Plant biodiversity and ecosystem services in an Asian developed city: land use and scale

（アジアの都市における植物の生物多様性と生態系サービス：土地利用とスケール）

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Abstract

Urbanization provides both challenges and opportunities in biodiversity conservation. The demand for urban ecosystem services increases with the rapid growth of the urban population. Urban dwellers can benefit from within-urban biodiversity and local ecosystem services. However, the pattern of urban biodiversity across land use and scales, especially in the cities of Asian developed countries, remains unclear. The evaluation of urban ecosystem services across urban heterogeneity has also been less addressed. In this study, a multi-scale analysis of urban plant diversity and ecosystem services was conducted with a ground-based investigation and model. To collect data, woody plants in 174 sample quadrats in Kyoto City were investigated.

The results of urban biodiversity analysis show that, at the land use level, residential areas were found to have the highest total richness with moderate to low evenness, while commercial areas exhibited low total richness. At the quadrat level, low-rise residential area had higher richness than most of the other land use types. Quadrat abundance and evenness were significantly different across land use types for trees but not for shrubs. Quadrat species composition was significantly different across land use types for shrubs, but not for trees. The driving factors for quadrat biodiversity were also analyzed.

The ecosystem services calculated include carbon storage and sequestration, air pollutants removal, and runoff reduction in the following chapter. Ecosystem services of different land use were compared at both quadrat and single-tree levels. No significant difference across land use for any of the ecosystem services was found at the quadrat level. However, the ecosystem services were different across land use at the single-tree level. With a species-specific analysis, it was revealed that the pattern of ecosystem services across land use varies with both the service tested and species.

The results of the study provide insights into urban biodiversity and ecosystem services design and management by identifying prior land use types for biodiversity improvement and highlighting the contribution of residential private yards. Furthermore, the results suggest that urban heterogeneity, scale, and multidimensionality should be considered when measuring urban biodiversity and ecosystem services.

Then an analysis was conducted to test the link between urban biodiversity, environment, and urban ecosystem services. The results show that quadrat biomass, as a surrogate of urban biodiversity, is best predicted by land cover proportion and biodiversity-related indexes. Among the biodiversity-related indexes, abundance is a better predictor than the others like richness and the Shannon index. Further development of biodiversity-ecosystem service link analysis tool is needed to fill the critical gap for the evidence-based decision in urban planning.

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Chapter 1 General Introduction

1.1 Background

1.1.1 Urbanization

Past decades have witnessed rapid urbanization worldwide. The urban population has been growing rapidly since 1950. Estimated with statistics data reported by national governments, the proportion of the urban population increased from 30% in 1950 (Population Division, United Nations, 2019) to 55% in 2019 (The World Bank Data, no date), and it is projected to be 68% in 2050 (Population Division, United Nations, 2019).

The increase of the urban population, together with economic development, resulted in the expansion of the cities. The expansion of urban areas is even faster than that of the urban population. A meta-analysis shows that urban area increases twice as fast as its population on average worldwide (Seto et al., 2011). The urban area is estimated at 0.35 million square kilometers in 1992 and increased to 0.74 million square kilometers in 2015 (McDonald et al., 2020), and it is projected to be 1.9 million square kilometers in 2030 (Güneralp and Seto, 2013; Seto et al., 2012). Furthermore, most urban area expansion in the future is projected to be in areas of limited economic development (Seto et al., 2013). Though the expansion rate of the urban area depends on the definition of 'urban area' or 'urban land use", a consensus is that in recent years, the urban area expansion has been fast and the trend will continue.

1.1.2 Impact of urbanization on biodiversity

Rapid urbanization has a significant environmental and ecological impact, one of which is biodiversity degradation (Elmqvist et al., 2013; Güneralp and Seto, 2013; Seto et al., 2012). Urbanization causes species loss in two ways: the indirect and the direct.

The indirect impact is caused by resources consumption of urban areas and pollution. Resource consumption rises with urbanization, that urban area requires loads of imported energy, food, and material in its 'upstream' and results in pollution in its 'downstream'. Urbanization changes climate by heat island effect and modifying precipitation patterns, which together has impacts on biodiversity (Seto et al., 2013). A case study is that Girgin et al. (2010) evaluated the impact of water heavy metal pollutants on insects in Ankara stream, Turkey. Urbanization also impacts the biodiversity of other areas (especially surrounding rural areas) through human migration. Community-scale research in Mexico suggests that the migration of population to a city may cause a decline of biodiversity in forest and agricultural land in the surrounding area due to the decrease of human management (Robson and Berkes, 2011). The scale of the impact varies from regional to global. The food production process for a city could cause degradation of biodiversity in the place of food source (Selinske et al., 2020).

Direct impact generally occurs with habitat change due to urbanization. Urbanization causes land use change/habitat transformation, that direct change of habitat extinct the original endemic species (McDonald et al., 2020). Furthermore, since urbanization preferred the location with higher biodiversity, it caused degradation of global biodiversity conservation (Seto et al., 2013). The problem could be even more serious since urbanization will be rapid in the area with low economic development that has limited investment capability in biodiversity conservation and mitigation (Seto et al., 2013). However, on the other hand, urbanization not only causes biodiversity loss but also biodiversity gain. Some taxon, like flora or birds, has been frequently proved to be of higher richness in urban areas than in surrounding nature or rural areas (Knapp, 2010a). Possible reasons include the increase of heterogeneity, edge effect, and human introduction of exotic species (Hope et al., 2003).

1.1.3 Impact of urbanization on ecosystem services

Ecosystem service is defined as the service provided by ecosystems that are essential to human survival, livelihood, health, and well-being (Costanza et al., 2014). Urbanization decreases ecosystem service by land use change, habitat fragmentation, and pollution.

With rapid urbanization, land use shifts from nature or agriculture to urban land. That shifting results into a change of process and pattern of biogeochemical circulation, and thus a decrease of ecosystem services. Furthermore, ecosystem service is affected by many other factors, for instance, land quality, water quality, and biodiversity. A case study in Shenzhen city, China, finds that ecosystem of the city decreased with the expansion of construction land use (Peng et al., 2015). A study of Bengal, India shows that ecosystem service value has been decreased by 25% from 1990 to 2017 (Das and Das, 2019). Another study at eco-region scale shows ecosystem services decrease with expanding urban area in South China (Su et al., 2012).

1.2 Research gaps

1.2.1 Importance of urban biodiversity and ecosystem services

A city is not only unique in its physical environment and socio-economic dimension, but also in its biological dimension. It is thus regarded as a novel ecosystem. Though urbanization degrades nature habitat, it also creates novel habitat (e.g., parks, yards, street trees, nature remnant. It has been proved that the biological diversity of some taxon (e.g., woody plant and birds) are even higher than the surrounding nature area (Knapp, 2010a; Pearse et al., 2018).

The unique community assemblage in cities results in a complex and high heterogeneous biodiversity pattern. From a perspective of process, William et al. (2009) proposed a conceptual framework for the effect of urbanization on biodiversity based on the environmental filter model, taking both biodiversity loss and gain into consideration. In their framework, urbanization affects biodiversity with four filters: habitat transfer, habitat fragmentation, environmental pressure of urbanization, and resident preference. Among the filters, environmental pressure usually refers to physical environmental factors, while resident preference is driven by socio-economic factors (Chamberlain et al., 2019; Chen et al., 2020; Hope et al., 2003; Kajihara et al., 2016). Socio-economic factor is a strong, and unique driving force for urban woody plant diversity in cities. The luxury effect is one example of the impacts of socio-economic factors, which states that wealthier neighbors generally have higher biodiversity (Chamberlain et al., 2020; Clarke et al., 2013; Leong et al., 2018).

Those new habitats, together with the biological community, also provide local ecosystem services that are essential to urban residents. Six important services for an urban area was identified, including air filtering, micro-climate regulation, noise reduction, water regulation, waste treatment, and recreation and culture services (Bolund and Hunhammar, 1999). The “key services” might differ across different cities, determined by environmental and socio-economic characteristics. For instance, air pollutants removal could be secondary important in cities with good air quality but is critical for cities with serious air pollution situations and more vulnerable people (Gómez-Baggethun et al., 2013). The relation between demand and supply of ecosystem services also varies with scale. Locally generated ecosystem services are more closely related to the living quality of the resident and some of them is irreplaceable by other distant sources (for example, mitigation of heat island effect) (Gómez-Baggethun et al., 2013). Considering the numerous population size in cities, the social and economic value of ecosystem services within cities can be surprisingly high (Gómez-Baggethun and Barton, 2013). Besides, a global assessment highlighted how massive urbanization is impacting ecosystems around the world negatively (Elmqvist et al., 2013). Therefore, an improvement of urban ecosystem services could potentially benefit city residents and mitigate the loss of ecosystem services globally.

With the growth of urban population and area, cities are becoming the place where people get in touch with nature in their daily life. Studying urban biodiversity is necessary not only for biological conservation, but also for human well beings. However, compared to other ecosystems, the biodiversity research in cities is lacking.

1.2.2 Research gaps

Despite the importance of the recognition of urban biodiversity and urban ecosystem services, the application of the concept and studies into practice is still lacking. One of the reasons is that among the factors studied, land use has been less addressed in previous research, though land use is the main pathway that humans modify cities. An integration of land use and urban ecology can enhance the application of urban ecology research. The detailed research gaps are identified in the following chapters.

1.2.3 Thesis outline

The main objectives of this research are: to study the urban biodiversity across land use; to study ecosystem services across land use; to link environmental factors, biodiversity, and ecosystem services in urban ecosystem services.

The thesis is organized into 6 chapters (Figure 1). Chapter 1 introduces the background and the impact of urbanization on nature, and the reasons why we should care about biodiversity and ecosystem services within cities. I also introduced the general idea of research gaps and research objectives. Chapter 2 introduces data collection and how the data set was processed. Chapter 3 focuses on urban biodiversity, taking woody plants as the target taxa. Chapters 4 focuses on urban ecosystem services provided by an urban forest, and analyzed ecosystem services across land use. In chapter 5, I linked urban environment, biodiversity, and urban ecosystem services. In chapter 6, I made a discussion and conclusion.

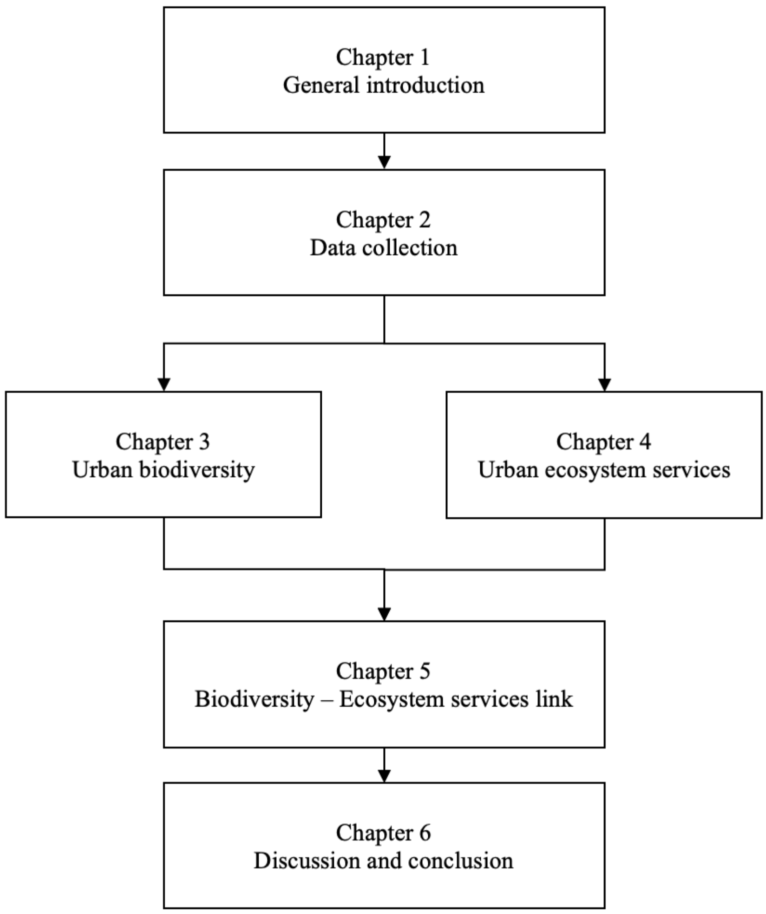

Figure 1. Research scheme for this thesis

Chapter 2 Data Collection and Preprocessing

2.1 Research area

Kyoto City (35°19′16″N-34°52′30″N, 135°33′33″E-135°52′43″E), the capital of Kyoto Prefecture, is located in Kyoto Basin of Kansai Region, Honshu Island, Japan, with an area of 828 square kilometers. The city is dominated by a humid subtropical climate with hot, humid summers, and cold, dry winters. The annual high and low temperatures are 20.8°C and 11.7°C, respectively, with an average precipitation of 1,491 mm.

Kyoto had experienced rapid urbanization before 1980s, as with many cities in Japan; its population has remained stable since then. It is one of "Cities designated by government ordinance of Japan" with a population of 1.47 million (0.73 million households) in 2019. The population density is 1,773 people per square kilometer. The area of the built-up area of the city is 144 square kilometers (Kyoto City Statistics Portal, 2019).

As a planned capital, Kyoto city was founded when Emperor Kammu relocated the capital in 794. The Japanese borrowed the basic city layout from Chang'an, China in the Tang-dynasty, part of which is a grid spatial system dividing the city into blocks. The land use pattern of the city, however, mainly formed in modern times. During the infrastructure promotion at the end of the Meiji era (1868-1912), city center was constructed based on the traditional commercial area. The administration boundary expanded significantly in the Taisho era (1912-1926), and an expansion and construction of the industrial area and residential area was achieved based on the urban planning laws (Ueno, 2010). Kyoto city is now, overall, a mono-centric city. The city center is mainly used as a commercial area. The industrial area is mostly located in the west and south of the city. The residential area is in the surrounding area (Figure 2).

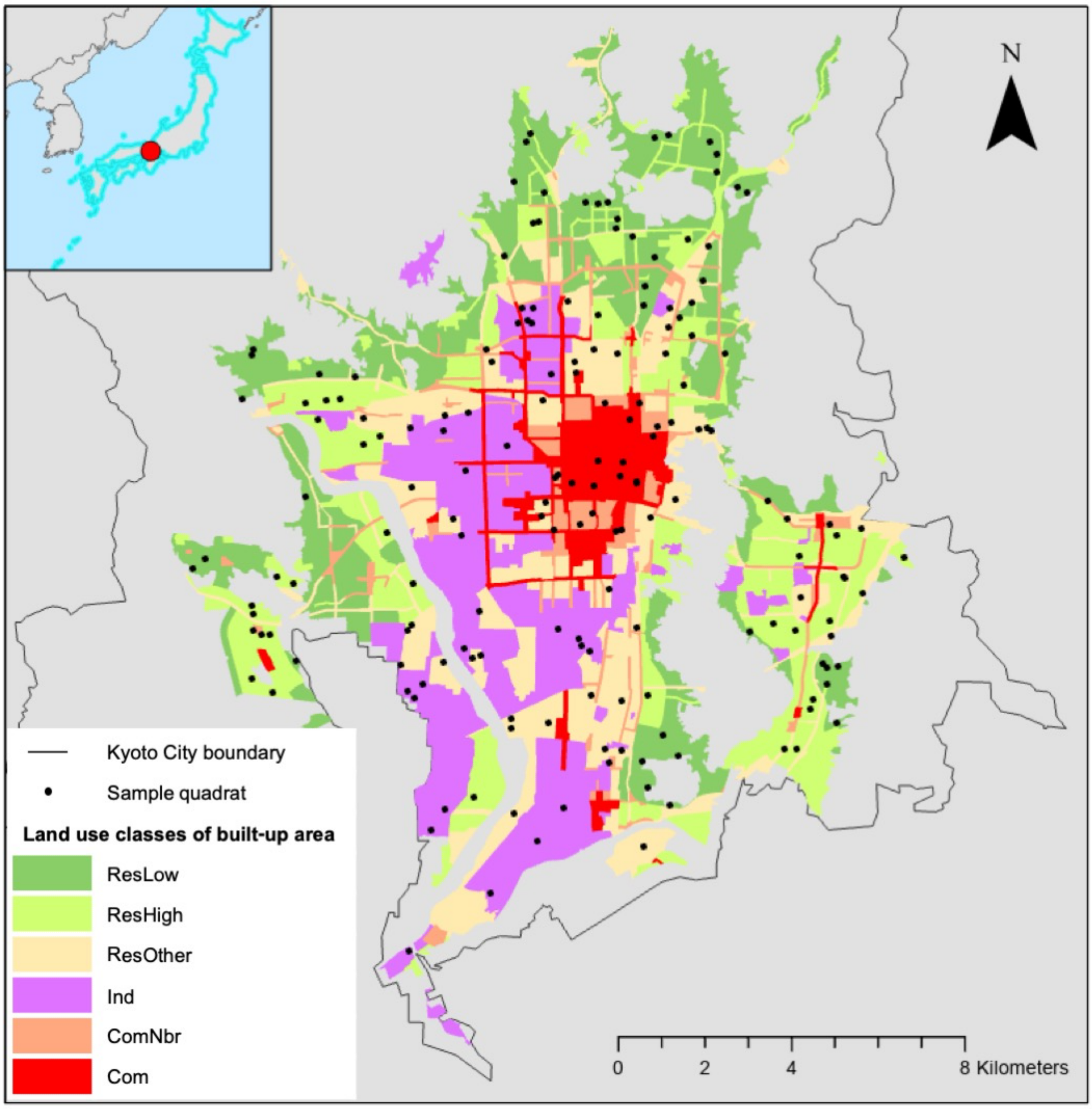

Figure 2. Land use of Kyoto City and distribution of investigated sample quadrats.
The abbreviations for land use: ResLow, low-rise residential area; ResHigh, mid- and high-rise residential area; ResOther, other residential area; Ind, industrial area; ComNbr, neighborhood commercial area; Com, commercial area.

