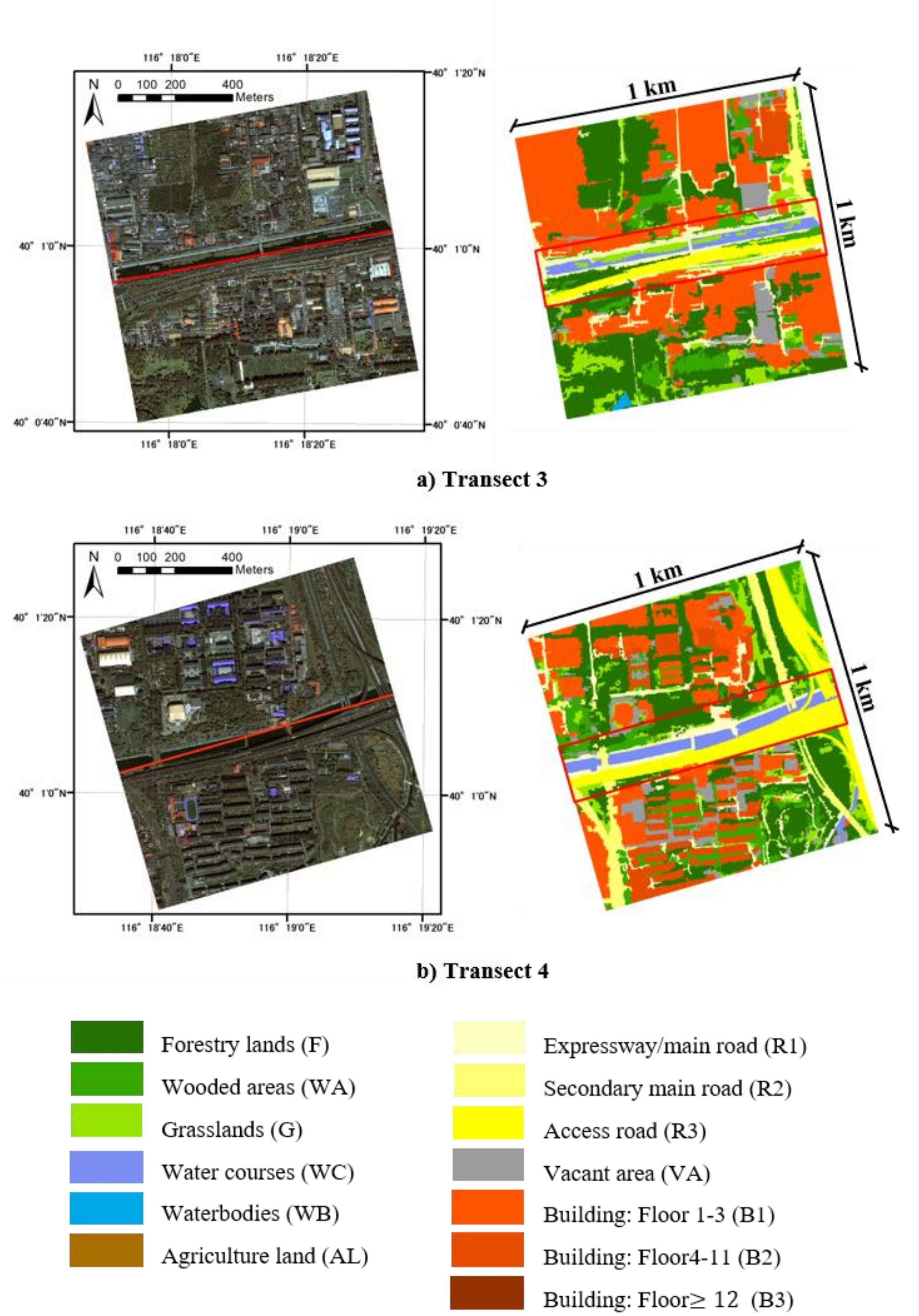


Figure A1- 2 The remote sensing image and the land cover classification based on object-based image analysis: transect 3 and 4

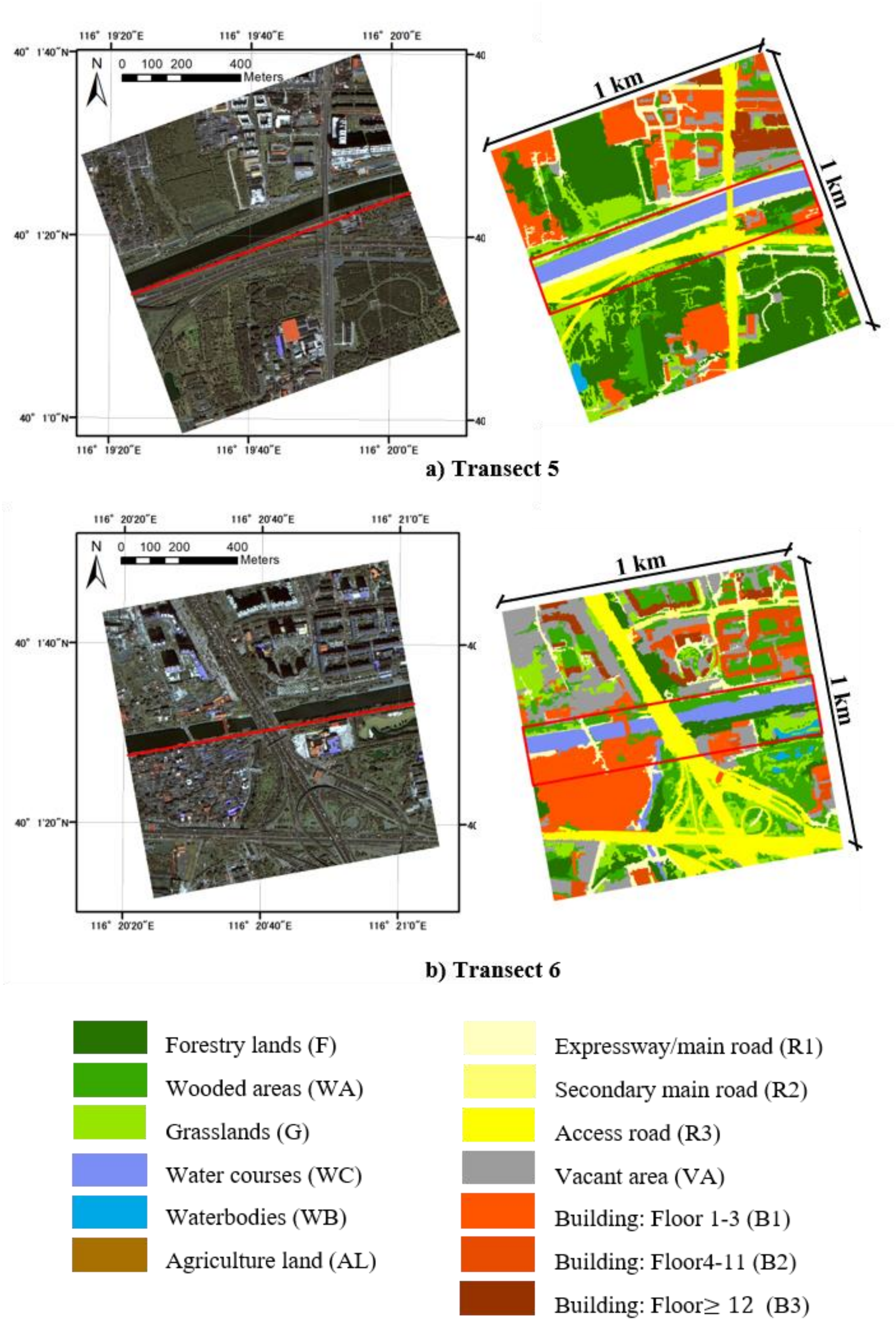


Figure A1- 3 The remote sensing image and the land cover classification based on object-based image analysis: transect 5 and 6.

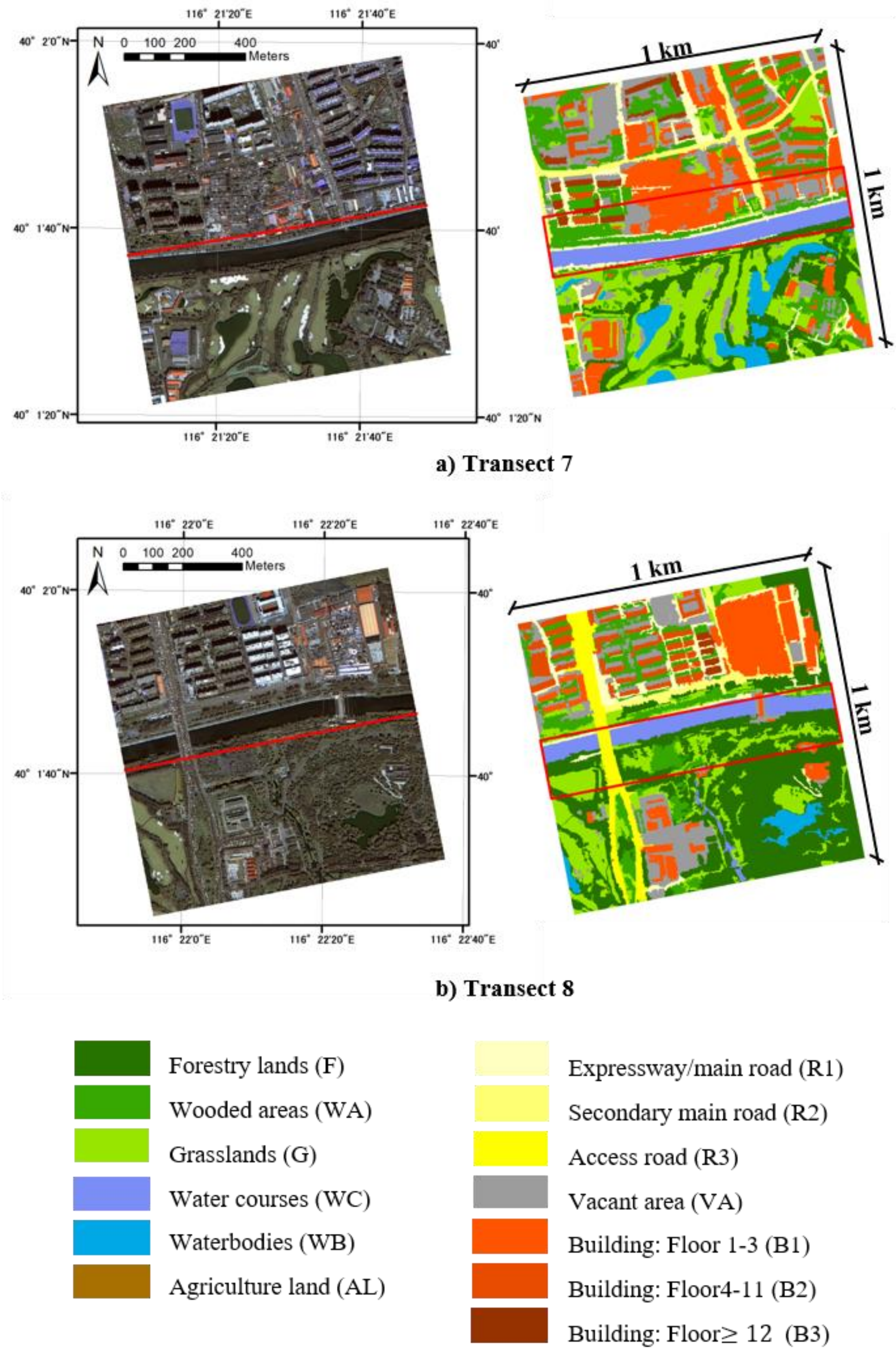


Figure A1- 4 The remote sensing image and the land cover classification based on object-based image analysis: transect 7 and 8.

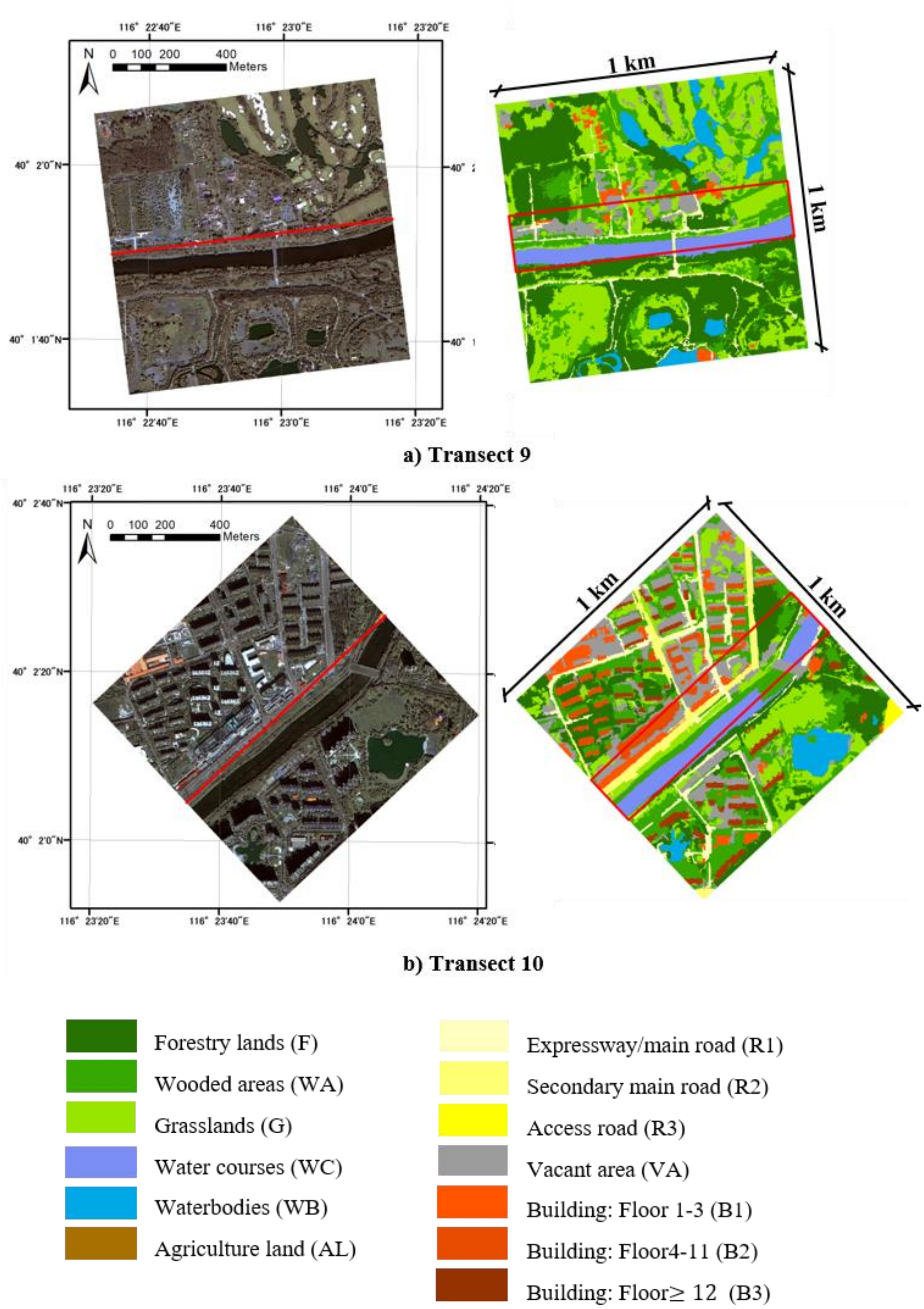
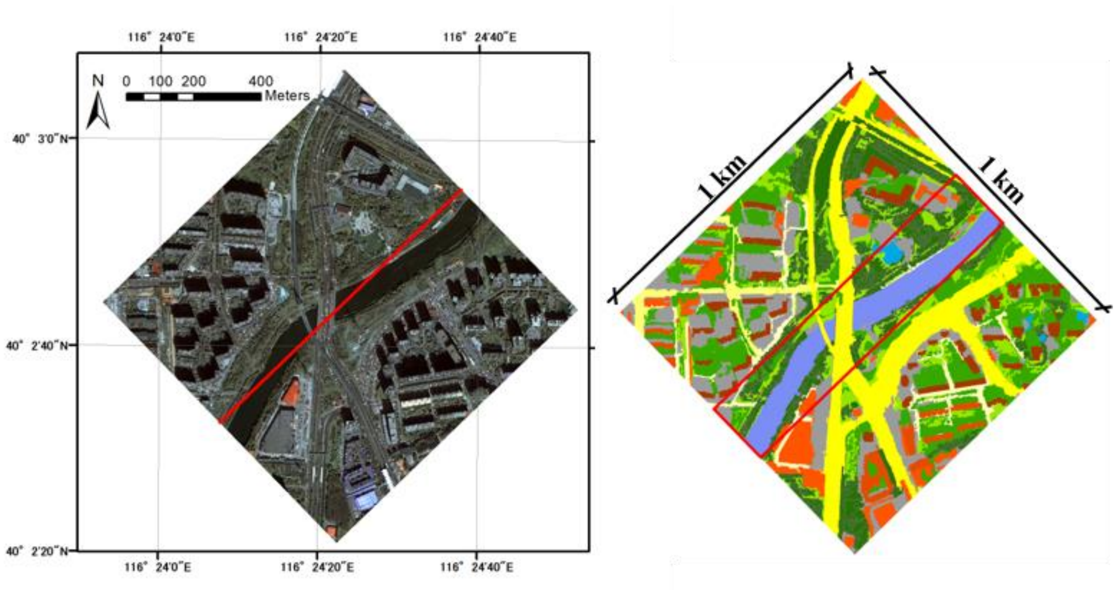
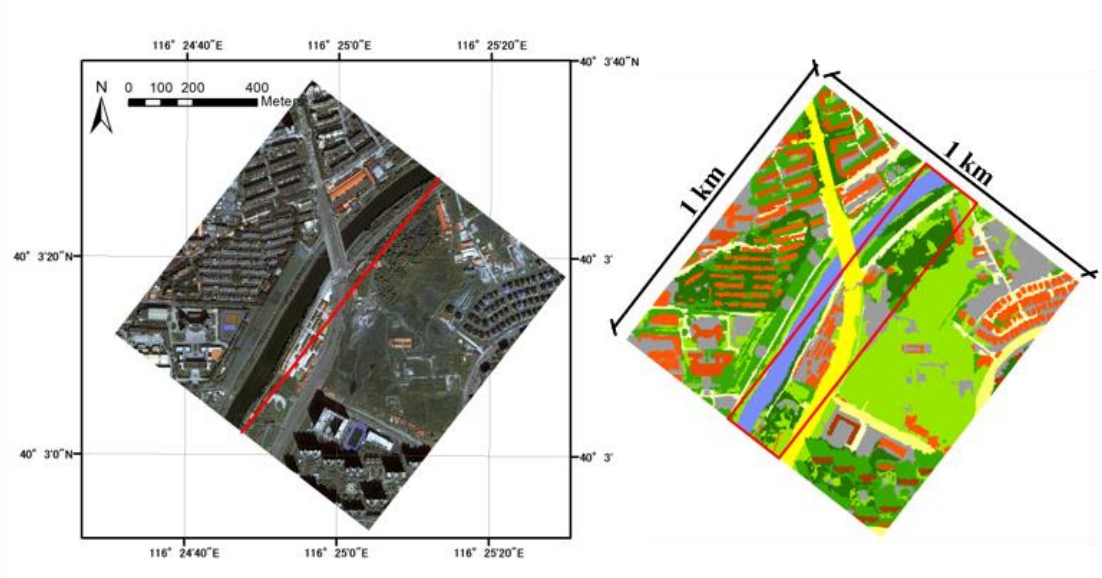