2.2 Land use classification

Urban planning system varies across countries due to different background of law, regulation, management and history; so does the concept of "land use" (Briassoulis, 2019). In this study, "land use" emphasizes more on the potential usage of human activities. Japan's urban planning system sets a framework for urban land use zoning, city infrastructure, and implementation of development projects under the *City Planning Law of Japan* (Ministry of Land, Infrastructure, Transport and Tourism, Government of Japan, 2021). "Land use zone" is the core and fundamental base of the system that categorizes land use of the built-up area into

3 main types (residential, commercial, and industrial) or 12 sub-types (Table 1). For each land use, a series of rules on the use, height, density and other attributes of buildings are set (Akashi, 2007). These 12 sub-types of land use were aggregated into 6 types in this study: Com (Commercial) area, ComNbr (Neighborhood commercial) area, ResLow (Low-rise residential) area, ResHigh (Mid/high-rise residential) area, ResOther (Other residential) area, and Ind (Industrial) area (Table 1). Among commercial areas, Com area is predominated by commercial and business buildings such as banks, cinemas, restaurants and department stores, though residential building and small factories are also permitted; ComNbr area is mainly designated to provide daily shopping services for residents. Ind area is mainly intended for factories, though residential and shop buildings are also permitted. Among residential areas, ResLow is mainly designated for low rise residential buildings, and only small shops/offices and elementary/junior high school buildings are permitted other than residential buildings; ResHigh area is designated for medium to high rise residential buildings, hospitals, universities, and shops/offices with a floor area under 1,500 m^2; ResOther area is designated as buffer zone for residential area, in which houses, shops, offices, and hotels with a floor area up to 3,000 m^2 are permitted.

Table 1. Land classification and the number of investigated quadrats in this study.

Land use type	Land use sub-type	Quadrat number	Quadrat proportion (%)
Com	Commercial area	14	8.0
ComNbr	Neighborhood commercial area	10	5.7
ResLow	Category I exclusively low-rise residential area Category II exclusively low-rise residential area	38	21.8
ResHigh	Category I mid/high-rise oriented residential area Category II mid/high-rise oriented residential area	41	23.6
ResOther	Category I residential area Category II residential area Quasi-residential area	42	24.1
Ind	Quasi-industrial area Industrial area Exclusively industrial area	29	16.7

2.3 Sampling and data collection

2.3.1 Plant data

In the field investigation conducted between May and August 2019, plant data was collected in a potential sample size of 200 quadrats (size: 20 m × 20 m), including alternative quadrats considering some quadrats may be inaccessible. The number of quadrats in each land use type were determined with stratified sampling method based on areas of land use types (Nowak et al., 2008) and a total of 174 quadrats were actually accessed and investigated (Figure 2, Table 1).

For each quadrat, I took photos of the surrounding environment and the vegetation. All woody plants higher than 30 centimeters in the quadrats were investigated. Plants higher than 2 meters were recorded as trees and others as shrubs (Chimaimba et al., 2020; Wang et al., 2012; Yang et al., 2017; Zhang et al., 2020). Data collected for each plant included: (1) species: only refers to original species, thus cultivars, subspecies, and varieties were not considered; information of scientific names and taxon followed the online database of plant names, *The Plant List*, as per Royal Botanic Gardens, Kew, and the Missouri Botanical Garden (Kalwij, 2012); (2) size-related data for trees: height, diameter at breast height (DBH), crown size; (3) size-related data for shrubs: height and crown area; (4) health status: canopy missing percentage, crown health condition, and crown light exposure; (5) growth status: planted or spontaneous; and (6) ownership: public or private. For the last attribute, “ownership”, since accessibility is emphasized in this research, some semi-public spaces like temple forest are classified as “public” because they are generally accessible to the public (Table 2).

Table 2. Category of urban green by ownership.

Ownership	Examples
Public	Temple forest and shrine forest, public garden, imperial palace, street tree, rain garden, park, river bank
Private	Traditional Japanese townhouse yard and other single-family house yard, green roof

Species provenance was determined according to the Introduced Plant Species List of Kyoto Prefecture (Kyoto Prefecture Web Site), the Seed Plant Species Dataset of Kyoto Prefecture (Kyoto Prefecture Web Site), and the Introduced Species List of Japan published by the National Institute of Environmental Studies of Japan. Native distribution information from the Missouri Botanical Garden database (Missouri Botanical Garden) and the Kew Science database (Kew Science) was also referenced for species not in the Japan database. The details of species provenance are shown in Table A1.

The data set, with several attributes of each plant, will then be applied in the following chapters with different topics. The species, number of trees and crown area of shrubs, grown status, ownership, and provenance will be used in urban biodiversity analysis. The whole data set other than ownership and provenance will be applied into urban ecosystem services research. The results of urban biodiversity and ecosystem services research will be further applied to the study of the link between biodiversity and ecosystem services.

2.3.2 Quadrat attribute data

The attribute data of accessed and investigated sampling quadrats were collected since they can act as impact factors for urban biodiversity and urban ecosystem services at the quadrat level. (1) The distance to the city center: The latitude and longitude of each quadrat were collected based on GoogleMap. Since Kyoto City is regarded as a monocentric city, I took Shijo-Kawaramachi station, located in the central business district, as the city center. Then *distm* function of *geosphere* package was applied to calculate the distance of each quadrat to the city center. (2) Land price: land price investigation point data of Kyoto Prefecture in 2019 was downloaded from the website of Ministry of Land, Infrastructure, Transport and Tourism (Ministry of Land, Infrastructure, Transport and Tourism, no date). The point data was then extended into a raster data covering the whole Kyoto City by Kriging interpolation in ArcGIS pro. Then the land price of the quadrats was equal to the land price of the corresponding pixel. (3) Population density: Ministry of Public Management, Home Affairs, Posts and Telecommunications of Japan published the demographic census data at neighborhood scale of 2010, then the spatial data was further scaled into 100-meter resolution mainly according to building type and area (Nishizawa, 2016). The population density of each quadrat was based on the 100-meter demographic mesh data. (4) Onsite land cover: The detailed land cover was mapped for each quadrat during the investigation. A quadrat was divided into different land covers including residential, transportation, temple/shrine, multi-family residential, agriculture, commercial-neighbor, water/wetland, park, cemetery, vacant land, commercial/industrial, institutional, and golf course. The proportion of each land cover in each quadrat was further calculated.

Chapter 3 Urban Plant Biodiversity

3.1 Introduction

Urbanization is occurring fast in the area of previously highly productive ecosystems (Marc L. Imhoff et al., 2004) and the area adjacent to biodiversity hotspots (Seto et al., 2013), thus causing biodiversity decline due to habitat loss and fragmentation (Seto et al., 2012). However, cities also provide new opportunities for biodiversity conservation. For instance, although the response of biodiversity to urbanization is unpredictable (McDonnell and Hahs, 2008), many researchers have found higher richness or abundance in some taxa such as birds and flora in urban and suburban areas, partially due to a considerable number of introduced species and habitat heterogeneity (Allen and O'Connor, 2000; Dangulla et al., 2019; Hansen et al., 2005). Cities may even provide shelter to rare species (Ives et al., 2016). Among related taxa, plants are viewed as a template for biodiversity (Ackleh et al., 2010; Brearley et al., 2010; Brown and Freitas, 2002; Dearborn and Kark, 2010; MacGregor-Fors, 2008), and are directly modified by anthropic activity. It is therefore essential to understand the relationship between human activity and plant diversity in urban ecosystems. It has also been recognized in recent years that cities play an important role in global biodiversity targets since urban residents have become more aware of biodiversity conservation needs and local authorities are more effective in managing surrounding biodiversity (Convention on Biological Diversity, 2010).

The past three decades have witnessed a significant growth of research interest in the urban plant diversity field. However, most such work has focused on European, Australian and Northern American cities, and less on cities in Central and Southern America, Africa and Asia (Ossola et al., 2018). Many urban plant diversity studies (Chimaimba et al., 2020; Raoufou et al., 2011; Wang et al., 2015, 2017) have also focused on rapidly urbanized areas rather than highly urbanized ones.

The dispersed green spaces, like private gardens, have been addressed less though they are proven essential to urban biodiversity and ecosystem services (Camps-Calvet et al., 2016). For urban biodiversity research in Japan, Tsuchiya and Saito (2018) reviewed 173 English- and Japanese-language articles published between 1975 and 2015, returning 71 articles on urban plants. I recorded the 71 articles and found that a total of 37 of these focused on spermatophytes, but most of them focused on large green spaces like urban parks and forest remnants (e.g., Okamura et al. (1998)'s research of Nagoya's urban parks).

Land use/land cover is driven by, and in turn, has impact on bio-physical and socio-economic factors; it is the results of, and constrains human activities (Briassoulis, 2019). However, the heterogeneity of urban plant diversity across land use/land cover has been less addressed. Since plantable space, greening goals, and human-related disturbances vary with land use/land cover (Bourne and Conway, 2014), land use/land cover has also been regarded as an urbanization gradient factor in relevant biodiversity research (Ortega-Álvarez et al., 2011; Porter et al., 2001). However, most studies have focused on single land use/land cover types (Bourne and Conway, 2014) or have not distinguished between different land use/land cover types (residential areas, commercial areas, etc.) in urban built-up areas (Dangulla et al., 2019).

Urban-rural gradient has also been widely used for urban plant diversity research (McDonnell and Hahs, 2008), while the results differ across different cities and taxa. For instance, research in Shanghai, China, indicates that the richness of woody plants, annual herbs, and perennial herbs shows a different pattern with the increasing distance to the city center(Wang et al., 2020). And for woody plants, Wang et al. (2020) found that the richness decreases along the urban-rural gradient in Shanghai, while Aronson et al. (2015) found no significant difference of the richness along the urbanization gradient in New York. While it is common that the diversity of native and exotic plants responds differently to the urbanization gradient. For instance, the richness of native species generally increases with the decrease of urbanization, while the richness of exotic species, by contraries, is usually higher in more urbanized areas (Aronson et al., 2015; Ranta and Viljanen, 2011). A point of dispute is that there are multiple gradients in a city, and some of them may conflict with another. Thus, a multi-gradient analysis is required.

In this chapter, I applied investigation data of woody plants across Kyoto City to reveal the urban biodiversity across land use at different scales. The data was aggregated at three levels: city, land use type, and quadrat. Analysis was conducted at these three levels to demonstrate the heterogeneity of urban plant diversity across land use and scales by comparing the richness, evenness, abundance, and species composition.

3.2 Data analysis

The species, number of trees and crown area of shrubs, grown status, ownership, provenance of the individual plant, the onsite land cover proportion, and urban gradient variables of each quadrat were extracted from the data set described in Chapter 2. The data for each tree and shrub was then aggregated into city level, land use level, and quadrat level. All data analysis was conducted in *R (version 4.0.3)*. Difference was considered significant as $p < 0.05$ where statistical comparison was applied. The code for data analysis is accessible on GitHub (https://github.com/kangjf1943/DoThesis).

At city level, *iNEXT* function of *iNEXT* package was applied to estimate the total species for Kyoto City using a rarefaction and extrapolation method (Chao et al., 2014; Hsieh et al., 2020). To evaluate evenness of species distribution, rank abundance curves was plotted from calculation results of the *rankabundance* function of the *BiodiversityR* package for trees and shrubs respectively, in which species abundance was presented using proportional abundance, and species rank is standardized by dividing the total richness. An evenness index, E_Q, as proposed by Smith and Wilson (1996) in relation to the rank abundance curve, was also calculated.

Similarly, at land use level, total species was estimated, and rank abundance curves were plotted for each land use type. The results were compared across land use types directly. To quantify the difference of species composition, Bray-Curtis dissimilarity was calculated across land use in pairs. Furthermore, to explore preferred-species choice, the occupancy rate was used to test ubiquitous species of each land use type using $OR_{ij} = N_{ij}/N_j$, where N_{ij} is the number of quadrats with the presence of species i in land use type j and N_j is the number of quadrats of land use type j.

At quadrat level, though there are many alternative biodiversity metrics, I only use species richness and evenness for final analysis. I did a pretest of the correlation between a series of metrics and found that richness is a good surrogate though not correlated with evenness (Table A2). Richness was the number of species in the quadrat. Evenness was estimated with Pielou's evenness index:

$$J = -\sum_{i=1}^{S} p_i \ln p_i \,/\, \ln(S)$$

where p_i is the quadrat's proportional number of trees or the proportional area of shrubs of the constituent species i, and S is the quadrat richness. As not all quadrat indexes follow normal distribution, Kruskal-Wallis rank sum test was applied to compare the richness, abundance, and evenness index of quadrats across land use types. For statistical group comparison where significant difference was detected, Dunn's test was used for post hoc pairwise comparison with the *dunn.test* package (Alexis, 2017). The quadrat species composition discrepancy between different land use types was compared using the *anosim* function of *vegan* package. With this method, inter-quadrat species composition dissimilarity was determined from Bray-Curtis dissimilarity. To avoid bias, only the quadrats with > 1 tree or > 5 square meters shrubs were analyzed. Nonmetric multidimensional scaling (nMDS) analysis was used to visualize the quadrat species composition dissimilarity with the *metaMDS* function of the *vegan* package.

To examine the impact of urban gradient and onsite land cover, Pearson correlations were performed to test the relationships between the variables of quadrats with species richness, abundance, and evenness within all plant dataset, tree dataset, and shrub dataset. The function *cor.test* was applied for the analysis. Then multiple linear regression model was applied to test the association of the urban gradient variables and onsite land cover proportion variables with the biodiversity metrics. The *regsubset* function of *leap* package was applied to perform the analysis.

3.3 Results

3.3.1 City level results

A total of 223 species of 157 genera and 77 families were recorded (Table A1). The most common plant family was Rosaceae (24 species), followed by Oleaceae (12 species), Cupressaceae (8 species), Ericaceae (7 species) and Leguminosae (7 species). Estimated total number of species for the whole city was 265 (Figure 3).

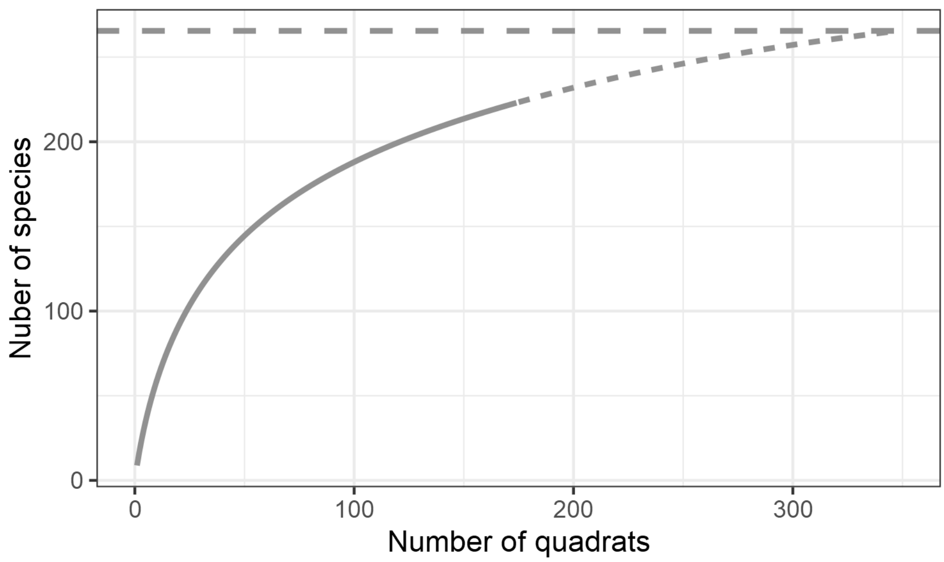

Figure 3. Sample-size-based rarefaction (solid line) and extrapolation (dotted line, up to double the investigated sample size) of number of species at city level.

The dashed horizontal line represents estimated total number of species with the extrapolation.

From the perspective of abundance, 1,240 trees and 1,233 m^2 of shrubs was recorded, with the most common families (in terms of individual numbers for trees and areas for shrubs) listed in Table 3. The most common five species for trees were *Quercus glauca* Thunb. (8.71%), *Nandina domestica* Thunb. (7.58%), *Osmanthus fragrans* Lour. (6.05%), *Acer palmatum* Thunb. (4.35%) and *Ligustrum lucidum* W.T.Aiton (4.27%), all of which belong to the top 10 tree families. The most common five species for shrubs were the *Rhododendron hirado* group (11.30%), *Nandina domestica* Thunb. (8.85%), *Rhododendron indicum* (L.) Sweet (8.21%), *Camellia sasanqua* Thunb. (4.18%) and *Photinia glabra* (Thunb.) Maxim. (3.62%), all of which belong to the top 10 shrub families. The distribution of the species was represented by rank abundance curves for trees and shrubs (Figure 4). The evenness of the trees (E_Q = 0.16) was higher than that of the shrubs (E_Q = 0.11).

Table 3. Top 10 families by abundance at city level. The abundance is represented by individual number for trees and area for shrubs.

Family	Abundance	Proportion (%)
Tree		
Oleaceae	191	15.40
Fagaceae	151	12.20
Rosaceae	98	7.90
Berberidaceae	95	7.66
Cupressaceae	74	5.96
Sapindaceae	67	5.40
Theaceae	66	5.32
Pinaceae	44	3.55
Cornaceae	39	3.14
Lauraceae	34	2.74
Shrub		
Ericaceae	262.00	21.30
Rosaceae	157.00	12.70
Berberidaceae	128.00	10.40
Theaceae	88.10	7.16
Oleaceae	62.00	5.03
Hydrangeaceae	48.00	3.90
Caprifoliaceae	42.20	3.42
Araliaceae	28.80	2.34
Garryaceae	28.50	2.32
Fagaceae	26.40	2.14

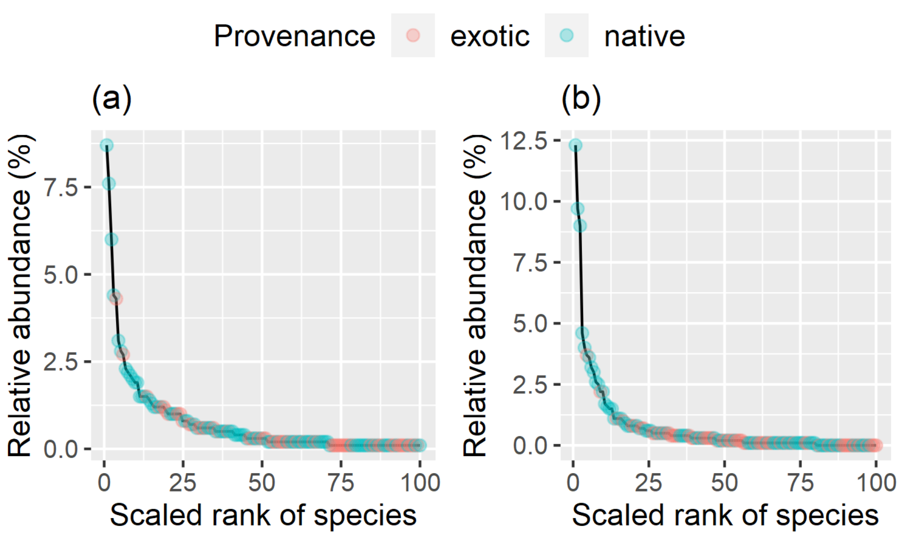

Figure 4. Rank abundance curves of (a) trees and (b) shrubs at city level, with the species rank being divided by observed richness.

Regarding plant attributes, 93% of the 1,240 tree individuals and 89% of the 1,233 m^2 of shrubs were planted rather than spontaneous. 72% of trees and 62% of shrubs were privately owned. In terms of provenance, 74% of tree individuals and 76% of shrubs belong to native species; though from the perspective of number of species, only 50% of the 223 species were native.

3.3.2 Land use level results

ResLow area had the highest estimated total richness, followed by ResOther area, ResHigh area, Ind area, Com area, and ComNbr area (Figure 5).

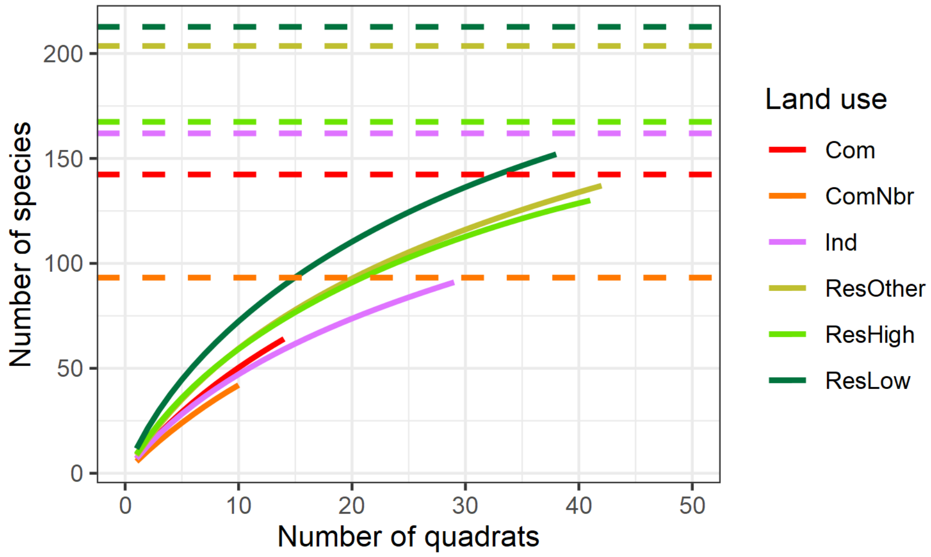

Figure 5. Sample-size-based rarefaction curves (solid lines) and estimated total number of species with extrapolation (dashed horizontal lines, up to 348 quadrats) for land use types.

The steep trend of rank abundance curves (Figure 6) shows the variation of distribution in species abundance across land use types. Further E_Q index calculation indicated that, for trees, Com area had the highest overall tree evenness ($E_Q = 0.31$), followed by ResOther area ($E_Q = 0.25$), Ind area ($E_Q = 0.24$), ResHigh area ($E_Q = 0.21$), ResLow area ($E_Q = 0.20$), and ComNbr area ($E_Q = 0.19$). In contrast, for shrubs, ComNbr area had the highest overall evenness ($E_Q = 0.15$), followed by Ind ($E_Q = 0.14$), ResLow ($E_Q = 0.14$), ResOther ($E_Q = 0.13$), Com ($E_Q = 0.12$), and ResHigh ($E_Q = 0.12$) areas.

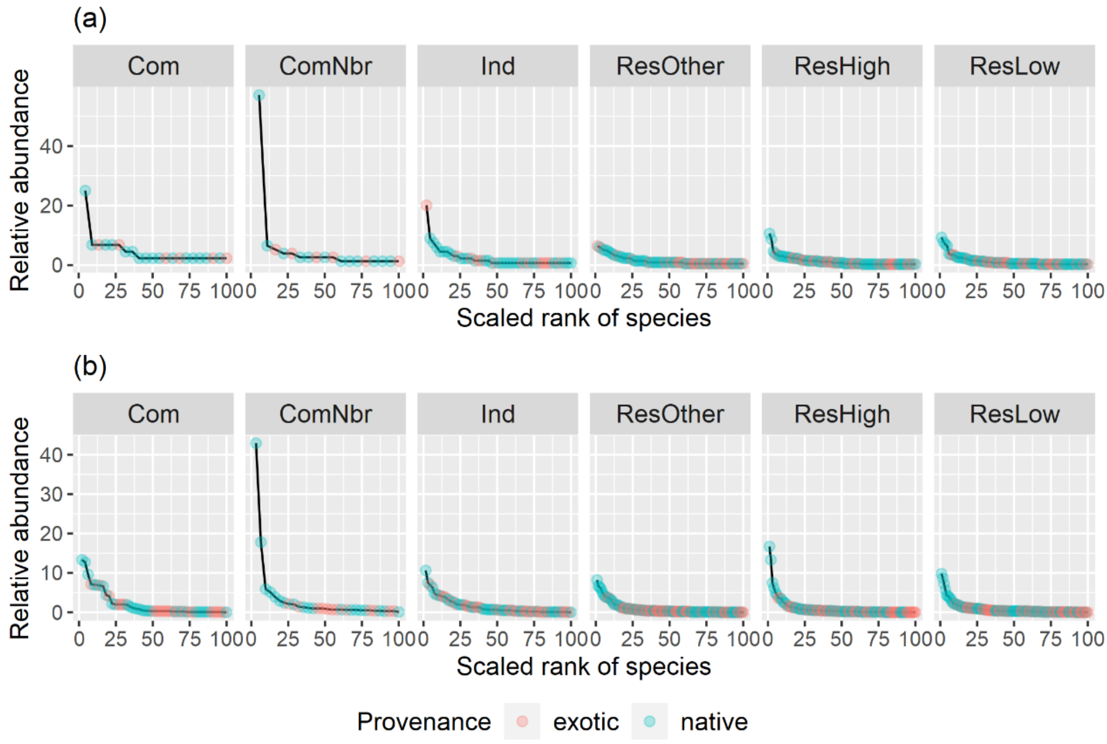

Figure 6. Rank abundance curves of (a) trees and (b) shrubs by land use types, with the species rank being divided by observed richness of the land use type.

The results of pairwise Bray-Curtis dissimilarity (Table 4) suggested a high dissimilarity (> 0.8) between ResLow area and the other land use excepting for ResOther area, and between ResHigh area and Com area for trees; while a high dissimilarity (> 0.8) between ResHigh area and Com area, ComNbr area, and Ind area for shrubs. The top 10 ubiquitous species for each land use type are listed in Table 5 to demonstrate the details of species composition.

Table 4. Dissimilarity (Bray-Curtis index) of species composition between land use types at land use level.

	Com	ComNbr	Ind	ResOther	ResHigh	ResLow
Trees						
Com	-					
ComNbr	0.52	-				
Ind	0.54	0.50	-			
ResOther	0.72	0.67	0.50	-		
ResHigh	0.89	0.74	0.78	0.72	-	
ResLow	0.89	0.83	0.84	0.74	0.82	-
Shrubs						
Com	-					
ComNbr	0.44	-				
Ind	0.49	0.55	-			
ResOther	0.66	0.67	0.60	-		
ResHigh	0.82	0.85	0.80	0.72	-	
ResLow	0.62	0.68	0.54	0.49	0.70	-

Table 5. Top 10 ubiquitous species for all the plants by occurency of number of quadrats.

Com	**ComNbr**	**ResLow**
Nandina domestica	*Nandina domestica*	*Nandina domestica*
Gardenia jasminoides	*Osmanthus fragrans*	*Osmanthus fragrans*
Mahonia japonica	*Ginkgo biloba*	*Hydrangea macrophylla*
Camellia japonica	*Cornus florida*	*Acer palmatum*
Quercus glauca	*Hibiscus syriacus*	*Camellia japonica*
Rhododendron indicum	*Acer palmatum*	*Camellia sasanqua*
Schefflera heptaphylla	*Ternstroemia gymnanthera*	*Rhododendron indicum*
Quercus myrsinifolia	*Podocarpus macrophyllus*	*Rhododendron hirado group*
Pieris japonica	*Cornus kousa*	*Cornus kousa*
Ginkgo biloba	*Pyracantha coccinea*	*Celtis sinensis*

continuous table:

ResHigh	**ResOther**	**Ind**
Nandina domestica	*Nandina domestica*	*Nandina domestica*
Hydrangea macrophylla	*Ligustrum lucidum*	*Rhododendron indicum*
Rhododendron indicum	*Rhododendron indicum*	*Acer palmatum*
Rhododendron hirado group	*Camellia sasanqua*	*Hydrangea macrophylla*
Osmanthus fragrans	*Osmanthus fragrans*	*Rhododendron hirado group*
Camellia sasanqua	*Celtis sinensis*	*Cinnamomum camphora*
Acer palmatum	*Aphananthe aspera*	*Camellia japonica*
Cinnamomum camphora	*Camellia japonica*	*Celtis sinensis*
Camellia japonica	*Hydrangea macrophylla*	*Prunus x yedoensis*
Zelkova serrata	*Rhododendron hirado group*	*Rhaphiolepis indica*

3.3.3 Quadrat level results

The results of quadrat richness comparison with Kruskal-Wallis rank sum test and post hoc Dunn's test indicated that quadrats of ResLow area had higher number of species than those of the other land use excepting for ResHigh area (Figure 7). Quadrat abundance and evenness were compared across land use types for trees and shrubs respectively (Figure 8). Kruskal-Wallis rank sum test indicated significant difference in abundance and evenness for trees while no difference for shrubs. Post hoc pairwise comparison suggested a higher tree abundance for residential areas. For instance, ResLow area quadrats had higher tree abundance than the other land use types excepting for Mid/high residential area. A lower quadrat tree evenness was revealed for ResLow area and ResHigh area.

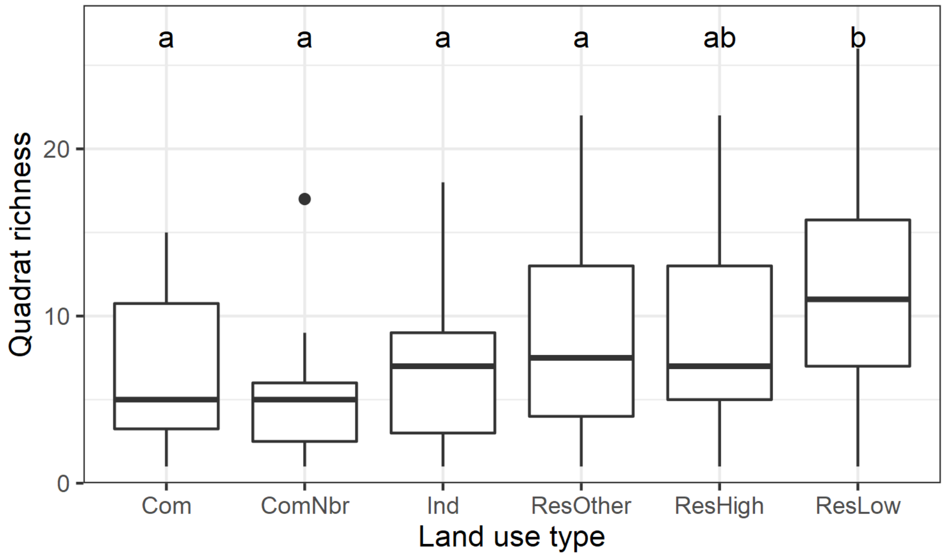

Figure 7. Comparison of quadrat richness across land use types. The figure shows the minimum, first quartile, median, third quartile, the maximum, and outlier if applicable. The different letters above the boxes indicate groups with statistically significant differences detected by post hoc pairwise comparison ($p < 0.05$).

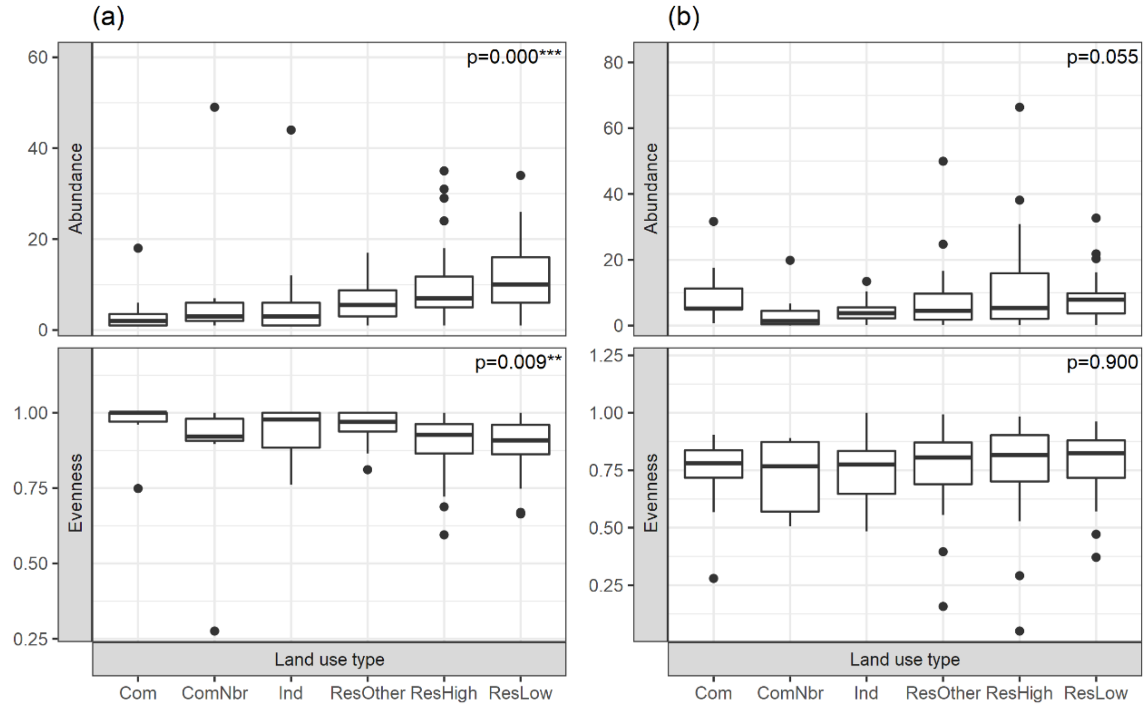

Figure 8. Comparison of quadrat abundance and evenness across land use types for (a) trees and (b) shrubs. The figure shows the p-values of Kruskal-Wallis rank sum test at the top-right of each sub-figure. The level of significance is denoted by asterisks: no asterisk, $p \geq 0.05$; *, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$.

Species composition differed across land use types for shrubs ($R = 0.059$, $p = 0.041$) but not for trees ($R = 0.021$, $p = 0.124$) at quadrat level (Figure 9). Further post-hoc pairwise comparison results revealed that the difference for shrubs mainly resulted from the difference between Com quadrats and ResLow quadrats ($p = 0.016$) and the difference between ResLow quadrats and ResHigh quadrats ($p = 0.041$).