Figure A1- 5 The remote sensing image and the land cover classification based on object-based image analysis: transect 9 and 10.



a) Transect 11



b) Transect 12

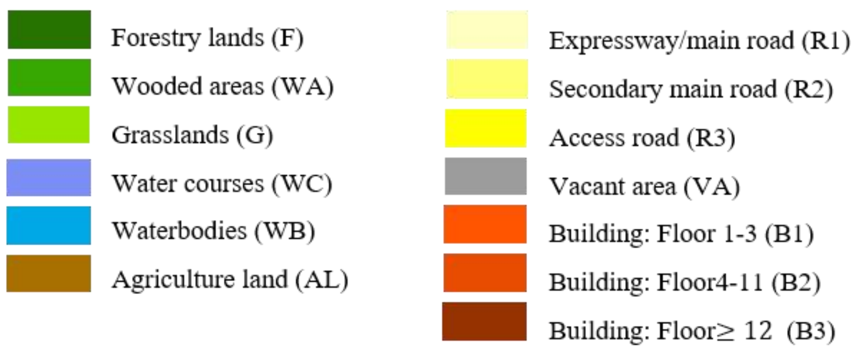


Figure A1- 6 The remote sensing image and the land cover classification based on object-based image analysis: transect 11 and 12.

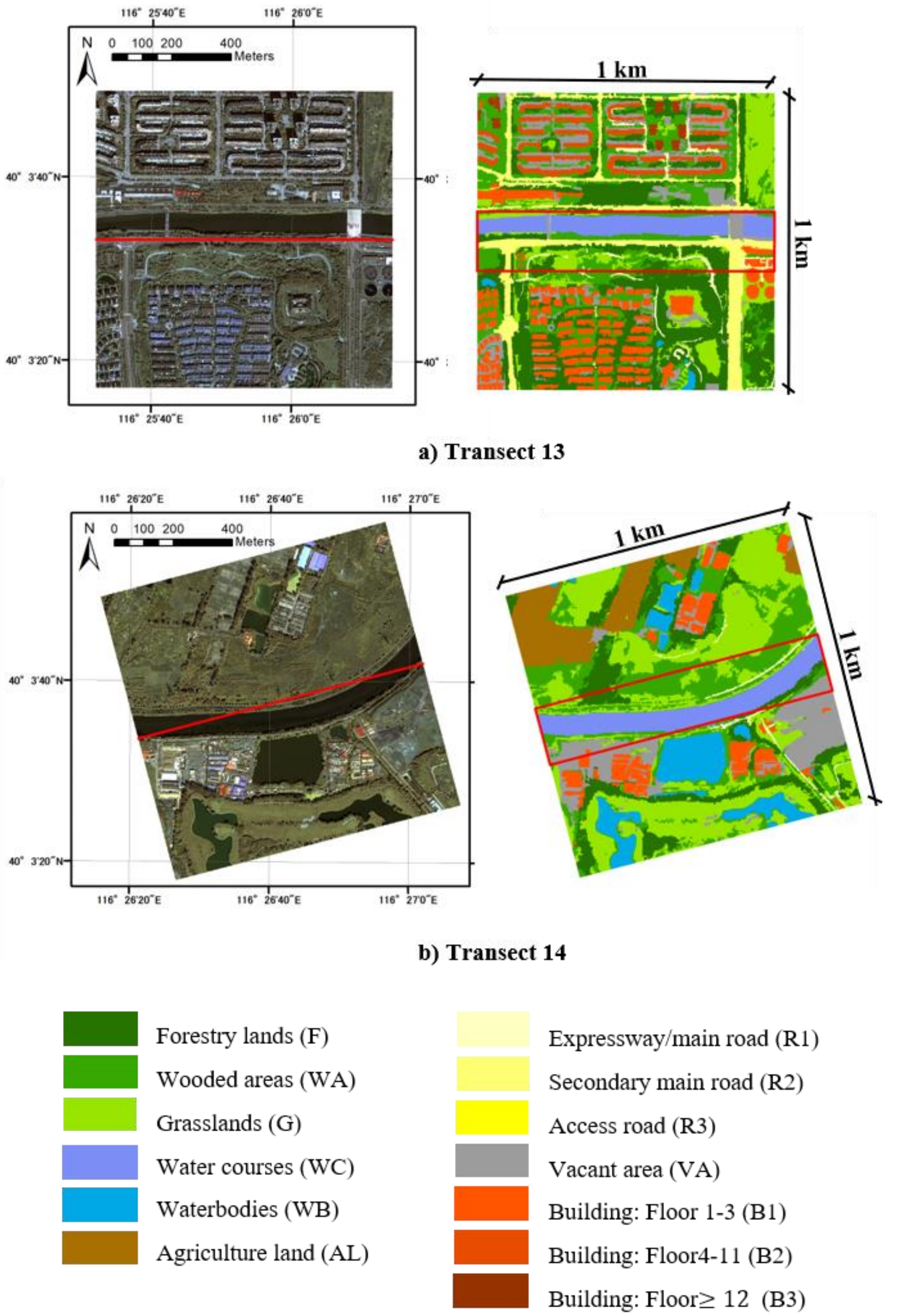
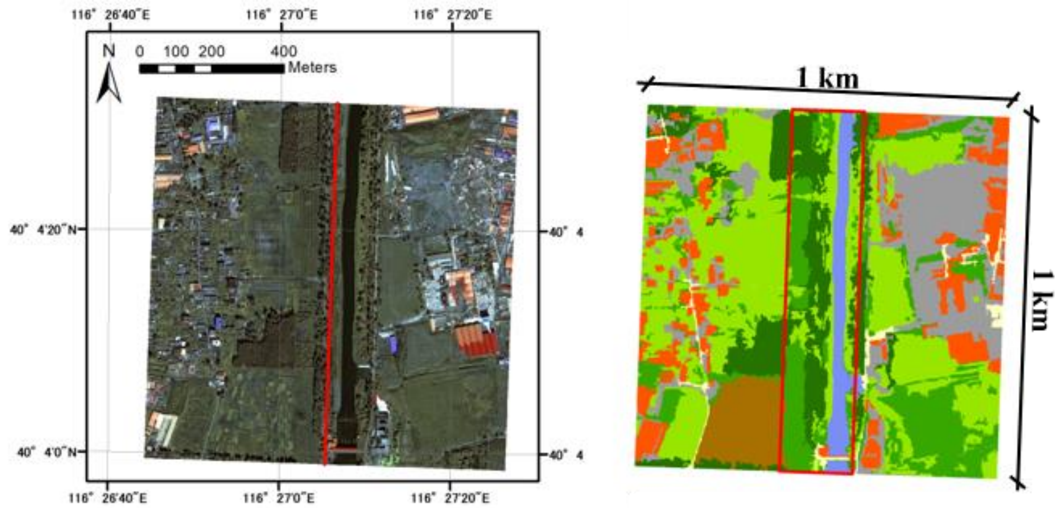
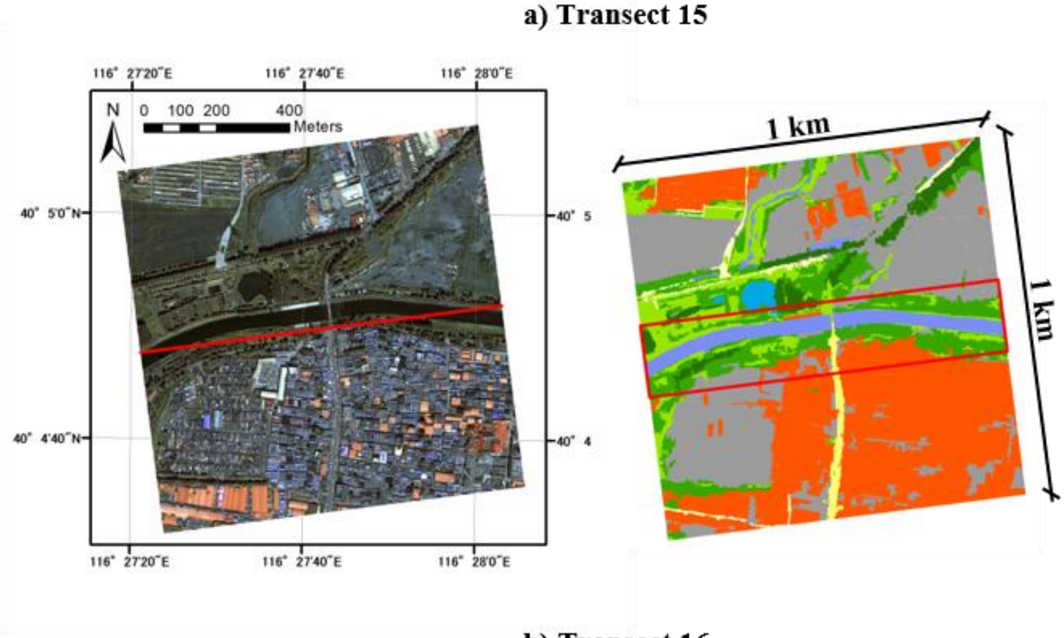


Figure A1- 7 The remote sensing image and the land cover classification based on object-based image analysis: transect 13 and 14.



a) Transect 15



b) Transect 16

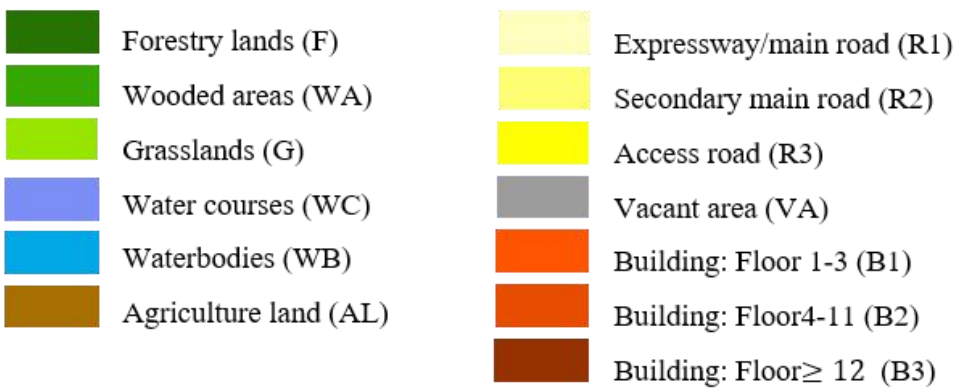
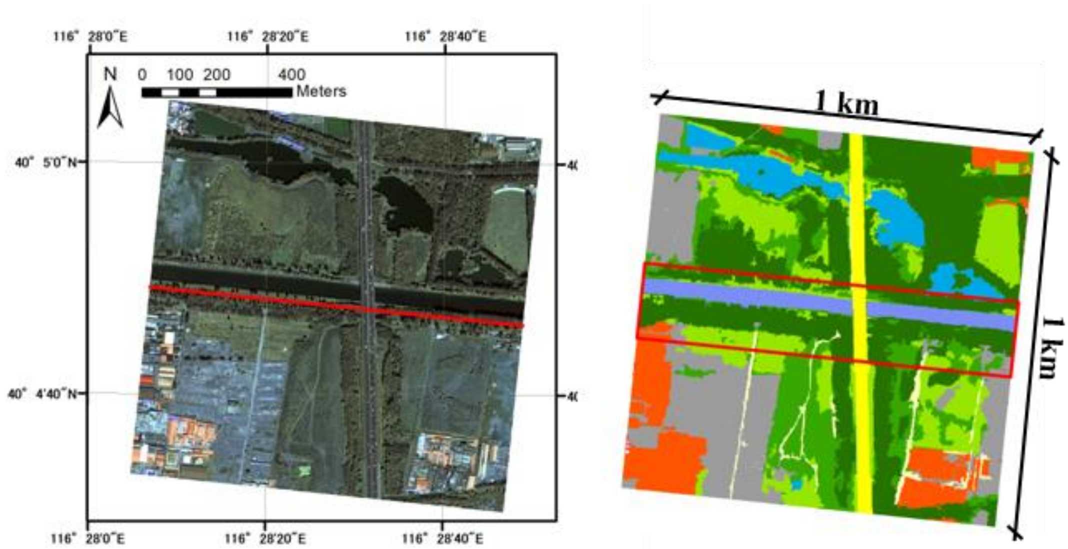
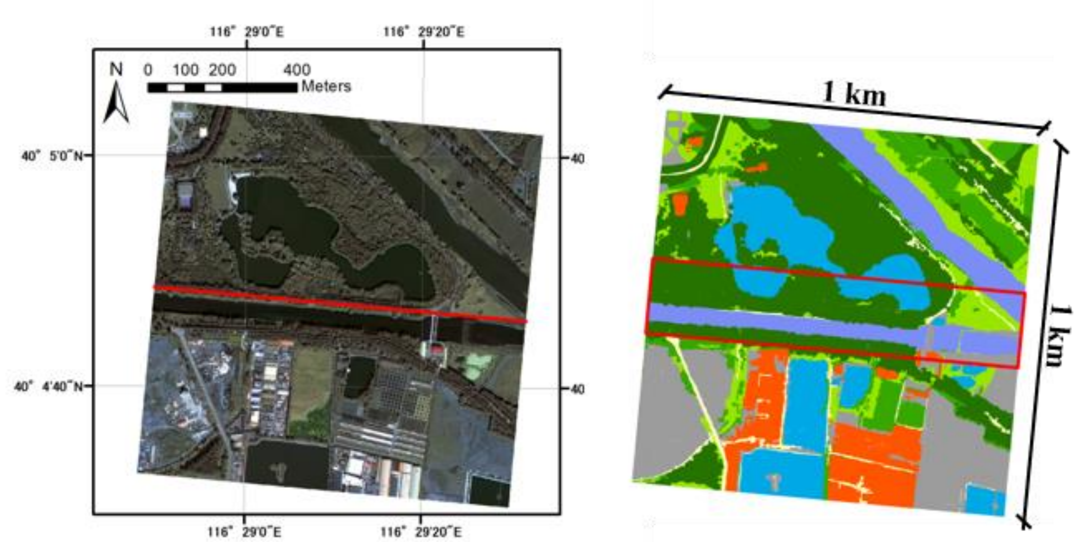