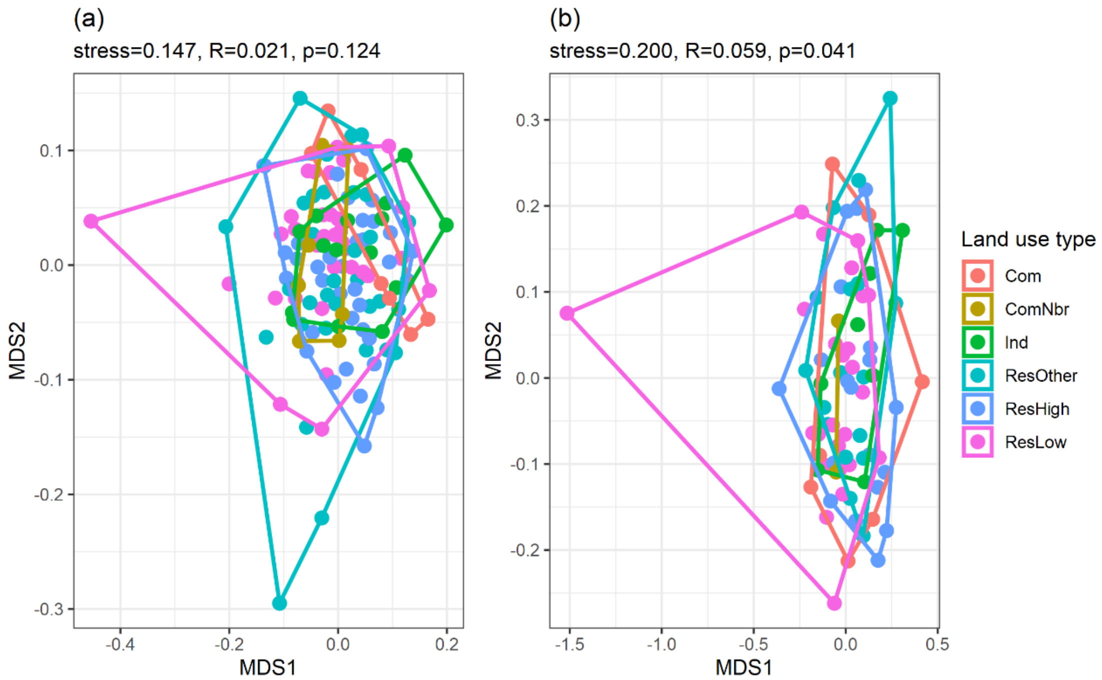

Figure 9. Comparisons of species composition of quadrats by land-use types for (a) trees and (b) shrubs.

3.3.4 Impact factors for quadrat biodiversity

The results of Pearson correlation test (Table 6) showed that urban gradient factors (distance to city center, land price, population density) had higher impact on tree dataset compared to all plant dataset and shrub dataset. The correlation between the biodiversity metrics (richness, abundance, and evenness) and for all plants or shrubs was not significant, while land price had a negative association with tree richness and abundance, distance to city center had a positive correlation with tree abundance and a negative association with tree evenness. Regarding the variables related to land cover proportion, the proportion of residential was positively correlated with richness of shrub, but the proportion of multi-family residential area had a negative association, which might indicate the contribution of residential yard to biodiversity. It was not surprising that the percentage of transportation had a negative association to richness of all plant dataset, tree dataset, and shrub dataset. Besides, the percentage of transportation also had a negative correlation with tree abundance.

Table 6. The Pearson correlation between the variables of quadrats and species richness, abundance, and evenness with test of all plant dataset, tree dataset, and shrub dataset. The variables of proportion of each land cover are denoted with the name of the land cover and "%" in the explanatory variable column.

Metrics	Explanatory variable	All plant	Tree	Shrub
Richness	distance to city center	0.07	0.15	0.02
	land price	-0.06	-0.19*	-0.02
	population density	0.01	-0.10	0.06
	agriculture%	-0.11	0.03	-0.07
	residential%	0.16	-0.01	0.29**
	multi-family residential%	-0.28	-0.13	-0.34*
	commercial/industrial%	-0.29	-0.59	-0.04
	institutional%	0.01	0.15	-0.09
	park%	-0.28	-0.11	-0.24
	transportation%	-0.31**	-0.25*	-0.30**
Abundance	distance to city center	-	0.20*	0.14
	land price	-	-0.18*	-0.04
	population density	-	-0.02	-0.08
	agriculture%	-	-0.15	-0.09
	residential%	-	-0.06	-0.12
	multi-family residential%	-	-0.15	-0.27
	commercial/industrial%	-	-0.37	-0.40
	institutional%	-	0.07	0.17
	park%	-	0.23	-0.03
	transportation%	-	-0.21*	0.00
Evenness	distance to city center	-	-0.18*	0.09
	land price	-	0.13	-0.10
	population density	-	-0.08	-0.09
	agriculture%	-	-0.04	-0.34
	residential%	-	0.07	0.03
	multi-family residential%	-	0.13	0.23
	commercial/industrial%	-	0.08	0.37
	institutional%	-	0.37	-0.16
	park%	-	-0.27	0.25
	transportation%	-	-0.02	-0.14

The results of linear regression (Table 7) also showed that urban gradient to tree dataset compared with all plant dataset and shrub dataset, and the effect was only limited for tree richness. On contrast, the metrics of the dataset were more likely to be better predicted by land cover proportion. The impact of the proportion of residential area and the proportion of multi-family residential area were similar as the results detected by Pearson correlation. The proportion of agriculture area had a negative association with tree richness since it is dominated by herbage. The proportion of commercial and industrial area was negatively associated with the richness of all plants and trees, the abundance of trees, and evenness of trees and shrubs, a possible reason is the available space for plants in industrial area or commercial area is limited.

Table 7. Multiple regression analyses for urban gradient measures and onsite land cover proportion to biodiversity metrics of all plant dataset, tree dataset, and shrub dataset. The variables of proportion of each land cover are denoted with the name of the land cover and "%" in the explanatory variable column. Significance is indicated as: *, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$.

Metrics	Explanatory variable	All plant	Tree	Shrub
Richness	distance to city center	0.00	-	0.00
	land price	-	0.00**	-
	population density	-0.01	-0.01*	0.00
	agriculture%	-6.48	-4.88*	-2.95
	residential%	3.70*	-	4.27***
	multi-family residential%	-3.31	-1.72	-1.84
	commercial/industrial%	-6.80**	-5.01**	-3.14
	institutional%	-2.63	-2.47	-1.23
	park%	-	-0.78	-
	transportation%	-1.43	-2.07*	-0.68
R square of the model		0.20	0.16	0.21
Abundance	distance to city center	-	0.00	0 .00
	land price	-	-	0.00
	population density	-	0.00	-0.01
	agriculture%	-	-2.95	-9.85
	residential%	-	4.27***	-3.94
	multi-family residential%	-	-1.84	-3.42
	commercial/industrial%	-	-3.14	-9.19*
	institutional%	-	-1.23	-
	park%	-	-	6.52
	transportation%	-	-0.68	-
R square of the model		-	0.21	0.14
Evenness	distance to city center	-	0.00	-
	land price	-	0.00	0.00
	population density	-	-0.01	0.00
	agriculture%	-	-9.85	-0.14
	residential%	-	-3.94	0.11*
	multi-family residential%	-	-3.42	0.12
	commercial/industrial%	-	-9.19*	0.17*
	institutional%	-	-	-0.02
	park%	-	6.52	-
	transportation%	-	-	0.01
R square of the model		-	0.16	0.10

3.4 Discussion

3.4.1 Urban exotic species proportions at city level

Despite a potentially high plant diversity, urban areas typically feature high concentrations of exotic species due to human introduction (Aronson et al., 2015). I collected the percentage data for exotic species from other studies (Table 8). The percentage in our research for Kyoto City (50%) is nearly average among the cities. The percentage of exotic species varies among city contexts, possibly in relation to climate (Avolio et al., 2015), history (Aronson et al., 2014) and development levels. For instance, in drier cities, trees of exotic species are often artificially cultivated as part of urban development efforts (Avolio et al., 2015). Research on Danish temporal changes in species composition suggested a faster increase in the presence of exotic species than for native species in urban development (Nielsen et al., 2019). Proportion differences may also be attributable to research methodology relating to geographical extent, target land use types and criteria of plant provenance. For instance, previous studies have shown that exotic plant species richness and abundance increase with greater urban land use coverage (Aronson et al., 2015; Pennington et al., 2010). Regarding the impact of target land use types covered in the research, Chimaimba et al. (2020) found that the proportion of exotic species in residential areas, along roads and at institutions was much higher than in graveyards, afforestation hills, and parks in Zomba of Malawi.

Table 8. The proportion of exotic species for woody plants or trees in some cities

Reference	Study area	Number of species recorded	Percentage of exotic species
Avolio et al. (2015)	three southern California counties, US	114	93%
Raoufou et al. (2011)	Lomé, Togo	297	69%
Ortega-Álvarez et al. (2011)	Mexico City, Mexico	89	66%
Guo et al. (2018)	Beijing, China	148	57%
Ouyang et al. (2015)	Xi’an, China	176	45%
Dangulla et al. (2019)	Zaria and Sokoto, Nigeria	56	39%
Chimaimba et al. (2020)	Zomba, Malawi	168	36%
Muthulingam and Thangavel (2012)	Chennai metropolitan city, India	45	31%
Godefroid and Koedam (2007)	city of Brussels, Belgium	702	23%
Zhao et al. (2013)	Chongming island, China	42	< 20%
This research	Kyoto city, Japan	223	50%

Although half the species recorded in the current research were exotic, the proportion was much lower regarding abundance (26% for trees and 24% for shrubs). The Malawi study conducted by Chimaimba et al. (2020) also suggested inconsistencies in exotic species percentage based on species number and abundance data. As ecosystem functionality may depend more on species dominance (Genung et al., 2020), the proportion of exotic species by abundance should be emphasized in future research in case of possible overestimation of the importance of rare exotic species.

3.4.2 Measurements of urban plant diversity

Biodiversity is a complex concept with controversy in measurement despite acknowledged definitions. From the perspective of scale, while biodiversity indexes at quadrat level has been widely applied in gradient-based studies (Aronson et al., 2015; Brunzel et al., 2009; Hahs and McDonnell, 2007; Vakhlamova et al., 2014a; Wang et al., 2020), indexes at aggregate level has been frequently used in discrete-variable-based research like the studies on biodiversity across land use types other than quadrat diversity (Bourne and Conway, 2014; Chimaimba et al., 2020; Guo et al., 2018; Liu et al., 2018; Yang et al., 2017). However, since biodiversity is scale-dependent, higher diversity at quadrat level does not necessarily represent higher diversity at aggregated level. Inconsistencies between quadrat richness and richness at aggregated level were found in previous research (Bourne and Conway, 2014; Ortega-Álvarez et al., 2011; Thompson et al., 2003). Thus the hierarchical framework of *alpha-*, *beta-* and *gamma-*biodiversity indexes should be applied (Robert H. Whittaker, 1977; Tuomisto, 2010; Whittaker, 1960; Whittaker et al., 2001) and scale should be emphasized to achieve a better comparability between research when calculating urban biodiversity.

For comparison of richness at aggregated level, most studies treated the sum of quadrat richness as a precise estimation of total richness at the scale of interest, which can be heavily biased since species number increases with sampling effort (e.g., sample quadrat area or the number of quadrats). It is not appropriate to compare richness without a calibration of sample size (e.g., Chimaimba et al. (2020) compared total richness of each land use measured with different number of quadrats). Here the results suggest the method based on species accumulation curves as a useful tool for a reliable total richness estimation to eliminate the bias caused by sample size (Chao et al., 2014; Willis, 2019).

3.4.3 Land use type, scale, and plant diversity

For comparison with other studies at quadrat level, average values of quadrat richness and abundance of this research were extracted and calculated for all woody plants and for trees across land use types (Table 9; results from other studies linearly converted with a 400 m^2 base). It should be noted that since many of the studies didn't indicate the distribution or statistical analysis results of quadrat richness or abundance, the comparison in this section only focuses on the average values. The mean quadrat richness for residential areas in Kyoto City (ResLow, 11.70; ResHigh, 9.05; ResOther, 8.52) is higher than that in Beijing (8.46, Guo et al., 2018) for woody plants; while that for trees in Kyoto City (ResLow, 5.77; ResHigh, 4.84; ResOther, 4.03) is lower than Mexico City (9.99, Ortega-Álvarez et al., 2011) and the Peel region of Canada (5.99, Bourne and Conway, 2014). Quadrat tree abundance for residential areas in this research (ResLow, 11.40; ResHigh, 9.68; ResOther, 6.41) is also lower than that of Mexico City (20.0, Ortega-Álvarez et al., 2011), but quadrat tree abundance of ResLow area and ResHigh quadrats are higher than that in the Peel region (9.0, Bourne and Conway, 2014).

Table 9. Comparison of quadrat richness and abundance across land use types in some cities. The area of sample quadrats differs across studies, so the richness and plant abundance has been linearly converted by a base of 400 m^2 in this table.

Reference	Study area	Biological group	Land use type	Richness	Abundance
Guo et al. (2018)	Beijing, China	woody plants	residential	8.46	-
			community park	7.95	-
			institutional	7.38	-
			commercial	6.01	-
			roadside	5.96	-
			riverside	5.21	-
			municipal park	5.11	-
			woodlot	2.53	-
Ortega-Álvarez et al. (2011)	Mexico City, Mexico	trees	residential	9.99	19.98
			residential-commercial	8.97	3.63
			commercial	6.58	5.40
			green area	4.12	19.98
Bourne and Conway (2014)	Peel region, Canada	trees	agriculture	2.00	2.99
			commercial	2.00	2.99
			golf	5.99	38.91
			institutional	4.99	11.97
			parkland	4.99	20.95
			residential	5.99	8.98
			transportation	4.99	20.95
			vacant	5.99	38.91
Dangulla et al. (2019)	Zaria, Malawi	trees	built-up	-	2.79
			farmland	-	1.42
			green area	-	5.48
			open space	-	1.51
			wetland/water	-	1.72
Dangulla et al. (2019)	Sokoto, Malawi	trees	built-up	-	1.93
			farmland	-	0.64
			green area	-	7.44
			open space	-	0.52
			wetland/water	-	2.57
This research	Kyoto, Japan	trees	Com	2.25	3.67
			ComNbr	2.89	8.56
			Ind	3.17	5.83
			ResOther	4.03	6.41
			ResHigh	4.84	9.68
			ResLow	5.77	11.40
This research	Kyoto, Japan	woody plants	Com	6.93	-
			ComNbr	5.60	-
			Ind	7.00	-
			ResOther	8.52	-
			ResHigh	9.05	-
			ResLow	11.70	-

Differences in environment, available space, greening goals and types of people making decisions (Bourne and Conway, 2014; Godefroid and Koedam, 2007) lead to different patterns of plant diversity across land use types and scale. In listed studies ((Bourne and Conway, 2014; Godefroid and Koedam, 2007), including this research), quadrat richness of residential areas are all higher than that of commercial areas (Bourne and Conway, 2014; Guo et al., 2018; Ortega-Álvarez et al., 2011). Furthermore, a similar pattern was found at land use level that the estimated total richness in residential areas is higher than that in industrial area and commercial areas in Kyoto City. The results are likely associated with the effects of space available for plants. Unlike residential areas, commercial areas are characterized by high-density commercial centers and offices, which lack green spaces and private gardens (Godefroid and Koedam, 2007). For industrial area, greening ratios for factory sites in Japan must exceed 10% legally (Ministry of Economy, Trade and Industry, Government of Japan, 2017), but planted species tend to be function-oriented as well as being easy to maintain. The results also emphasize the critical impact of residents' preferences. Unlike city planners and factory owners who may choose plants from a limited number of species to achieve public greening goals, residents tend to select from a larger species pool for cultivation on their properties (Avolio et al., 2018). The results here also suggest that the dispersed green spaces like private gardens are essential for maintaining urban biodiversity (Goddard et al., 2010). Whereas, on the other hand, the demolition of traditional private garden due to the lifestyle change (Niino et al., 2021) might lead to a decrease in the urban plant diversity in Kyoto City.

Even within residential areas, diverse building types and residential conditions lead to differences in plant diversity (Bourne and Conway, 2014; Troy et al., 2007). Among three types of residential areas in Kyoto City, ResLow area has the highest estimated total richness at land use level and highest quadrat richness. The results are partially attributable to larger plantable areas and species turnover rates in ResLow area. In Japan's cities, ResLow area generally features low-rise houses with yard, along with limited numbers of other buildings with lower greening rates such as recreational facilities, hotels and institutions. Residential buildings in ResHigh and ResOther area are generally high-rise structures such as multi-family apartments with no gardens.

Despite higher richness at both land use level and quadrat level, ResLow area exhibited moderate to low evenness. This finding is consistent with Bourne and Conway (2014), and might result from a mix of varied unique species in residential areas due to diverse choices among homeowners as opposed to relatively uniform greening in commercial and industrial areas. The outcomes further suggest that evenness should be applied as a supplement for richness in plant diversity research.