Figure A1- 8 The remote sensing image and the land cover classification based on object-based image analysis: transect 15 and 16.



a) Transect 17



b) Transect 18

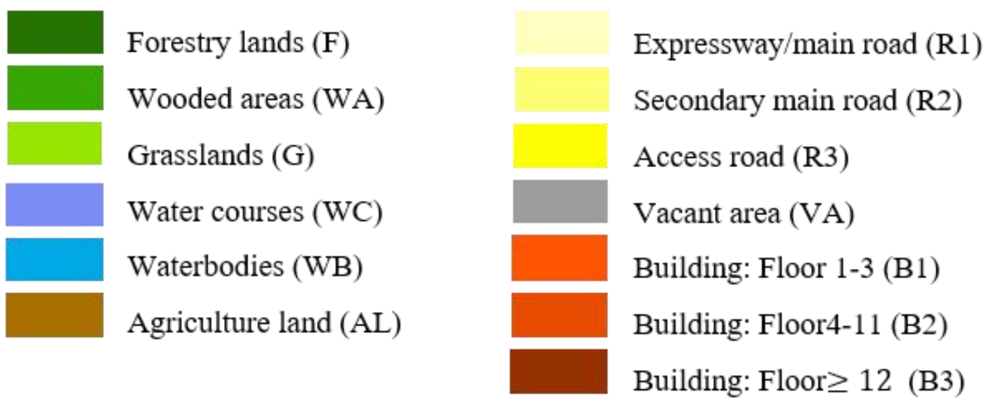


Figure A1- 9 The remote sensing image and the land cover classification based on object-based image analysis: transect 17 and 18.

Table A1 The land cover in each transect grid, corresponding to Appendix Figure A1 (unit= ha).

	TRANSECT	F	G	WA	AL	W	R1	R2	R3	B1	B2	B3	VA
Buffer= 500 m	1	8.8	17.8	17.1	0.0	3.9	9.5	0.0	7.1	6.9	19.4	1.7	7.5
	2	7.6	28.6	22.2	2.8	5.2	5.5	0.0	6.4	16.1	0.0	0.0	5.3
	3	26.7	7.2	7.3	0.0	2.5	8.2	2.0	3.5	35.4	2.2	0.0	5.0
	4	25.2	6.4	9.8	0.0	3.8	6.3	5.6	10.8	15.6	11.1	0.0	5.4
	5	32.0	12.1	8.8	0.0	6.3	7.7	0.3	9.6	13.3	2.4	2.6	5.0
	6	5.5	9.5	18.8	0.0	6.1	5.2	1.6	15.2	14.6	4.7	3.0	15.9
	7	10.3	17.4	16.8	0.0	9.6	6.5	3.4	0.0	15.8	3.8	2.0	14.4
	8	31.9	13.4	9.8	0.0	7.2	4.9	2.1	5.0	10.3	5.1	0.6	9.7
	9	32.9	31.3	16.2	0.0	10.1	2.9	0.0	0.0	1.7	0.0	0.0	4.7
	10	15.9	14.1	21.8	0.0	8.6	5.9	4.1	0.4	4.1	4.9	4.2	15.9
	11	17.1	9.4	18.4	0.0	7.0	3.3	2.0	14.9	5.7	2.5	5.0	14.7
	12	11.2	22.6	17.0	0.0	5.1	6.4	2.1	5.6	9.2	4.2	1.2	15.4
	13	26.1	9.7	20.8	0.0	6.0	1.8	9.8	0.0	10.9	4.9	0.9	9.1
	14	11.4	28.6	19.6	8.5	15.5	0.9	0.0	0.0	5.6	0.0	0.0	9.9
	15	12.3	32.8	16.9	4.9	4.0	1.5	0.0	0.0	11.0	0.0	0.0	16.7
	16	3.5	8.7	13.9	0.0	5.3	1.3	1.1	0.0	33.4	0.0	0.0	32.7
	17	26.1	17.6	16.0	0.0	10.0	0.5	0.0	3.6	9.5	0.0	0.0	16.3
	18	30.4	9.7	8.9	0.0	24.1	2.6	0.0	0.0	8.1	0.0	0.0	16.1
Buffer= 100 m	1	1.9	2.5	4.1	0.0	2.5	2.1	0.0	4.5	1.2	0.0	0.0	1.1
	2	0.8	2.2	6.0	0.0	3.2	1.7	0.0	5.5	0.2	0.0	0.0	0.4
	3	1.9	1.9	1.7	0.0	2.3	4.7	0.3	3.5	3.5	0.0	0.0	0.3
	4	3.2	1.1	1.6	0.0	3.5	2.4	0.9	6.5	0.5	0.0	0.0	0.3
	5	1.5	1.8	2.1	0.0	5.9	2.6	0.0	5.1	0.7	0.0	0.0	0.3
	6	1.6	1.4	2.8	0.0	5.2	1.0	0.0	1.7	4.9	0.1	0.0	1.2

7	0.8	0.4	3.7	0.0	5.4	2.3	0.3	0.0	4.4	0.2	0.4	2.3
8	7.7	4.0	1.4	0.0	5.0	0.4	0.0	0.9	0.2	0.0	0.0	0.4
9	2.4	4.2	4.4	0.0	5.0	1.1	0.0	0.0	0.3	0.0	0.0	2.5
10	1.4	2.1	3.4	0.0	4.6	0.8	2.3	0.0	1.3	1.2	0.0	2.9
11	5.9	2.2	1.0	0.0	6.7	0.4	0.0	1.6	0.5	0.0	0.0	1.6
12	3.9	3.5	1.7	0.0	3.5	1.3	0.0	2.6	1.2	0.0	0.0	2.3
13	3.2	2.3	4.2	0.0	5.7	0.5	2.8	0.0	0.6	0.0	0.0	0.7
14	1.2	3.4	6.2	0.0	7.2	0.3	0.0	0.0	0.0	0.0	0.0	1.7
15	5.9	4.6	5.2	0.2	3.9	0.2	0.0	0.0	0.1	0.0	0.0	0.0
16	0.7	3.1	6.7	0.0	4.0	0.0	0.2	0.0	2.8	0.0	0.0	2.5
17	8.9	2.7	2.4	0.0	4.0	0.1	0.0	0.7	0.3	0.0	0.0	0.8
18	9.6	1.4	0.2	0.0	7.4	0.3	0.0	0.0	0.1	0.0	0.0	1.0

Note: F= forestry land; G= grassland; WA= wooded area (mainly includes shrubs and grassland, as well as a couple of trees); AL= agricultural land/ cropland; W= waterbody/ water course; R1= expressway/ main road; R2= secondary main road; R3= access road; B1= building with 1-3 floors; B2= building with 4-11 floors; B3= building with more than 11 floors; VA= vacant area.

Table A2 Vegetation composition and human disturbance investigated from field surveys.

Transect	Ground-level			Shrub-level	Tree-level			Overall	Human Disturbance			
	Grass (%)	Leaves (%)	Bared (%)	Shrubs (%)	Number	Height (h> 10m)	Crown (d> 6m)	Richness	Pedestrians	Bicycles	Motorbikes/ Scooters	Cars
1	0.55	0.37	0.08	0.08	11.33	1.33	2.67	13.00	3.33	1.00	5.00	11.67
2	0.68	0.17	0.15	0.05	14.33	1.67	2.67	7.00	2.00	3.67	2.33	3.00
3	0.63	0.12	0.25	0.04	14.33	1.00	1.67	5.33	2.33	1.67	6.33	7.67
4	0.42	0.22	0.30	0.03	13.67	1.00	2.67	5.67	12.33	23.00	25.67	24.33
5	0.75	0.07	0.18	0.07	17.67	0.67	1.67	5.67	1.67	2.33	5.67	6.00
6	0.50	0.02	0.48	0.03	13.00	2.67	3.33	5.00	23.33	5.67	6.33	7.33
7	0.58	0.13	0.28	0.06	10.33	0.00	0.00	6.33	19.67	2.33	0.67	0.00
8	0.30	0.07	0.63	0.06	13.33	4.33	6.67	5.33	5.67	12.67	15.33	42.00
9	0.68	0.10	0.22	0.08	12.67	2.67	3.33	3.67	1.33	3.67	10.00	18.00
10	0.65	0.05	0.30	0.03	16.67	0.00	0.00	3.67	1.67	3.00	13.33	21.67
11	0.58	0.07	0.35	0.01	12.67	0.00	0.00	3.33	5.33	4.00	5.67	14.33
12	0.78	0.07	0.15	0.04	12.67	3.00	6.00	3.67	2.67	3.33	1.00	0.00
13	0.60	0.22	0.18	0.09	10.00	0.00	0.67	4.00	6.67	1.33	5.67	14.33
14	0.37	0.07	0.57	0.03	12.33	1.67	2.67	3.67	5.67	0.33	0.00	0.00
15	0.52	0.05	0.43	0.00	13.00	5.00	0.67	5.00	4.33	5.00	21.33	85.67
16	0.70	0.10	0.20	0.00	10.00	1.33	5.67	2.67	6.33	0.00	0.00	0.00
17	0.77	0.13	0.10	0.00	13.67	7.00	5.00	3.67	1.33	0.67	0.67	0.33
18	0.88	0.08	0.03	0.00	17.33	10.00	14.00	3.33	0.67	0.33	0.00	0.00

Note: Environmental variables showed in the Table were collected from field surveys (September 2018). A total of 54 plots were investigated, 3 plots were randomly selected in each transect (size= 10×10 m). Thus, here shows the average value of 3 plots in each transect.

Table A3 The species list in the riparian areas of Tsing River

Family	Scientific name	Common Name	Foraging Guild	Habitat Type	Residence Type	Dispersal Distance (km)	Total Number
Podicipedidae	*Tachybaptus ruficollis*	Little grebe	O	W	S	0.39	851
Ardeidae	*Ardea cinerea*	Grey heron	C	W	T	7.50	55
	Ardeola bacchus	Chinese pond heron	O	W	S	2.36	29
	Bubulcus ibis	Cattle egret	C	CL/G/W	S T	3.14	1
	Ardea alba	Great egret	C	W	S	5.35	63
	Egretta garzetta	Little egret	O	W	S	3.51	195
	Ardea intermedia	Intermediate egret	C	W	S	4.49	8
	Nycticorax nycticorax	Black-crowned night heron	C	W	S	4.40	60
	Ixobrychus sinensis	Yellow bittern	C	W	S	1.26	11
	Botaurus stellaris	Eurasian bittern	C	G/W	S T	5.19	1
Anatidae	*Anser cygnoides*	Swan goose	O	CL/G/W	S	11.21	16
	Anser fabalis	Taiga bean goose	O	CL/G/W	S	10.48	1
	Tadorna ferruginea	Ruddy shelduck	O	CL/G/W/WA	S T	7.15	186
	Anas crecca	Eurasian teal	O	W	T W	2.88	46
	Anas falcata	Falcated duck	O	W	T	4.11	5
	Anas platyrhynchos	Mallard	O	W	R	5.19	2581
	Anas poecilorhyncha	Indian spot-billed duck	O	W	T	5.92	23
	Mergus merganser	Common merganser	O	F/W	T W	7.14	15
Accipitridae	*Milvus migrans lineatus*	Black-eared kite*	C	F/WA	T	6.15	1
	Accipiter nisus	Eurasian sparrowhawk*	C	F/WA	T W	2.62	11
	Buteo buteo	Common buzzard*	C	F/WA	T W	5.85	16
	Circus cyaneus	Hen harrier*	C	G/W/WA	T	3.75	2

Falconidae	*Falco amurensis*	Amur falcon*	C	F/W/WA	T	1.18	4
	Falco tinnunculus	Common kestrel*	C	F/W/WA	R	1.69	17
Rallidae	*Amaurornis phoenicurus*	White-breasted waterhen	O	W/WA	S	0.39	2
	Gallinula chloropus	Common moorhen	O	W	S	0.53	13
Charadriidae	*Vanellus vanellus*	Northern lapwing	O	G/W	T	6.00	6
	Charadrius placidus	Long-billed plover	O	W	T	2.78	10
	Charadrius dubius	Little ringed plover	O	CL/G/W	T	2.00	1
Scolopacidae	*Tringa ochropus*	Green sandpiper	O	G/W	T	1.82	90
	Actitis hypoleucos	Common sandpiper	C	G/W	T	5.44	6
	Gallinago gallinago	Common snipe	O	W	T	2.30	6
Laridae	*Chlidonias leucopterus*	White-winged tern	C	W	S	7.25	35
Columbidae	*Streptopelia orientalis*	Oriental turtle dove	H	CL/F/WA	S	0.72	1
	Streptopelia chinensis	Spotted dove	H	CL/G/WA	R	0.59	107
Cuculodae	*Cuculus micropterus*	Indian cuckoo	O	F/WA	S	0.54	16
	Cuculus canorus	Common cuckoo	C	F/WA	S	0.53	12
Apodidae	*Apus apus*	Common swift	C	B/F/WA	S	2.63	33
	Apus pacificus	Pacific swift	C	O	T	2.90	7
Alcedinidae	*Megaceryle lugubris*	Crested kingfisher	C	W/WA	R	4.61	2
	Alcedo atthis	Common kingfisher	C	W/WA	R	3.75	45
Upupidae	*Upupa epops*	Hoopoe	C	F/WA	R	1.98	53
Picidae	*Picus canus*	Grey-headed woodpecker	O	F/WA	R	0.97	24
	Dendrocopos major	Great spotted woodpecker	O	F/WA	R	0.61	47
	Dendrocopos canicapillus	Grey-capped pygmy woodpecker	O	F/WA	R	0.35	19
Alaudidae	*Alauda arvensis*	Eurasian skylark	O	G	T	0.17	10
Hirundinidae	*Hirundo rustica*	Barn swallow	C	B	S	0.86	591
	Cecropis daurica	Red-rumped swallow	C	B	S	1.10	70