3.4.4 Urban gradient, land cover, and quadrat plant diversity

Urban gradient analysis has been widely applied in urban biodiversity research (Vakhlamova et al., 2014b). While many patterns of plant diversity along the urban gradient were found in previous studies, the results could be determined by the target taxa, provenance, the metrics of urban gradient, and the scale of research. (1) Target taxa. The urban area has more woody plant species than the surrounding less developed areas like suburban and rural area in Shanghai (Wang et al., 2020), while a research in Kazakhstan focusing on spontaneous plants showed that the quadrat richness and Shannon index are positively associated with the distance of the quadrat to city center (Vakhlamova et al., 2014b). In this study, I also found a positive but not significant correlation between the distance of quadrat to city center with richness, which is partially consistent with the results of woody plants research in Shanghai. (2) Provenance. A study of Tampere of Finland suggests that the richness of non-native species peaks at suburban area, while the richness of native species is higher in the suburban and the rural (Ranta and Viljanen, 2011). (3) The surrogate metrics for urbanization. Though distance to city center has been widely used as the proxy of urban gradient, it is controversial that it assumes a gradual change of urbanization along the physical transect, and it can conflict with other urban gradient metrics like population density or another socio-economic gradient. Theoretically, the distance metrics is more suitable for a monocentric city with an urban core area. Other than a single measure of urban gradient, complex metrics has also been applied. For instance, a combined urbanization index (Hahs and McDonnell, 2006) was applied in a Melbourne's urban gradient research (Hahs and McDonnell, 2007). Another study used Delphi method which is based on experts' experience and knowledge to determine the urbanization gradient (Porter et al., 2001). (4) Scale. The scales of urbanization might be limited in urban built-up area or extend to surround rural area or even natural area (e.g., forest). The presence of peak diversity value also varies with the scale.

It turns out that the distance of the quadrats to city center is not a good explanatory variable for biodiversity prediction in this study. Among the test, only the association between tree abundance and the distance to city center was positive and significant revealed by Pearson correlation test. The result of higher diversity in the urban core is explained by the high demand for green space in urbanized areas (Wang et al., 2020), which is paradoxical with the high built-up density of urban core (Troy et al., 2007). And intermediate disturbance hypothesis was frequently applied to explain the so-called suburban peak pattern of plant diversity(Ortega-Álvarez et al., 2011; Wang et al., 2020). While the results from the studies and the hypothesis might not be applicable to predict our result for the following reason: (1) the Kyoto city is characterized with mixed function, which leads to a more evenly spatial distribution of anthropic disturbance, that the distance to the city center is not a good presenter of disturbance intensity; (2) the suburban peak pattern are usually found for spontaneous plants and herbs, while rarely for woody plants which are mainly planted (Ortega-Álvarez et al., 2011); (3) and the pattern is also related to boundary and scale (Aronson et al., 2015) – since our study is limited to Kyoto city built-up area, the extent of our study area might not include less developed rural area.

Land price is negatively correlated with tree richness and abundance, and population is negatively associated with tree abundance revealed by linear model in this study. The results of land price is contrast to luxury effect, which generally reveal the positive relationship between median income and biodiversity (Chamberlain et al., 2019). This result indicates the difference pattern of biodiversity along land price gradient and income gradient. According to the spatial data, land price is generally higher in city center, while the reason for that is more likely because of the commercial value of the land. In many cities, wealthier people tend to live in the surrounding aera even the suburban of cities because they can afford a bigger house and the commuting cost. The seemingly opposite results require further research with more comprehensive socio-economic dataset.

Compared to urban gradient variables, on site land cover proportion variables turn out to be better predictors for the biodiversity metrics. The positive impact of residential area and the negative impacts of the proportion of multi-family residential area, commercial and industrial area, and transportation area support the conclusion that private land has critical contribution to urban biodiversity. Among the habitat change, agriculture is another key factor driving biodiversity loss globally. In this study, agriculture area percentage has a negative impact on tree richness since it is dominated by herbages like crops and vegetables.

3.5 Conclusion

In this study, I explored woody plant diversity across land use and scale in Kyoto City. At land use level, residential areas had higher total richness with moderate to low overall evenness, while commercial areas had relatively lower total richness. A high species composition dissimilarity was identified between residential areas and other land use types. At quadrat level, ResLow area had higher richness than the other land use excepting for ResHigh area. Quadrat abundance and evenness were different across land use types for trees but not for shrubs. Quadrat species composition was significantly different across land use types for shrubs, but not for trees. The test of relationship of urban gradient and onsite land cover with the quadrat biodiversity metrics reveals that onsite land cover proportion variables are better predictors for the metrics. Specifically, residential area percentage has a positive impact while multi-family residential area and transportation percentage have a negative association with the metrics.

From the perspective of urban management practice, this research identified prior land use types for biodiversity improvement. For instance, the commercial areas are characterized with lower plant diversity but more visitors and higher population density. An improvement of plant diversity in these land use types can enhance the potential ecosystem services benefit due to a higher beneficiary population and higher accessibility of the plants in public space. The contribution of ResLow area for urban biodiversity conservation was also further proved.

The results also suggest that the mechanism underlying urban plant diversity requires further research, especially in regard to the heterogeneous impact of different land use types at various of scales.

Chapter 4 Urban Ecosystem Services

4.1 Introduction

The world's urban population is expected to increase from 55% in 2019 (The World Bank Data, no date) to 68% by 2050 (Population Division, United Nations, 2019), which inturn, leads to a growth of demand for ecosystem services in cities. The relation between demand and supply of ecosystem services varies with scale. Locally generated ecosystem services are more closely related to the living quality of the resident, and some of them are irreplaceable by other distant sources of ecosystem services (for example, mitigation of heat island effect) (Gómez-Baggethun et al., 2013). Considering the numerous population size in cities, the social and economic value of ecosystem services within cities can be surprisingly high (Gómez-Baggethun and Barton, 2013). Besides, a global assessment highlighted how massive urbanization is impacting biodiversity and ecosystems around the world negatively (Elmqvist et al., 2013). Therefore, an improvement of urban ecosystem services could potentially benefit city residents and mitigate the loss of ecosystem services globally.

Yet despite the importance of urban ecosystem services evaluation (Gómez-Baggethun et al., 2013), most of the studies and the implementation of the research findings into land use policy, are from North America, Europe, and China (Haase et al., 2014; Ordóñez-Barona et al., 2019) (case studies see New York City (Kremer et al., 2016b) and Berlin (Larondelle and Lauf, 2016)). Besides, even though related evaluation tools like i-Tree have been widely applied in many cities around the world, urban ecosystem services research in Japan has been less addressed. A pilot study evaluated the ecosystem services of street trees in Kawasaki City in Japan using i-Tree (Hirabayashi et al., 2016). Other than that, only some case studies using a similar approach were found (Hirabayashi et al., 2019; Kawaguchi et al., 2021; Tan et al., 2021).

Ecosystem services are estimated with a variety of methods, including indicators and valuation. Indicators are used to quantify the state and change of the objects of interest. Some of the commonly used indicators are crop yield for food production, carbon storage and carbon sequestration for climate change mitigation, and runoff reduction for hydrological regulation. Regarding the valuation, two methods are applied to estimate ecosystem services' monetary value. One is the traditional economic method using firsthand data, including the stated preference method and revealed preference method. Though the empirical, field-based method can provide more accurate results (Zhao and Sander, 2018), it is time-consuming and limited on the scale. Therefore, the other method, value transfer (or 'benefit transfer') is widely used in ecosystem evaluation (Kremer et al., 2016a), for which the monetary value estimation of one location (the 'reference ecosystem') is transferred to another (the 'target location')

(Costanza et al., 2017). The value transfer method is frequently applied in regional services estimation based on the area of land use/land cover types and per unit area ecosystem service value of each type. In these studies, cities are categorized as ‘urban area’ or ‘built-up area,’ and the ecosystem service of the category is estimated with a constant per unit area ecosystem service value. Particularly, the per unit area ecosystem service value for urban ecosystem from Costanza et al. (2014) has been widely applied (e.g., see Arowolo et al., 2018; Yi et al., 2017). Some other research modified the per unit area ecosystem service value based on the local context like scarcity value effect (Bryan et al., 2018). However, the land use/land cover-based value transfer method could cause uncertainty in urban ecosystem service estimation since it ignores the high heterogeneity in cities and rapid change of land use/land cover (Haase et al., 2014; Kremer et al., 2016a). To get a more specific per unit area ecosystem service value for urban ecosystems, within-city research and inter-city comparison research is needed.

Among the service providers in urban ecosystems (e.g., forest patches, waterways and lakes, parks, brownfields, urban agriculture (Haase, 2013; Haase et al., 2014; Kim et al., 2015)), urban forest is one of the foremost. As a crucial local ecosystem services provider in cities, urban forest functions in many services like carbon storage and sequestration, noise reduction, air quality improvement, energy conservation, and recreation (Bolund and Hunhammar, 1999; Gómez-Baggethun et al., 2013). However, the ecosystem services of urban forests might have been underestimated since many previous studies focused on remnant forests or street trees (e.g., (Szkop, 2016; Tang et al., 2016)), partially due to data availability. However, the dispersed green spaces such as private gardens have been less studied, despite the fact that their importance to urban ecosystem services has been proved (Camps-Calvet et al., 2016; Haase, 2013; Kim et al., 2015).

To estimate the ecosystem services of urban forests more precisely, i-Tree Eco has been applied worldwide in more than one hundred countries. Developed by the United States Department of Agriculture, i-Tree Eco allows users to calculate several ecosystem services (carbon storage and sequestration, pollutants removal, runoff reduction, etc.) of each tree with field investigation data of tree species, size, and condition. Though i-Tree Eco enhances users to manage urban forest more accurately, even at a single-tree level, most research only presented the results of inferred total ecosystem services of the whole research area (e.g., see (Nowak et al., 2016)) or results by species (Kiss et al., 2015; Ning et al., 2016). One possible reason is being guided by the automatically generated report of the tool. These results, however, provide little information on the link between within-city heterogeneity and urban ecosystem services. Only a few research reported ecosystem services across land use/land cover within cities (Baro et al., 2014; Kim et al., 2015).

To address the gaps mentioned above, I conducted an urban ecosystem services evaluation at a Japanese city, Kyoto. The study is partially aimed at enriching the data base of urban ecosystem services with detailed ground-based investigation data and i-Tree Eco tool. Another main objective of this chapter is to link urban heterogeneity and urban ecosystem services by comparing ecosystem services across land use. I expected that ecosystem services would differ across land use types.

In this study, a pre-stratified sampling method based on the area of land use classes was applied for field data collection, then i-Tree Eco tool was used to calculate the urban forest structure, tree compensatory value, and ecosystem services. The ecosystem services, including carbon storage and sequestration, air pollutants removal, and rainwater runoff reduction, were estimated for the entire study area and allocated to each tree, then further grouped by quadrat. I compared ecosystem services at both quadrat level and single-tree level across land use classes. For a better understanding of the link between heterogeneity and ecosystem service, the results of Kyoto City were also compared with the studies of other cities.

4.2 Methods

4.2.1 Data preparation

A subset of data with the following information of each tree was extracted from the data set formed in Chapter 2: (1) species; (2) size-related data for trees: height, diameter at breast height (DBH), crown size; (3) health status: canopy missing percentage, crown health condition, and crown light exposure. It should be noted that this chapter mainly focuses on efficiency of ecosystem services (per 20 m × 20 m quadrat ecosystem services) across land use, thus a evaluation of ecosystem services for each individual plant is required. While i-Tree Eco, the tool applied for ecosystem service calculation here, can only produce the output of individual for trees while not for shrubs. Therefore, only the subset for tree was extracted for the analysis of this chapter from the complete data set. The number of trees and other information for each land use are shown in Table 10.

Table 10. Sample quadrats by stratified sampling method in this research.

Land use class	Area (ha)	Proportion of area	Number of quadrats	Number of trees
ResLow	3519	24%	35	399
ResHigh	3027	21%	38	368
ResOther	3113	21%	34	218
Ind	3213	22%	23	134
ComNbr	864	5%	9	77
Com	1009	7%	12	44

4.2.2 Evaluation of ecosystem services and monetary value

i-Tree model has been widely used to help managers and researchers to quantify urban forest structure, ecosystem services, and tree monetary value. I calculated three values of each tree: compensatory value, representing compensation for the loss of a tree (Council of Tree and Landscape Appraisers, 1992; Nowak, 2002a); monetary value of carbon storage, representing the cumulative result of net carbon sequestration for years; annual ecosystem services, including carbon sequestration, air pollutants removal, and runoff reduction. Though cultural service is one of the most critical components of ecosystem services in cities, i-Tree is not capable of calculating it for now. I will briefly introduce the method for structure and ecosystem services evaluation, and valuation of tree monetary value in the following sections; for more details refer to i-Tree method documentation (Nowak, 2020). To improve the accuracy of results, a modified i-Tree model with local parameters of Kyoto City was applied (see Table A3 for model details and parameters list). The basic operating steps are to input the data into i-Tree Eco software interface with a new program, and submit the program to i-Tree server, then the output will be sent to the user after the calculation is finished. However, generally, the output is a report with inferred ecosystem services of the whole research area in PDF format. I cooperated with an environmental modeler, Dr. Hirabayashi, in this study that the original output data of each tree from i-Tree Eco was available.

(1) Structure

Leaf area is estimated based on species, total height, crown base height, crown width, and percent crown missing. The method is a species-specific regression equation with a shading coefficient (percent light intensity intercepted by foliated tree crowns) for deciduous urban species, while a shading coefficient of 0.91 is applied for conifer trees (i-Tree, 2019). Leaf area index (LAI) is calculated with leaf area and adjusted with the overlap of tree crowns or light exposure. Leaf biomass is calculated based on leaf area with species-specific convert factor. Total biomass for each tree is calculated using species-specific allometric equations from the literature with DBH and total height (Nowak, 2002b, 1994).

(2) Carbon storage and carbon sequestration

Carbon storage is estimated based on biomass and carbon content. For evergreen and palm species, leaf biomass is added. Carbon sequestration is estimated based on the growth rate. The growth rates are estimated with the measurement of radial growth increments (Nowak, 1994), duration of the growing season, and the growth adjustment factor of crown health and crown light exposure (i-Tree, 2019). For valuation of the ecosystem services, the social cost for carbon in Japan (10,600 Yen, which is about 96 US dollars per ton carbon) from the Japanese government document (Ministry of the Environment, Government of Japan, 2019) was applied.

(3) Air pollutants removal and health benefits

Air pollution removal is estimated using the percent tree cover and leaf area index. The pollutants estimated include nitrogen dioxide (NO_2), ozone (O_3), particulate matter less than 2.5 μm ($PM_{2.5}$), and sulfur dioxide (SO_2). In the locations supported more sufficiently in i-Tree Eco (e.g., cities in the US and Canada), the tree data is merged with local pre-processed weather and air pollution concentration data for the evaluation of pollutants removal. However, in this study, since Kyoto City is not officially supported by default, the local weather data was entered from local monitor stations manually. The value of air pollutant removal is assessed by the BenMAP method (Nowak et al., 2014) that estimates avoided costs for adverse health incidences based on the air quality improvement and medical records across the US.

(4) Rain water runoff reduction

Runoff reduction in i-Tree Eco is estimated based on the difference between the runoff with current tree cover and that without trees. In the simulation, rainfall interception of trees and runoff are calculated mainly by precipitation, leaf area index, and infiltration with a time step of an hour (Wang et al., 2008). One limitation of the model is that the water reaching pervious surface is assumed to be absorbed by the soil, while the water reaching the impervious surface is assumed to become urban surface runoff. Besides, though the impervious cover rate is estimated by JAXA satellite imagery, the number is assumed to be constant across the research area. To reflect the local economic benefit of the ecosystem service, I used the stormwater control facilities cost estimation of Suita City of Japan (719 yen per m^3, which is about 7 US dollars per m^3) for the valuation (Kawaguchi et al., 2021).

(5) Compensatory value

The compensatory value of trees is estimated using the guideline of Council of Tree and Landscape Appraisers (Council of Tree and Landscape Appraisers, 1992) in i-Tree Eco (Nowak, 2020). The compensatory value of a tree is determined by replacement cost, DBH, and a location-specific per unit trunk area cost. For palm trees, the cost to clear trunk is also considered. The values of these parameters have been compiled for numerous states in the US;

while for other countries, an average value of replacement cost and per unit trunk area cost is applied.

4.2.3 Data analysis

The collected and calculated data for each single tree was then added to get a quadrat dataset, including average DBH, average LAI, the total number of trees, and the total ecosystem services of each quadrat. Since the assumption of normality for the metrics is violated in this case, the non-parametric statistic method, Kruskal-Wallis rank sum test, was used to analyze the difference of DBH, LAI, the number of trees, and each ecosystem service among land use classes. For the statistical group comparison where a significant difference was detected, Dunn's test was then applied for a post hoc pairwise comparison. Similarly, at the single-tree level, Kruskal-Wallis rank sum test and Dunn's test were used to test the differences of DBH, LAI, and each ecosystem service across land use classes respectively.

Furthermore, a species-specific analysis was used to compare the single-tree ecosystem services across land use classes by species. To achieve a robust result, only widespread species presenting across a sequence of land use classes with at least 3 individuals for each land use class were analyzed. The target species include *Acer palmatum*, *Ginkgo biloba*, *Ligustrum lucidum*, *Nandina domestica*, *Osmanthus fragrans*, *Podocarpus macrophyllus*, *Prunus x blireana*, *Quercus x alvordiana*, and *Zelkova serrata*.

All the analysis was conducted in R (version 4.0.3), and the difference was considered significant at $p < 0.05$. The $kruskal.test$ function was applied for Kruskal-Wallis rank sum test and $dunn.test$ function from $dunn.test$ package was used for the post hoc comparison.

4.3 Results

4.3.1 DBH and LAI

DBH is related to both age and species of trees. Trees with DBH $\leq$ 15 cm accounted for a large proportion across the land use classes (Figure 10). Ind zone had more trees with DBH > 15 cm than others, probably reflecting the low manage intensity of Ind zone and being constrained by both planting goals and limited space. ComNbr zone was characterized by a larger proportion of trees with DBH $\leq$ 15 cm than the others.

The LAI for most trees was under 6 (Figure 11). The proportion of LAI $\leq$ 3 is higher in ComNbr zone, followed by Ind and Com, and residential areas. Among the residential areas, ResLow and ResHigh were chartered by a lower proportion of LAI $\leq$ 3 and a higher proportion of LAI > 6, which might result from the maintenance of the house owners.

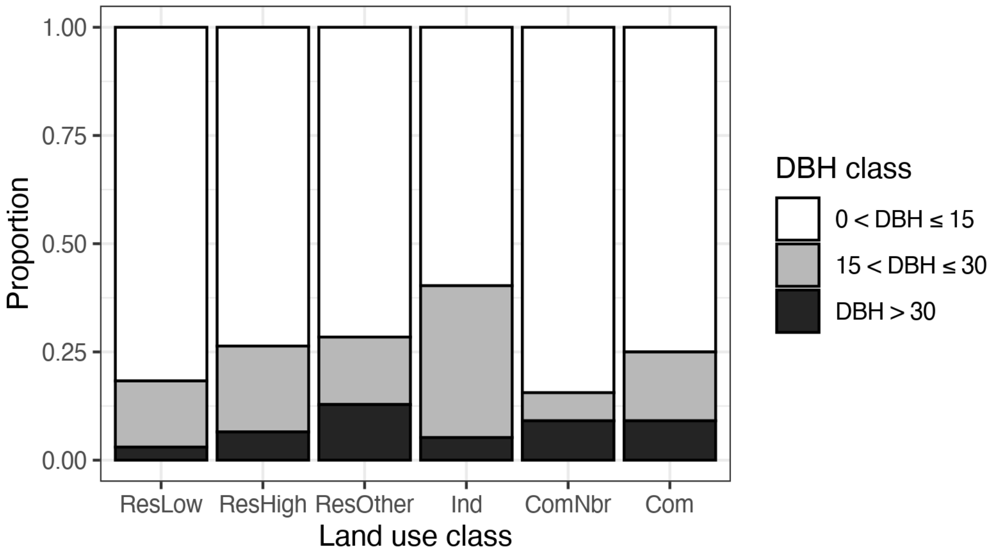

Figure 10. DBH (diameter at breast height) distribution across land use classes.

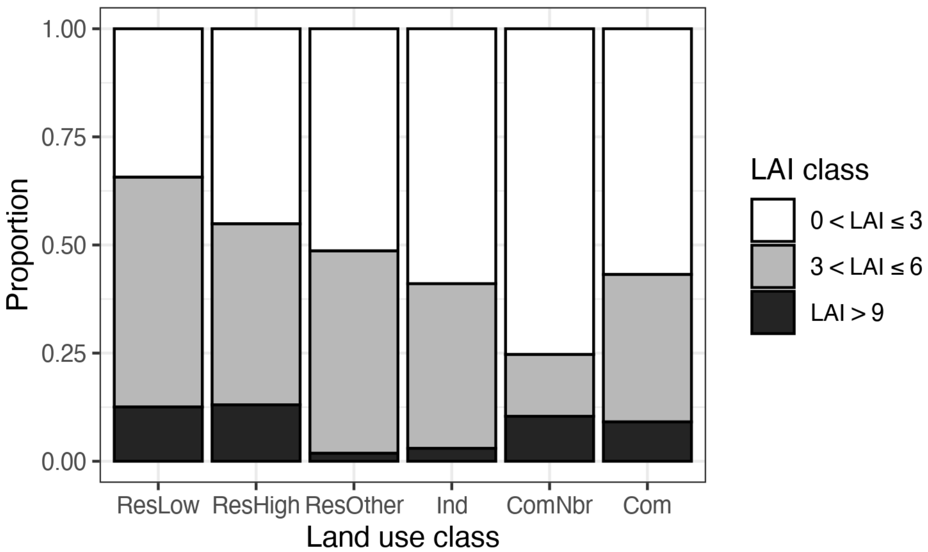

Figure 11. LAI (leave area index) distribution across land use classes.

4.3.2 Total monetary value of trees

The sum, average number, and median of quadrat annual ecosystem service values, carbon storage value, and compensatory value were calculated (Table 11). The average annual ecosystem service value is 30% as much as the average carbon storage value, while the compensatory value of trees is 121 times of annual ecosystem services. Due to the data distribution, the differences became smaller if comparing by median values. The quadrat annual ecosystem service median value is 63% of quadrat carbon storage median value, though the quadrat compensatory value is still 110 times of quadrat annual ecosystem service median value.

Table 11. Valuation of ecosystem services and compensatory value (unit: US dollars).

Item	Sum	Quadrat average	Quadrat median
Annual ecosystem services	4,285	28	10
Carbon storage	14,339	95	16
Compensatory value	518,712	3,435	1,128

Regarding the composition of quadrat annual ecosystem service values (Figure 12), $PM_{2.5}$ removal value accounted for about half of the total value, followed by O_3 removal, carbon sequestration, and runoff reduction value. NO_2 removal and SO_2 removal values only account for a small fraction of the total annual ecosystem service value.

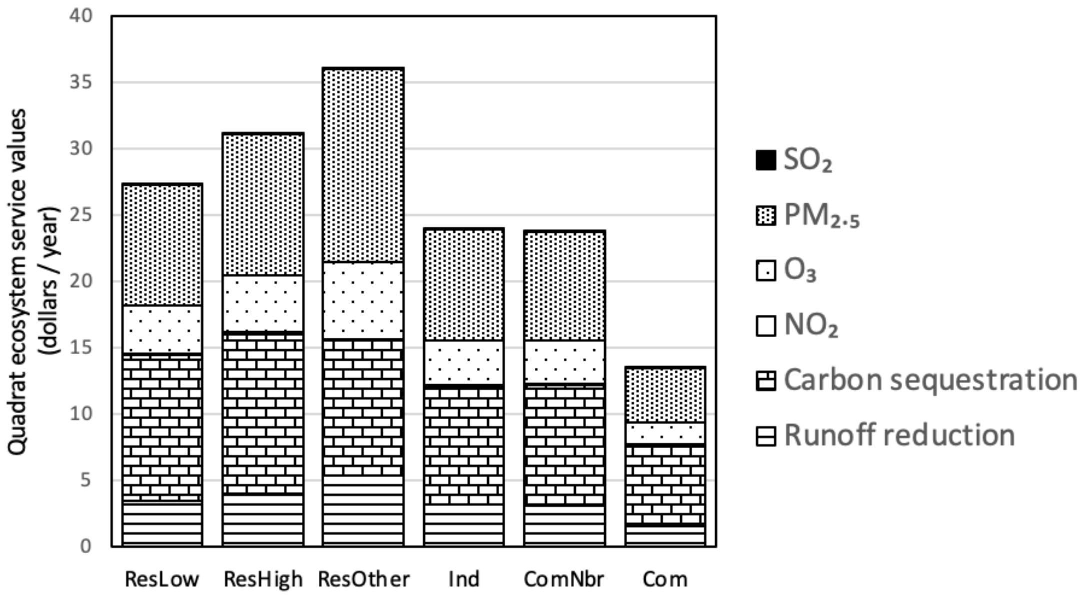

Figure 12. Average valuation of quadrat annual ecosystem services across land use classes.

But it should be noted that, though ecosystem services valuation is convenient for inter-ecosystem-services comparison, the monetary value varies with valuation method covering different aspects of total economic value; and it varies by local context like market value of the services. For example, in this case, the social cost for carbon in Japan is 10,600 Yen (about 96 US dollars) per ton (Ministry of the Environment, Government of Japan, 2019), while the monetary value for carbon is 188 US dollars per ton carbon for the US (Interagency Working Group on Social Cost of Greenhouse Gases, United States Government, 2016); the stormwater control facilities cost of Suita City, Japan is 719 yen (about 7 US dollars) per m^3 (Kawaguchi et al., 2021), while that estimation in the US is 2.36 US dollars per m^3. The comparison of monetary value between ecosystem services should thus be cautious. Researchers should pay attention to indicators of ecosystem services as well, rather than focusing on valuation only.

4.3.3 Quadrat ecosystem services across land use

No significant difference was found for quadrat ecosystem services across land use classes (Table 12).

Table 12. Chi-square statistics of comparison of ecosystem services across land use by Kruskal-Wallis rank sum test (the level of significance is denoted by asterisks: no asterisk, p ⩾ 0.05; *, p < 0.05; **, p < 0.01; ***, p < 0.001).

Scale	Carbon sequestration	NO_2 removal	O_3 removal	$PM_{2.5}$ removal	SO_2 removal	Runoff reduction
Quadrat level	7.68	9.53	9.04	8.00	9.34	7.97
Single-tree level	24.42***	55.68***	53.94***	51.98***	54.21***	51.68***

4.3.4 Single-tree ecosystem services across land use

Different from quadrat ecosystem services, the differences for all ecosystem services at the single-tree level across land use classes (Table 12) were significant. The average values and median values of ecosystem services at single-tree levels are shown in Table 13 and Table 14. Post hoc comparison indicates that the single-tree carbon sequestration in Com zone and Ind zone were higher than that in ResLow zone, and the carbon sequestration in Ind zone is higher than that in ComNbr zone. Trees in ComNbr zone had significantly lower air pollutants removal and runoff reduction than the others.

Table 13. Average values of ecosystem services of each land use at quadrat (per 20 m × 20 m quadrat) and single-tree level (per tree).

Scale	Land use	Carbon sequestration (kg)	NO_2 removal (g)	O_3 removal (g)	$PM_{2.5}$ removal (g)	SO_2 removal (g)	Runoff reduction (m3)
Quadrat level	ResLow	58.86	37.30	125.12	8.67	16.29	1.44
	ResHigh	63.89	43.05	145.62	10.20	18.90	1.70
	ResOther	53.87	64.47	208.21	13.69	27.42	2.28
	Ind	47.56	33.43	113.61	8.01	14.73	1.33
	ComNbr	48.53	33.80	112.92	7.78	14.72	1.30
	Com	32.36	14.61	52.78	4.00	6.72	0.67
Single-tree level	ResLow	5.16	3.27	10.98	0.76	1.43	0.13
	ResHigh	6.60	4.44	15.04	1.05	1.95	0.18
	ResOther	8.40	10.05	32.47	2.13	4.28	0.35
	Ind	8.16	5.74	19.50	1.37	2.53	0.23
	ComNbr	5.67	3.95	13.20	0.91	1.72	0.15
	Com	8.83	3.98	14.39	1.09	1.83	0.18

Table 14. Median values of ecosystem services of each land use at quadrat (per 20 m × 20 m quadrat) and single-tree level (per tree).

Scale	Land use	Carbon sequestration (kg)	NO2 removal (g)	O3 removal (g)	PM2.5 removal (g)	SO2 removal (g)	Runoff reduction (m3)
Quadrat level	ResLow	35.31	15.49	51.95	3.43	6.84	0.57
	ResHigh	22.88	17.92	54.38	3.24	7.31	0.54
	ResOther	20.83	12.76	40.77	2.54	5.39	0.42
	Ind	11.36	7.76	29.84	2.14	3.73	0.36
	ComNbr	19.66	6.00	19.61	1.31	2.57	0.22
	Com	12.78	2.54	8.65	0.61	1.12	0.10
Single-tree level	ResLow	2.45	1.13	3.66	0.26	0.47	0.04
	ResHigh	2.62	1.70	5.20	0.31	0.70	0.05
	ResOther	2.73	1.51	4.76	0.31	0.64	0.05
	Ind	4.31	1.85	5.99	0.38	0.78	0.06
	ComNbr	2.23	0.24	0.91	0.07	0.11	0.01
	Com	3.59	1.11	3.46	0.20	0.47	0.03

4.3.5 Species-specific analysis

The results of the species-specific comparison (Table 15) indicate that ecosystem services across land use classes were significantly different for most of the species, but not for all the species. *Osmanthus fragrans* and *Podocarpus macrophyllus* showed no difference for carbon sequestration; *Acer palmatum*, *Prunus × blireana* and *Zelkova serrata* showed no difference in all of the ecosystem services across land use classes. A further post hoc pair-wise comparison suggested that the land use class with higher ecosystem services varied with species. For example, *Ginkgo biloba* showed lower ecosystem services in Com (Table 16), while *Ligustrum lucidum* showed higher ecosystem services in Ind zone (Table 17).

Table 15. Chi-square statistics of species-specific comparison of ecosystem services across land use classes by Kruskal-Wallis rank sum test (the level of significance is denoted by asterisks: no asterisk, $p \geq 0.05$; *, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$).

Species	Distribution (> 3 Individuals per land use type)	Carbon sequestration	NO2 removal	O3 removal	PM2.5 removal	SO2 removal	Runoff reduction
Acer palmatum Thunb.	Ind, ResOther, ResHigh, ResLow	4.80	6.12	6.12	6.12	6.12	6.12
Ginkgo biloba L., 1771	Com, ComNbr, Ind, ResHigh	9.86 *	8.04 *	8.04 *	8.04 *	8.04 *	8.04 *
Ligustrum lucidum Ait.	Ind, ResOther, ResHigh, ResLow	18.14 ***	23.75 ***	23.75 ***	23.75 ***	23.75 ***	23.75 ***
Nandina domestica Thunb.	ComNbr, Ind, ResOther, ResHigh, ResLow	14.38 **	23.92 ***	23.92 ***	23.92 ***	23.92 ***	23.92 ***
Osmanthus fragrans Lour.	Ind, ResOther, ResHigh, ResLow	7.51	12.01 **	12.01 **	12.01 **	12.01 **	12.01 **
Podocarpus macrophyllus (Thunb.) Sweet, 1818	Com, ResOther, ResHigh, ResLow	7.51	10.89 *	10.89 *	10.89 *	10.89 *	10.89 *
Prunus × blireana	Com, Ind, ResOther, ResHigh, ResLow	0.78	3.49	3.49	3.49	3.49	3.49
Quercus × alvordiana	Com, ComNbr, Ind, ResOther, ResHigh, ResLow	54.36 ***	72.05 ***	72.05 ***	72.05 ***	72.05 ***	72.05 ***
Zelkova serrata (Thunb.) Makino	Com, Ind, ResOther, ResHigh, ResLow	7.19	5.53	5.53	5.53	5.53	5.53

Table 16. The post hoc comparison results of ecosystem services for *Ginkgo biloba* at single-tree level. The table shows the difference of mean values of land use 1 and land use 2. The level of significance is denoted by asterisks: no asterisk, p ⩾ 0.05; *, p < 0.05; **, p < 0.01; ***, p < 0.001.

Land use 1	Land use 2	Carbon sequestration	NO2 removal	O3 removal	PM2.5 removal	SO2 removal	Runoff reduction
Com	ComNbr	-0.87	-2.80*	-2.80*	-2.80*	-2.80*	-2.80*
Com	Ind	0.43	-1.20	-1.20	-1.20	-1.20	-1.20
Com	ResHigh	1.83	-1.38	-1.38	-1.38	-1.39	-1.39
ComNbr	Ind	1.33	1.51	1.51	1.51	1.51	1.51
ComNbr	ResHigh	3.04*	1.79	1.78	1.79	1.79	1.79
ComNbr	Com	1.34	0.00	0.00	0.00	0.00	0.00

Table 17. The post hoc comparison results of ecosystem services for *Ligustrum lucidum* at single-tree level.

The table shows the difference of mean values of land use 1 and land use 2. The level of significance is denoted by asterisks: no asterisk, p ⩾ 0.05; *, p < 0.05; **, p < 0.01; ***, p < 0.001.

Land use 1	Land use 2	Carbon sequestration	NO2 removal	O3 removal	PM2.5 removal	SO2 removal	Runoff reduction
Ind	ResHigh	1.60	1.52	1.52	1.52	1.52	1.52
Ind	ResLow	3.53**	3.70***	3.70***	3.70***	3.70***	3.70***
Ind	ResOther	3.02**	3.88***	3.88***	3.88***	3.88***	3.88***
ResHigh	ResLow	1.93	2.13	2.13	2.13	2.13	2.13
ResHigh	ResOther	0.68	1.37	1.37	1.37	1.37	1.37
ResLow	ResOther	-1.58	-1.24	-1.24	-1.24	-1.24	-1.24

4.4 Discussion

4.4.1 Ecosystem services across land use

Though not statistically significant, among the 6 land use classes in this research, residential zones have higher average and median quadrat carbon sequestration than the others (Table 13 and Table 14). A similar result was found in Roanoke of Virginia that per hectare carbon storage and annual carbon sequestration in the residential area is higher than that in the commercial area (Kim et al., 2015). A potential reason is that the quadrats of residential zones tend to have more trees or higher LAI than the other land use classes do. On the other hand, surprisingly, at single-tree level, Com and Ind zones have higher average carbon sequestration than ResHigh and ResLow zones, which is opposite to that at quadrat level. To test the reason for the difference, I compared the urban forest structural indexes in this study. The related indexes were found to be significantly different among the land use (Table 18). The further post hoc comparison found that DBH, LAI, and number of trees at the quadrat level in residential zones is significantly higher than those in Com zone and Ind zone. Whereas, at the single-tree level, DBH is higher in Com and Ind zones than ResLow zone.