Motacillidae	*Motacilla flava*	Western yellow wagtail	C	G/W	T	0.72	3
	Motacilla cinerea	Grey wagtail	C	G/W	T	0.70	28
	Motacilla alba lugens	White wagtail	O	G/W	T	0.89	86
	Anthus hodgsoni	Olive-backed pipit	O	F/WA	T	0.81	79
	Anthus spinoletta	Water pipit	O	CL/G/W	T	0.72	203
Pycnonotidae	*Pycnonotus sinensis*	Light-vented bulbul	O	F/G/WA	R	0.35	339
Laniidae	*Lanius cristatus*	Brown shrike	O	WA	R	0.46	15
	Lanius sphenocercus	Chinese grey shrike	C	G/WA	S	0.73	2
Dicruridae	*Dicrurus macrocercus*	Black drongo	C	WA	S	1.56	29
Sturnidae	*Sturnus cineraceus*	Grey starling	O	G/WA	R	0.35	195
	Acridotheres cristatellus	Crested myna	O	G/WA	R	0.40	42
Corvidae	*Cyanopica cyana*	Azure-winged magpie	O	F/WA	R	0.14	1312
	Pica pica	Eurasian magpie	O	F/WA	R	0.25	1288
	Corvus dauuricus	Daurian jackdaw	O	CL/F/G/WA	R	0.62	484
	Corvus macrorhynchos	Large-billed crow	O	F/WA	R	0.98	79
	Corvus corone	Carrion crow	O	F/WA	R	0.93	114
Turdidae	*Luscinia svecica*	Bluethroat	C	F/WA	T	0.40	15
	Phoenicurus auroreus	Daurian redstart	O	F/WA	T	0.43	2
	Saxicola torquata	Common Stonechat	O	G/WA	S	0.48	33
	Turdus merula	Common blackbird	O	F/WA	R	1.33	3
	Turdus naumanni	Naumann's thrush	O	F/WA	R	1.09	23
Paradoxor	*Paradoxornis webbiana*	Vinous-throated parrotbil	O	F/WA	R	1.10	18
Sylviidae	*Acrocephalus orientalis*	Oriental reed warbler	C	G/W	S	1.90	10
	Acrocephalus bistrigiceps	Black-browed reed warble	C	G/W/WA	S	0.87	7
	Acrocephalus aedon	Thick-billed Warbler	C	G/WA	T	1.78	3
	Phylloscopus inornatus	Yellow-browed warbler	C	F/WA	S T	0.87	10
	Phylloscopus proregulus	Pallas's leaf warbler	C	F/WA	S T	0.74	31

	Cisticola juncidis	Zitting cisticola	O	G/WA	T	0.87	9
Muscicapidae	*Ficedula parva*	Red-breasted flycatcher	C	F/WA	T	0.24	8
Paridae	*Parus major*	Great tit	O	F/WA	R	0.25	5
	Parus palustris	Marsh tit	O	F/WA	R	0.22	52
Ploceidae	*Passer montanus*	Eurasian tree sparrow	O	B/CL/G/WA	R	0.28	5570
Fringillidae	*Carduelis sinica*	Grey-capped greenfinch	O	WA	R	0.81	50
	Eophona migratoria	Chinese grosbeak	O	F/WA	S T	1.56	5
Emberizidae	*Emberiza spodocephala*	Black-faced bunting	O	F/WA	T	0.47	26
	Emberiza pusilla	Little bunting	O	F/WA	T	0.41	29
	Emberiza pallasi	Pallas's reed bunting	O	F/G/WA	T	0.38	21

Note: Feeding guilds: I= Insectivorous; O= Omnivorous; C= Carnivorous; H= Herbivorous. Habitat type: B= Building; CL= Cropland; F= Forestry land (with dense trees); G= Grassland; WA= Wooded area (mixture of trees and shrubs); W= Water. Residence type: R= Resident birds; S= Summer migratory birds; T= Traveling bird; W= Winter migratory birds.

Table A4 Bird census data (May 2016- April 2017)

Scientific name	**Bird census data from surveys once per month (n= 12)**											
	201605	201606	201607	201608	201609	201610	201611	201612	201701	201702	201703	201704
Tachybaptus ruficollis	2	10	3	7	5	20	81	186	239	253	36	9
Ardea cinerea	1	1	3	2	1	3	12	5	17	4		6
Ardeola bacchus	3	4	13	5	4							
Bubulcus ibis				1								
Ardea alba	4	14	8	4	33							
Egretta garzetta	5	31	39	79	41							
Ardea intermedia				8								
Nycticorax nycticorax	2	9	34	10	5							
Ixobrychus sinensis		1	7	3								
Botaurus stellaris											1	
Anser cygnoides							16					
Anser fabalis										1		
Tadorna ferruginea	1		3		2	2	9	10	101	56	2	
Anas crecca						1		11	4	22	8	
Anas falcata										4	1	
Anas platyrhynchos	42	85	135	121	80	278	290	570	433	326	200	21
Anas poecilorhyncha	5		4	1			3	3		4	1	2
Mergus merganser										15		
Milvus migrans lineatus			1									
Accipiter nisus					3		1	1	3			3
Buteo buteo								8	5	1	1	1
Circus cyaneus										1	1	

Falco amurensis	1	1		1		1						
Falco tinnunculus	1			2	1		7	2	3		1	
Amaurornis phoenicurus		2										
Gallinula chloropus		1		3	3	5	1					
Vanellus vanellus											6	
Charadrius placidus									2	7	1	
Charadrius dubius											1	
Tringa ochropus				1	9	6	14	22	10	10	18	
Actitis hypoleucos				6								
Gallinago gallinago					5	1						
Chlidonias leucopterus				33							2	
Streptopelia orientalis									1			
Streptopelia chinensis	4	7	12	11	3	11	9	12	14	6	13	5
Cuculus micropterus	6	3	7									
Cuculus canorus		7	5									
Apus apus	5	21	5									2
Apus pacificus			2									5
Megaceryle lugubris									1	1		
Alcedo atthis	8	1	4	6	4	4	7	3	2	2		4
Upupa epops	2	4		3	2	7	15	6	1	1	7	5
Picus canus	5		6			2	2	2		2	2	3
Dendrocopos major	5	2	4	4			15	3	8	2		4
Dendrocopos canicapillus	1	3	2		4	1	2			1	2	3
Alauda arvensis						7				3		

Hirundo rustica	142	163	133	60	17							76
Cecropis daurica	4	17	39	3	5							2
Motacilla flava	3											
Motacilla cinerea				1								27
Motacilla alba lugens	1	3	5	6	1	2	4	6	4	6	31	17
Anthus hodgsoni						53	2					24
Anthus spinoletta						55	52	63	12	17	2	2
Pycnonotus sinensis	14	24	11	14	18	36	75	9	1	4	71	62
Lanius cristatus	5	2	7	1								
Lanius sphenocercus								2				
Dicrurus macrocercus	2	7	17	3								
Sturnus cineraceus	17	127	19			17	4		2		4	5
Acridotheres cristatellus	7	10	11	9		1	4					
Cyanopica cyana	22	95	116	74	75	128	212	205	97	110	144	34
Pica pica	65	89	83	56	91	212	78	74	111	187	184	58
Corvus dauuricus						152	108	24	100	100		
Corvus macrorhynchos	2		2	1	1	4	23	7	17	13	7	2
Corvus corone		5				4	32	30	8	9	26	
Luscinia svecica					13	2						
Phoenicurus auroreus						1					1	
Saxicola torquata				1	32							
Turdus merula				2		1						
Turdus naumanni							6	13			4	
Paradoxornis webbiana			4	3		11						

Acrocephalus orientalis	3	5	2									
Acrocephalus bistrigiceps	1				6							
Acrocephalus aedon	3											
Phylloscopus inornatus					7	2						1
Phylloscopus proregulus					2	7	15					7
Cisticola juncidis			9									
Ficedula parva					8							
Parus major		2	1					1		1		
Parus palustris							8			5	39	
Passer montanus	456	611	582	479	478	410	601	397	511	364	425	256
Carduelis sinica	5	9	2				2		1	12	15	4
Eophona migratoria	2		1	1		1						
Emberiza spodocephala			2									24
Emberiza pusilla	1				13				1		4	10
Emberiza pallasi						1	3	1	1		5	10