Table 18. Chi-square statistics of comparison of structure metrics across land use classes by Kruskal-Wallis rank sum test (the level of significance is denoted by asterisks: no asterisk, $p \geqslant 0.05$; *, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$).

Scale	DBH	LAI	Number of trees
Quadrat level	11.99*	32.74***	30.20***
Single-tree level	19.20**	47.66***	-

Different from carbon sequestration, air pollutants removal and runoff reduction represent more local ecosystem services. At the quadrat level, though not statistically significant, the average and median value of air pollutants removal are higher in residential zones, followed by Ind or ComNbr, and the minimum value is in Com zone (Table 13 and Table 14). However, due to a higher pollutants emission in Com zone and Ind zone, those results might suggest a mismatch of ecosystem services supply and demand. An improvement of air pollutants removal services in commercial and industrial areas can potentially benefit public health. Yet, it should also be noted that since the air pollutants data collected by the monitors distributed in the city was aggregated for the simulation, which leads to uncertainties of the results. The air pollutants removal services in Com zone and Ind zone might be thus underestimated.

The species-specific analysis results suggest that the solution to promote urban ecosystem services should be species-specific rather than one-size-fits-all. For instance, as ubiquitous species, *Ligustrum lucidum* might be less managed thus has higher ecosystem services in Ind zone; on contrast, it has lower ecosystem services in ResLow zone possibly due to intense pruning considering residents' aesthetics. A further species-specific study of the underlying causal relationship will enhance a more targeted urban plant management.

A limitation of the comparison should be addressed in which the concept of land use could vary among the research. Land use represents the actual practice and intended use of economic and cultural activities of a certain place, which is driven by both biophysical and socio-economic factors (Turner et al., 1995). Land use classification systems are distinguished regarding the scale and purpose of their development (Briassoulis, 2019). In our study, "land use" is a classification under City Planning Law of Japan that describes the potential use and limitation of building types of an area of land; while some other classification systems may emphasize more on actual use of the land.

4.4.2 Impact of scale

An implicit assumption is that ecological processes remain consistent across different extents and grain (Uchida et al., 2021), while the convenient assumption usually fails because of the complexity of ecological systems, especially under the high heterogeneity of urban context (Pickett et al., 2017).

Ecological patterns vary across the scale, so is the urban ecosystem service pattern (Pickett et al., 2017; Uchida et al., 2021). Grain is one component of the scale concept (Wu and Li, 2006). Quadrat and single-tree levels represent two different levels of grain in our study. No significant difference was found among quadrat ecosystem services across land use classes, while variances were detected at the single-tree level for all ecosystem services (Table 12). These results support the theoretical conclusion that variance between sample quadrats generally decreases with the increasing of quadrat grain (Wiens, 1989). The results probably indicate that the variance of ecosystem services at the single-tree scale is “averaged” at the quadrat scale. Compared to single-tree level, the ecosystem services at quadrat level are affected by more factors including species, size, and the number of trees, which are further determined by the characteristics related to land use types such as available space and management intensity.

4.4.3 Comparison of ecosystem services between cities

To compare our results with other studies, the mean value of carbon storage, carbon sequestration, and runoff reduction by land use classes on a per hectare-of-land basis were calculated (Table 19). Both carbon storage (11.51-17.41 ton carbon per hectare) and annual carbon sequestration (1.35-1.60 ton carbon per hectare) of residential zones in this study are lower than that in Roanoke, Virginia, the U.S. (37°16′N 79°56′W; 37.00 and 2.28 ton carbon per hectare), while those ecosystem services efficiency of the industrial zone of this study (9.95 and 1.19 ton carbon per hectare) is higher than that of Roanoke (7.31 and 0.48 ton carbon per hectare) (Kim et al., 2015). Besides, annual carbon sequestration of residential zones is higher in this research than that of a study for Barcelona, Spain (0.35 and 1.33 ton carbon per hectare for high-density and low-density residential areas) (Baro et al., 2014). The annual runoff reduction ranges in this research was from 16.71 to 56.88 m^3/ha, which is similar to a study in green spaces of Luohe, China (24.5 - 51.1 m^3/ha) (Song et al., 2020).

Table 19. Ecosystem services efficiency in Kyoto City.

Land use	Carbon storage (ton/ha)	Carbon sequestration (ton/ha/year)	Runoff reduction (m3/ha/year)
ResLow	11.51	1.47	36.08
ResHigh	12.82	1.60	42.48
ResOther	17.41	1.35	56.88
Ind	9.95	1.19	33.35
ComNbr	12.45	1.21	32.38
Com	6.99	0.81	16.71

To compare the results of air pollutants removal with other research, the results of this study were converted into grams per year per square meter tree cover (Table 20). The results in Kyoto City is comparable to those of a study in Strasbourg city, France (NO_2: 0.92 $g/year/m^2$, O_3: 3.73 $g/year/m^2$, $PM_{2.5}$: 0.30 $g/year/m^2$) except for the result of SO_2 removal (0.07 $g/year/m^2$) (Selmi et al., 2016).

Table 20. Average air purification efficiency of Kyoto City ($g/year/m^2$).

Land use	NO_2 removal	O_3 removal	$PM_{2.5}$ removal	SO_2 removal
ResLow	1.01	3.21	0.21	0.42
ResHigh	1.10	3.53	0.23	0.47
ResOther	1.00	3.16	0.20	0.42
Ind	1.04	3.36	0.22	0.44
ComNbr	1.07	3.46	0.23	0.45
Com	0.71	2.35	0.16	0.31

It should be noted that the raw data distribution and sampling method may differ among these studies thus confounding the comparison of the results. For instance, the research in Luohe, China focuses on green space rather than random sample quadrats over the city (Song et al., 2020); and though also presents the results of quadrat ecosystem services in different land use, the study in Roanoke, Virginia mainly focuses on urban vacant (Kim et al., 2015). Furthermore, the quadrats with no woody plant are usually excluded for analysis, which also brings bias to the comparison of the results. The workbook of i-Tree Eco (2019) has provided a solid base for standard workflow in field investigation, but the difference mentioned above still highlights the barrier in multi-city research. Besides, the results of air pollutants removal are strongly affected by local air quality. Under the same condition, air pollutants removal is higher in the area with higher air pollutants concentrations.

4.4.4 Ecosystem service evaluation in cities

In the prevalent land use/land cover-based methodology of ecosystem service evaluation, the per unit value of the urban area is usually considered constant (e.g. see Arowolo et al., 2018;

Yi et al., 2017). Though no significant difference was detected for quadrat ecosystem services across land use classes in this research, it could be a result of the high heterogeneity of urban ecosystems rather than homogeneity. Land use is a rough classification under the context of law and policy in Japan. Within a certain land use type, there can be several onsite land cover classes, and quadrat ecosystem services of certain land use classes could be ‘averaged’ by this mixture. The relationship between land use or land cover and ecosystem services requires further research based on high-resolution geographic data and sampling at a different scale.

Another reason for further enrichment of the urban ecosystem services benchmark database is the heterogeneity between cities. As the comparison of our results with other studies in the previous section shows, per unit ecosystem services could vary across different cities. This comparison indicates that the local context must be considered when evaluating urban ecosystem services.

4.5 Conclusions

The main purpose of this chapter is to demonstrate the link between heterogeneity of a city and urban ecosystem services, and the potential contribution of dispersed green to urban ecosystem services. This study captured the structure, ecosystem services, and monetary values of the urban forest in Kyoto City. For urban forest structure analysis, Ind zone has more mature trees with higher DBH than the other land use; residential zones have a higher proportion of trees with larger LAI. The comparison across land use classes shows that ecosystem services are different across land use at the single-tree level, though no significant difference was detected at the quadrat level. The results indicate that the comparison varies with scale. Though less addressed in previous research and not statistically significant, the residential zones have higher average and median ecosystem services values than the other land use. The result suggests a potentially important contribution of dispersed green space (like private yards) to urban ecosystem services. For a more comprehensive and precise evaluation of ecosystem services of urban forests, further research considering heterogeneity, scale effect, and varieties of green space type is needed.

The results also provided insight for practice. I identified a mismatch of air pollutants removal and emission across land use types that the air cleaning services of urban forests in commercial areas should be improved. Furthermore, a species-specific method can help in making urban planning aimed at increasing ecosystem services.

Chapter 5 Link Biodiversity and Ecosystem Services

5.1 Introduction

The research subfield of the link between biodiversity and ecosystem services (BES) / ecosystem function (BEF) goes back to 1990s. Though the cascade concept model, describing the “cascade” from ecosystem functions, to ecosystem services, then to benefits and values, distinguishes the two concepts (“function” and “services”) (Potschin and Haines-Young, 2017), there might be a risk of unnecessary complication of the straightforward definition of ecosystem services (Costanza et al., 2017). Therefore, hereafter, I only use the term “ecosystem service(s)”. Before 1990s, both biodiversity and ecosystem services are regarded to response to environment, and biodiversity was thought to be affected by ecosystem function. When research interest on the impacts of species loss on ecosystem services emerged from 1992 Earth Summit in Rio de Janeiro, many theories and experiments have been proposed and conducted (van der Plas, 2019). Two mechanisms have been widely applied in BES research: complementarity effect and selection effect (Loreau and Hector, 2001). Complementarity effect states the utility of resource by diverse species (Tilman et al., 1997) and the positive interaction between species (Bertness and Leonard, 1997). Selection effect emphasizes the impact of identity of certain species on ecosystem services. A positive BES link is supported by a wide range of studies based on experiments (e.g., see Isbell et al., 2011; Lange et al., 2015; Verheyen et al., 2016). However, the BES experiments mainly focus on the single direction impact of biodiversity on ecosystem service, while minimum the effect of environment, which leads to bias of the conclusion compared to BES in the real world (Genung et al., 2020). Therefore, the third paradigm rises recently to embrace the complexity of the real system, that ecosystem service is regarded to be the function of environment and biodiversity.

Despite a rapid growth of this research topic, most of the studies are for non-urban context. Among the 258 articles review by Van der Plas (2019), less than 10 come from urban ecosystems. Considering that positive BES is a common motivation for urban biodiversity research and conservation, especially with green infrastructure projects becoming prevalent over the world, it is necessary to understand the true BES link in cities.

BES in cities can be different comparing to that of non-urban context, with the impact of human socio-economic factors (Schwarz et al., 2017). Firstly, urban ecosystem differs from “natural ecosystems”, in abiotic environment. It is a novel ecosystem with harsh environmental factors including higher temperature, less impervious surface, limited and fragmented habitat, and dominant by human. The novel system shapes the biological assemblage and process within it, even evolution (Johnson and Munshi-South, 2017). The community in cities are always characterized with less complex food web (Faeth et al., 2005). Short plants or individuals with more seeds tend to succeed. Or may have a higher proportion of wind-pollinated plants, or plants dispersed by animals (Knapp, 2010b). Second, the habitats in cities are limited by impervious area and plantable area, which directly influence biodiversity patterns (Figueroa et al., 2018). Furthermore, the top-down effect of urban planning determines the pattern of organism in a city with the impact of socio-economic factors (Hope et al., 2003). And residential preference also has an impact on the flora pattern in a bottom-up manners (Avolio et al., 2015). Besides, from the perspective of methodology, similar to field-observation-based BES research, it is challenging to separate the effect of environment and biodiversity on ecosystem services. Even only focusing on the effect of biodiversity, the link between ecosystem services and different dimensions of biodiversity (species diversity like richness, functional diversity, etc.) is confounding.

The evidence of the direction of BES is unclear in cities despite an increasing attention on urban biodiversity and urban ecosystem services. Ziter (2016) reviewed 77 articles with 133 assessment on urban ecosystem services in 2016 and found that only 47 considered BES link. In most of the 47 assessments, BES is reported non-correlative. And urban ecosystem services are considered as being more relevant to species composition, functional traits, or structures (24 assessments), rather than on the magnitude of a given biodiversity metric. Based on Ziter's review (2016), Schwarz et al. (2017) reviewed 994 mentions of urban BES in 317 studies. Only 24% of the mentions examine and BES link empirically, among which only half (52%) demonstrated positive relationship. According to their counting-based study, the relationship between BES varies with the metrics used and services.

In this chapter, based on the sampling investigation in Kyoto city, an analysis of BES with the urban woody plants data set is conducted. Biomass is used as the surrogate of ecosystem services in the subsequent analysis, since it is related to many key ecosystem services in cities, like carbon storage, flood mitigation, and noise reduction. Ecosystem services is considered as the function of both environment and biodiversity in this study. Thus, the factors of both perspectives are used as predictive variables. Besides, it should be emphasized that, since the causal relationship between environment, biodiversity, and ecosystem services is still very unclear for now (especially for that in cities), I do not intend to uncover the mechanism underlying urban BES. The main purpose is to explore the basic statistical relationship of urban BES.

5.2 Method

5.2.1 Data preparation

Among the 174 accessible quadrats in the investigation, trees were recorded in 151 quadrats. In this study, the sub data set of trees was extracted from the complete data set for the subsequent analysis. The attributes of each tree in the subset including scientific name of the species, DBH, height, and health status. The biomass of each tree was calculated with i-Tree Eco, as stated in Chapter 4. The single-tree data was then grouped by quadrat ID into a quadrat data set. For example, the total biomass of each quadrat was calculated as the sum of biomass of each tree in the quadrat.

Five indexes related to diversity and abundance in each quadrat were calculated, including: (1) abundance: the number of trees, (2) species richness (S; the number of species recorded in each quadrat), (3) Shannon index ($H' = -\sum_{i=1}^{S} p_i \ln p_i$, where p_i is the quadrat's proportional number of trees or the proportional area of shrubs of the constituent species i), (4) Simpson's index of diversity ($D = 1 - \sum_{i=1}^{S} p_i^2$), and (5) Pielou's evenness index ($J = H'/\ln(S)$).

The other impact factors are derived from the data set used in Chapter 3, including: distance to city center, land price, population density, the proportion of area for each land.

5.2.2 Data analysis

Multiple linear regression model was applied to test the association of the environmental factors, biodiversity indexes, and biomass. Environmental factors include: (1) urban gradient factors: distance to city center, land price, population density, and (2) the proportion of area for each land cover: agriculture, water/wetland, park, temple/shrine, cemetery, vacant, golf course, residential, multi-family residential, institutional, transportation, commercial neighbor, commercial/industrial. All subset regression was used to select the "best" model with a certain set of predictive variables. The $regsubset$ function of $leap$ package was applied to perform the analysis. The same data analysis process is then applied to sub-data set of different land use types. The same data analysis process was then applied to the subset by land use types to select the "best" model for each land use. Here, land use types are aggregated to residential (ResLow, ResHigh, and ResOther), industrial (Ind), and commercial (Com, and ComNbr).

5.3 Results

The results of linear regression are shown in Table 21. For all land use results, biomass is best predicted by land cover factors and biodiversity indexes. Among the urban gradient factors, though land price is selected as one of the predictive variables in the final model, it has no significant impact on biomass. Abundance and Shannon index are positively correlated to biomass, but richness is negatively correlated to biomass.

The results for residential area are similar to that of all land use results, that biomass is also best predicted by land cover factors and biodiversity indexes. For industrial area, only the positive effect of abundance is found to be significant. The effects urban gradient factors are only found to be significant for commercial area, that both distance to city center and population density have a negative effect, while land price has a negligibly positive effect. Other than that, land cover factors (proportion of temple/shrine, multi-family residential building, neighborhood commercial building, commercial/industrial building) and biodiversity indexes (abundance, richness, and evenness) are positively correlated with biomass for commercial data set.

Table 21. Multiple regression analyses for environmental factors and biodiversity indexes to biomass of tree data set. The variables of proportion of each land cover are denoted with the name of the land cover and "%" in the explanatory variable column. Significance is indicated as: *p < 0.05, ** p < 0.01, *** p < 0.001.

Predictive variable	All land use	Residential	Industrial	Commercial
Distance to city center	NA	-0.07	NA	-0.64**
Land price	0	NA	0.01	0.00**
Population density	NA	NA	NA	-2.01**
Agriculture%	NA	875.17	NA	NA
Water/wetland%	NA	NA	NA	NA
Park%	4925.11***	5007.24***	NA	NA
Temple/shrine%	1874.06**	2725.75*	-235.48	6805.87***
Cemetery%	NA	NA	NA	NA
Vacant%	8666.05***	16309.45***	3207.99	NA
Golf course%	NA	NA	NA	NA
Residential%	NA	NA	NA	NA
Multi-family residential%	NA	NA	-1455.81	1176.19***
Institutional%	1365.19*	2000.43**	NA	NA
Transportation%	NA	NA	NA	NA
Commercial neighbor%	NA	NA	-16602.7	7969.93***
Commercial/industrial%	1541.43*	2425.53*	NA	NA
Abundance	100.51***	67.92**	414.38**	134.62***
Richness	-248.65*	NA	-699.42	262.44***
Shannon index	1026.06*	NA	NA	NA
Simpson index	NA	NA	-1359.76	NA
Evenness	NA	1277.21	11020.63	15275.89***

5.4 Discussion

5.4.1 The impact of environment and biodiversity on ecosystem function

Biomass, as the surrogate of ecosystem services in this study, is best predicted by land cover factors and biodiversity indexes, while urban gradiant factors are only shown in the "best" model for commercial area. Among most of the data sets, the land cover with potentially higher habitat area, including parks vacant, and shrines/temples, are positively correlated with biomass. However, it is difficult to explain the positve correlation between the proportion of commercial/industrial area and biomass. The reason for the later result could be due to the function-oriented monoculture in the quadrats with higher proportion of commercial/industrial area. The sub data set of quadrat with commercial/industrial land cover was extracted (n = 11) and a linear model was adapted. The result, though not significant, shows a negative correlation between the proportion of the commercial/industrial area and richness (estimate statistic = -5.73, p-value = 0.06).