Table A5 Dispersal distances (DD) of species

Family	Scientific name	Body mass (kg)	Modelled DD (km)	Observed DD (km)	Parameter (*p*)	If there are references (YES or NO)	
Podicipedidae	*Tachybaptus ruficollis*	0.225	0.39			NO	
Ardeidae	*Ardea cinerea*	1.750	7.50	7.50	5.27	YES	Paradis et al. 1998
	Ardeola bacchus	0.280	2.36			NO	
	Bubulcus ibis	0.440	3.14			NO	
	Ardea alba	1.025	5.35			NO	
	Egretta garzetta	0.525	3.51			NO	
	Ardea intermedia	0.775	4.49			NO	
	Nycticorax nycticorax	0.750	4.40			NO	
	Ixobrychus sinensis	0.103	1.26			NO	
	Botaurus stellaris	0.975	5.19			NO	
Anatidae	*Anser cygnoides*	3.450	11.21			NO	
	Anser fabalis	3.100	10.48			NO	
	Tadorna ferruginea	1.689	7.15			NO	
	Anas crecca	0.398	2.88			NO	
	Anas falcata	0.700	4.11			NO	
	Anas platyrhynchos	1.015	5.19	5.19	5.14	YES	Paradis et al. 1998
	Anas poecilorhyncha	1.250	5.92			NO	
	Mergus merganser	1.686	7.14			NO	
Accipitridae	*Milvus migrans lineatus*	1.160	6.15			NO	
	Accipiter nisus	0.300	2.62	2.62	5.6	YES	Marquiss and Newton, 1982 Paradis et al. 1998 Newton, 2001
	Buteo buteo	1.073	5.85			NO	

	Circus cyaneus	0.530	3.75			NO	
Falconidae	*Falco amurensis*	0.190	1.18			NO	
	Falco tinnunculus	0.335	1.69	1.69	3.37	YES	Paradis et al. 1998
Rallidae	*Amaurornis phoenicurus*	0.251	0.39			NO	
	Gallinula chloropus	0.400	0.53	0.53	0.94	YES	Paradis et al. 1998
Charadriidae	*Vanellus vanellus*	0.275	6.00	6.00	13.53	YES	
	Charadrius placidus	0.081	2.78			NO	
	Charadrius dubius	0.048	2.00			NO	
Scolopacidae	*Tringa ochropus*	0.107	1.82		7.43	YES*	Jackson, 1994
	Actitis hypoleucos	0.610	5.44			NO	
	Gallinago gallinago	0.155	2.30			NO	
Laridae	*Chlidonias leucopterus*	0.770	7.25		8.55	YES*	Paradis et al. 1998
Columbidae	*Streptopelia orientalis*	0.280	0.72		1.61	YES*	Paradis et al. 1998
	Streptopelia chinensis	0.205	0.59			NO	
Cuculodae	*Cuculus micropterus*	0.138	0.54			NO	
	Cuculus canorus	0.135	0.53	0.53	1.87	YES	Koleček et al. 2015
Apodidae	*Apus apus*	0.041	2.63	2.63	19.67	YES	Paradis et al. 1998
	Apus pacificus	0.048	2.90			NO	
Alcedinidae	*Megaceryle lugubris*	0.500	4.61			NO	
	Alcedo atthis	0.360	3.75	3.75	7.14	YES	Paradis et al. 1998
Upupidae	*Upupa epops*	0.900	1.98	1.98		YES	Bötsch et al. 2012
Picidae	*Picus canus*	0.150	0.97			NO	
	Dendrocopos major	0.072	0.61	0.61	3.2	YES	Paradis et al. 1998
	Dendrocopos canicapillus	0.030	0.35			NO	
Alaudidae	*Alauda arvensis*	0.040	0.17	0.17		YES	Paradis et al. 1998
Hirundinidae	*Hirundo rustica*	0.021	0.86	0.86	9.8	YES	Paradis et al. 1998 Shields, 1984

	Cecropis daurica	0.031	1.10			NO	
Motacillidae	*Motacilla flava*	0.021	0.72	0.72	8.21	YES	Paradis et al. 1998
	Motacilla cinerea	0.020	0.70			NO	
	Motacilla alba lugens	0.029	0.89	0.89	8.28	YES	Paradis et al. 1998
	Anthus hodgsoni	0.025	0.81			NO	
	Anthus spinoletta	0.021	0.72			NO	
Pycnonotidae	*Pycnonotus sinensis*	0.041	0.35	0.35		YES	Weir and Corlett, 2007
Laniidae	*Lanius cristatus*	0.044	0.46		3.27	YES*	Paradis et al. 1998
	Lanius sphenocercus	0.092	0.73			NO	
Dicruridae	*Dicrurus macrocercus*	0.057	1.56			NO	
Sturnidae	*Sturnus cineraceus*	0.105	0.35		1.46	YES*	Paradis et al. 1998
	Acridotheres cristatellus	0.130	0.40			NO	
Corvidae	*Cyanopica cyana*	0.100	0.14			NO	
	Pica pica	0.250	0.25	0.25	0.60	YES	Paradis et al. 1998
	Corvus dauuricus	0.285	0.62			NO	
	Corvus macrorhynchos	0.591	0.98			NO	
	Corvus corone	0.550	0.93	0.93	1.36	YES	Paradis et al. 1998
Turdidae	*Luscinia svecica*	0.018	0.40			NO	
	Phoenicurus auroreus	0.020	0.43			NO	
	Saxicola torquata	0.024	0.48			NO	
	Turdus merula	0.121	1.33	1.33	5.03	YES	Greenwood and Harvey, 1976
	Turdus naumanni	0.088	1.09			NO	
Paradoxor	*Paradoxornis webbiana*	0.012	1.10			NO	
Sylviidae	*Acrocephalus orientalis*	0.031	1.90		16.92	YES*	Paradis et al. 1998
	Acrocephalus bistrigiceps	0.009	0.87			NO	
	Acrocephalus aedon	0.028	1.78			NO	
	Phylloscopus inornatus	0.009	0.87			NO	

	Phylloscopus proregulus	0.007	0.74			NO	
	Cisticola juncidis	0.009	0.87			NO	
Muscicapidae	*Ficedula parva*	0.014	0.24	0.24		YES	Part and Gustafsson, 1989
Paridae	*Parus major*	0.017	0.25	0.25	3.26	YES	Paradis et al. 1998
	Parus palustris	0.014	0.22			NO	
Ploceidae	*Passer montanus*	0.024	0.28	0.28		YES	Paradis et al. 1998
Fringillidae	*Carduelis sinica*	0.021	0.81		9.25	YES*	Paradis et al. 1998
	Eophona migratoria	0.059	1.56			NO	
Emberizidae	*Emberiza spodocephala*	0.021	0.47		5.34	YES*	Paradis et al. 1998
	Emberiza pusilla	0.017	0.41			NO	
	Emberiza pallasi	0.015	0.38			NO	

Table A6 Correlation between landscape connectivity and biodiversity in different seasons

		Connet_1	**Connet_2**	**Connet_3**	**Connet_4**	**Connet_5**
BA	Spr.	0.42	.565*	.569*	.572*	.573*
	Sum.	.564*	.699**	.708**	.712**	.715**
	Atm.	0.15	0.229	0.233	0.236	0.238
	Win.	0.128	0.237	0.245	0.249	0.252
	Year	0.338	0.464	.471*	.475*	.477*
BR	Spr.	0.38	.525*	.530*	.532*	.534*
	Sum.	0.444	.612**	.620**	.624**	.626**
	Atm.	0.21	0.369	0.375	0.379	0.382
	Win.	0.239	0.384	0.389	0.391	0.392
	Year	0.354	.520*	.526*	.530*	.532*
BD	Spr.	0.455	.564*	.568*	.570*	.571*
	Sum.	0.163	0.301	0.307	0.311	0.313
	Atm.	0.268	0.419	0.425	0.429	0.431
	Win.	0.162	0.201	0.202	0.202	0.202
	Year	0.296	0.433	0.438	0.441	0.443

Note: 1) ** Correlation is significant et the 0.01 level (2-tailed); * Correlation is significant et the 0.05 level (2-tailed). 2) BA= bird abundance; BR= bird richness; BD= bird diversity. 3) Connect_1= landscape connectivity with the distance threshold of 1000 m; similarly, Connecti_2= landscape connectivity with the distance threshold of 2000 m. Connecti_3= landscape connectivity with the distance threshold of 3000 m. Connecti_4= landscape connectivity with the distance threshold of 4000 m. Connecti_5= landscape connectivity with the distance threshold of 5000 m.