Among the biodiversity indexes, though richness and Shannon indexes are shown in the "best" model for some data sets, abundance is a more predictive variable. An explanation regarding the impact of abundance is more individual hypothesis (Yee and Juliano, 2007), which states that with the increase of abundance, the risk of species extinction decreases. Thus, abundance has a positive impact on richness and ecosystem functions. In the dataset of this research, a correlation between abundance and richness was also found with a linear model (estimate statistic = 0.25, p-value < 0.01). Another critical relative factor is relative abundance and the identity of the species. Most species with lower abundance tend to have lower function in our dataset (Figure 13). Thus, the effect of species loss between higher function quadrat and lower function quadrat depends on the turnover rate of the species. If the turnover rate of the higher-abundance species is also higher, then it results into a larger variation of function.

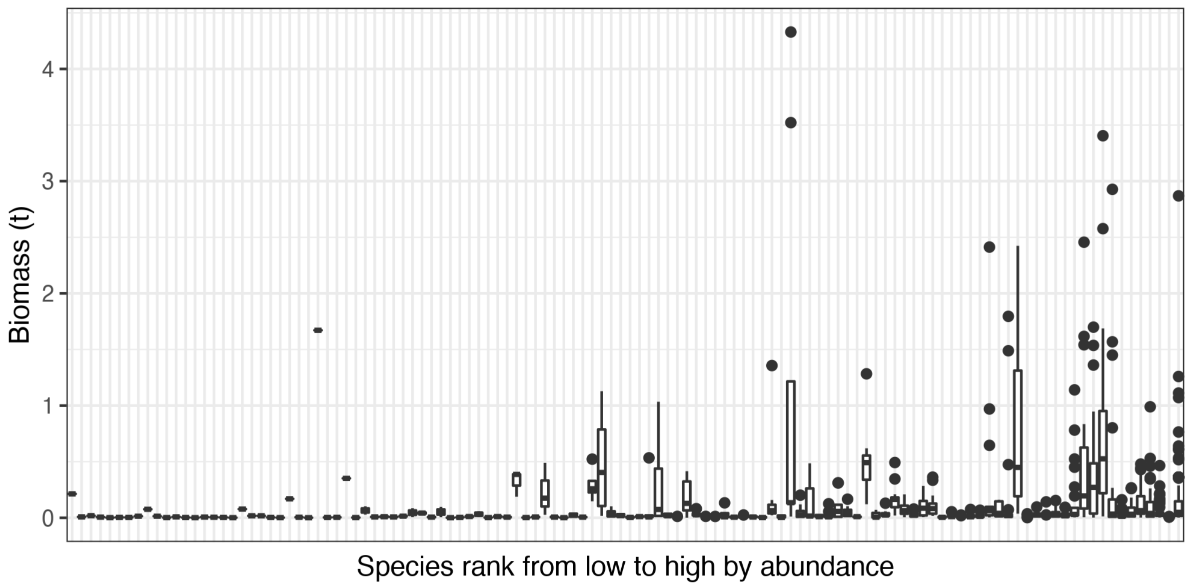

Figure 13. Biomass range of each species along abundance order.

5.4.2 Methodology for urban BES research

For BES research in urban ecosystems, there is a paucity of empirical study with cause-effect relationship statistical analysis. In this sense, certain statistical method like structural equation modelling can be useful, since it can separate the effects of impact factors into different layers with a hierarchical hypothesis. For instance, though richness or other diversity metrics have impacts on ecosystem function, they are also affected by other factors of an urban site including available greening space, land use, and human management. Besides, there is also a lack of BES experimental studies in cities. Schwarz et al. (2017) reviewed 337 articles with 228 tested BES relationship on urban BES research and found that only 30% are tested with experiments, and the proportion become even smaller for urban woody plants.

A major bottle neck of studying BES in urban ecosystems is that researchers have lacked analytical tools. For nature ecosystems research, real-world differs from experiments in three ways: first, the community in experiments are generally with lower dominance; second, aggregate abundance in experiment is controlled; third, BES varies with spatial scale and temporal scale while the scales for experiments are limited (Winfree et al., 2015). Considering that differences, it is difficult to separate the effect of different component of biodiversity (richness, composition, and abundance) on ecosystem functions. The analytical innovation, ecological Price equation, was developed to solve the problem with field investigation data (Fox, 2006). Ecological Price equation can divide the effects of richness, species identification and other context (e.g., abundance). Though it was for partioning the drivers of microevolutionary change at the very first (Price, 1972), it was developed for biodiversity – function analysis (Fox, 2006). With ecological Price equation, the change of function between a pair of sites can be divided into three additive parts: richness effect, which indicates that the higher function is simply resulted from more species; composition effect, which indicates that higher function site has species with higher function on average; and context dependence effect, which indicate that the contribution of species presenting at both site is higher in the higher function site.

However, the applicability for that method to urban ecosystems is still debatable. Urban ecosystems differ from “natural ecosystems” and related experiments in many ways. First, similar to “natural ecosystems”, cities is characterized with high species turnover rate – it is even larger than natural ecosystems regarding woody plants, since the species richness of woody plants is generally higher than that in natural ecosystems (Pearse et al., 2018). Second, the community of some land use in cities (e.g., residential yard) is generally with lower dominance while some are not (e.g., industrial area). Third, since the available space in cities is limited, it is difficult to collect data systematically from quadrats with controlling the other factors. The most critical point is that the condition for applyting the method or the theories on

BES is hardly fullfilled in a city. Complementarity effect and selection effect all assume interactions between organisms. However, the tree density in a city is generally too low to achieve the inter-species or inter-organism interaction.

5.5 Conclusion

Biomass is best predicted by land cover factors and biodiversity-related indexes, rather than urban gradient factors. The proportion of land cover with higher available habitat are generally positively correlated with biomass. Among the biodiversity-related indexes, abundance is a better predictive variable compared with the other indexes like richness and Shannon index. A possible explanation is that the tree density is too low that the organisms or species do not interact intensively with each other. The tools for urban BES analysis need further study.

Chapter 6 Discussion and Conclusions

A series of research were conducted for this thesis to understand the pattern and scale of urban biodiversity and urban ecosystem services, and their linkage. To this end, data on woody plants were collected with sampling quadrat across the built-up area of Kyoto City. The following objectives were emphasized for this thesis: (1) What is the pattern of plant diversity across land use types in a city? And does the pattern vary with scale? (2) What is pattern of ecosystem services provided by urban forest across land use types in a city? Does the pattern vary with scale? (3) What is the linkage between biodiversity and ecosystem function / service under urban context? Then in the following chapters, I used ecological tools and data science to answer the questions.

6.1 Urban biodiversity

In Chapter 3, I evaluated woody plant diversity across land use and scales in Kyoto City. At land use level, residential areas had higher total richness with moderate to low overall evenness, while commercial areas had relatively lower total richness. A high species composition dissimilarity was identified between residential areas and other land use types. At quadrat level, ResLow area had higher richness than the other land use excepting for ResHigh area. Quadrat abundance and evenness were different across land use types for trees but not for shrubs. Quadrat species composition was significantly different across land use types for shrubs, but not for trees. The research identified prior land use types for biodiversity improvement. For instance, the commercial areas are characterized with lower plant diversity but higher visitors. An improvement of plant diversity in these land use types can enhance the potential ecosystem services benefit due to a higher beneficiary population and higher accessibility of the plants in public space. The contribution of ResLow area for urban biodiversity conservation was also further proved.

6.2 Urban ecosystem services

In Chapter 4, I focused on the pattern of ecosystem services across land use. The main objective is to demonstrate the link between heterogeneity of a city and urban ecosystem services, and the potential contribution of dispersed green to urban ecosystem services. This chapter captured the structure, ecosystem services, and monetary values of the urban forest in Kyoto City. The ecosystem services evaluated in this chapter includes carbon storage and sequestration, air pollutants removal, and runoff reduction. Here I would like to just focus on the ecosystem services part.

The pattern of ecosystem services was compared at both quadrat and single-tree level. The comparison shows that ecosystem services are different across land use at the single-tree level, though no significant difference was detected at the quadrat level. The results indicate that the comparison varies with scale. Though less addressed in previous research and not statistically significant, the residential zones have higher average and median ecosystem services values than the other land use. The result suggests a potentially important contribution of dispersed green space (like private yards) to urban ecosystem services. From a perspective of methodology of ecosystem services evaluation, for a more comprehensive and precise evaluation of ecosystem services of urban forests, further research considering heterogeneity, scale effect, and varieties of green space type is needed.

6.3 BES in urban ecosystems

Biomass is best predicted by land cover factors and biodiversity-related indexes, rather than urban gradient factors. The proportion of land cover with higher available habitat are generally positively correlated with biomass. Among the biodiversity-related indexes, abundance is a better predictive variable compared with the other indexes like richness and Shannon index. A possible explanation is that the tree density is too low to that the organisms or species do not interact intensively with each other.

6.4 Further research proposal

Urban biodiversity and ecosystem research, due to the complexity of cities and the the tangling effect of land scape and human activities, one of the most important target of the field is to provide insight to urban planning and management practice. To achieve this end, the next step of urban biodiversity pattern research is to study the process, the further, to model urban biodiversity. This study takes land use into consideration for two reasons. The first is that land use has a direct impact on biodiversity pattern. The second is that, based on the results of the analysis of driving factors of urban biodiversity pattern, it is reasonable to take land use and land cover as part of the predictive variables when developing an urban biodiversity model.

Regarding urban ecosystem services, since the benefit of ecosystem services not only depends on the supply of the services, but also the demand. Therefore, mapping of ecosystem services is critical for related urban planning. The research results of this study provide basic parameter for future mapping and spatial analysis of urban ecosystem services.

For the link between urban biodiversity and ecosystem services, it is essential to take the impact of urban environment into consideration. The paradigm of BES research based on experiments or under natural ecosystem context usually mainly focus on the impact of biodiversity. However, due to the low density of organisms in cities, the mechanisms of inter-species interaction do not work in the similar way as they are under non-urban context. Second, the tools for urban BES research need further development. Linear models has been widely used in the previous urban BES studies, however, it is challenging to separate the effects of different dimensions of biodiversity.

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Appendix

Table A1. Family, genus, and provenance of all the species recorded in this study

Species	Family	Genus	Provenance
Justicia brandegeeana	Acanthaceae	Justicia	native
Viburnum odoratissimum	Adoxaceae	Viburnum	exotic
Viburnum plicatum	Adoxaceae	Viburnum	exotic
Cotinus coggygria	Anacardiaceae	Cotinus	exotic
Mandevilla sanderi	Apocynaceae	Mandevilla	native
Nerium oleander	Apocynaceae	Nerium	native
Trachelospermum asiaticum	Apocynaceae	Trachelospermum	exotic
Ilex latifolia	Aquifoliaceae	Ilex	exotic
Ilex crenata	Aquifoliaceae	Ilex	exotic
Ilex rotunda	Aquifoliaceae	Ilex	exotic
Dendropanax trifidus	Araliaceae	Dendropanax	exotic
Fatsia japonica	Araliaceae	Fatsia	exotic
Hedera canariensis	Araliaceae	Hedera	native
Hedera helix	Araliaceae	Hedera	native
Schefflera heptaphylla	Araliaceae	Schefflera	exotic
Chamaedorea elegans	Arecaceae	Chamaedorea	native
Dypsis lutescens	Arecaceae	Dypsis	native
Rhapis humilis	Arecaceae	Rhapis	native
Trachycarpus fortunei	Arecaceae	Trachycarpus	native
Cordyline fruticosa	Asparagaceae	Cordyline	exotic
Dracaena marginata	Asparagaceae	Dracaena	native
Dracaena cambodiana	Asparagaceae	Dracaena	native
Begonia grandis	Begoniaceae	Begonia	native
Mahonia fortunei	Berberidaceae	Mahonia	native
Mahonia eurybracteata	Berberidaceae	Mahonia	native
Mahonia japonica	Berberidaceae	Mahonia	native
Nandina domestica	Berberidaceae	Nandina	exotic
Buxus microphylla	Buxaceae	Buxus	exotic
Sarcococca ruscifolia	Buxaceae	Sarcococca	native
Schlumbergera truncata	Cactaceae	Schlumbergera	native
Chimonanthus praecox	Calycanthaceae	Chimonanthus	native
Aphananthe aspera	Cannabaceae	Aphananthe	exotic
Celtis sinensis	Cannabaceae	Celtis	exotic
Euonymus japonicus	Celastraceae	Euonymus	exotic
Euonymus alatus	Celastraceae	Euonymus	exotic
Cercidiphyllum japonicum	Cercidiphyllaceae	Cercidiphyllum	exotic

Species	Family	Genus	Provenance
Sarcandra glabra	Chloranthaceae	Sarcandra	exotic
Dendranthema morifolium	Compositae	Dendranthema	native
Osteospermum ecklonis	Compositae	Osteospermum	native
Cornus kousa	Cornaceae	Cornus	exotic
Cornus officinalis	Cornaceae	Cornus	exotic
Cornus florida	Cornaceae	Cornus	native
Cornus hongkongensis	Cornaceae	Cornus	native
Crassula ovata	Crassulaceae	Crassula	native
Chamaecyparis obtusa	Cupressaceae	Chamaecyparis	exotic
Chamaecyparis pisifera	Cupressaceae	Chamaecyparis	exotic
Cryptomeria japonica	Cupressaceae	Cryptomeria	exotic
Cunninghamia lanceolata	Cupressaceae	Cunninghamia	native
Cupressus macrocarpa	Cupressaceae	Cupressus	native
Juniperus chinensis	Cupressaceae	Juniperus	native
Juniperus scopulorum	Cupressaceae	Juniperus	native
Platycladus orientalis	Cupressaceae	Platycladus	native
Cycas revoluta	Cycadaceae	Cycas	exotic
Cyrilla racemiflora	Cyrillaceae	Cyrilla	native
Daphniphyllum macropodum	Daphniphyllaceae	Daphniphyllum	exotic
Diospyros kaki	Ebenaceae	Diospyros	exotic
Elaeagnus macrophylla	Elaeagnaceae	Elaeagnus	exotic
Elaeagnus umbellata	Elaeagnaceae	Elaeagnus	exotic
Elaeagnus multiflora	Elaeagnaceae	Elaeagnus	exotic
Elaeagnus pungens	Elaeagnaceae	Elaeagnus	exotic
Enkianthus perulatus	Ericaceae	Enkianthus	exotic
Pieris japonica	Ericaceae	Pieris	exotic
Rhododendron hirado group	Ericaceae	Rhododendron	exotic
Rhododendron indicum	Ericaceae	Rhododendron	exotic
Rhododendron kurume group	Ericaceae	Rhododendron	exotic
Rhododendron spp.	Ericaceae	Rhododendron	native
Vaccinium oldhamii	Ericaceae	Vaccinium	exotic
Mallotus japonicus	Euphorbiaceae	Mallotus	exotic
Triadica sebifera	Euphorbiaceae	Triadica	exotic
Castanea crenata	Fagaceae	Castanea	exotic
Quercus myrsinifolia	Fagaceae	Quercus	exotic
Quercus glauca	Fagaceae	Quercus	exotic
Quercus salicina	Fagaceae	Quercus	exotic
Quercus phillyraeoides	Fagaceae	Quercus	exotic
Aucuba japonica	Garryaceae	Aucuba	exotic
Gelsemium sempervirens	Gelsemiaceae	Gelsemium	native
Ginkgo biloba	Ginkgoaceae	Ginkgo	native
Corylopsis pauciflora	Hamamelidaceae	Corylopsis	exotic

Species	Family	Genus	Provenance
Distylium racemosum	Hamamelidaceae	Distylium	exotic
Loropetalum chinense	Hamamelidaceae	Loropetalum	exotic
Deutzia scabra	Hydrangeaceae	Deutzia	exotic
Hydrangea macrophylla	Hydrangeaceae	Hydrangea	exotic
Hydrangea quercifolia	Hydrangeaceae	Hydrangea	native
Hydrangea arborescens	Hydrangeaceae	Hydrangea	native
Hypericum calycinum	Hypericaceae	Hypericum	native
Hypericum monogynum	Hypericaceae	Hypericum	native
Hypericum patulum	Hypericaceae	Hypericum	native
Callicarpa japonica	Lamiaceae	Callicarpa	exotic
Callicarpa dichotoma	Lamiaceae	Callicarpa	exotic
Lavandula angustifolia	Lamiaceae	Lavandula	native
Lavandula dentata	Lamiaceae	Lavandula	native
Mentha canadensis	Lamiaceae	Mentha	native
Rosmarinus officinalis	Lamiaceae	Rosmarinus	native
Akebia quinata	Lardizabalaceae	Akebia	exotic
Cinnamomum camphora	Lauraceae	Cinnamomum	native
Laurus nobilis	Lauraceae	Laurus	native
Lindera lancea	Lauraceae	Lindera	exotic
Persea americana	Lauraceae	Persea	native
Acacia baileyana	Leguminosae	Acacia	native
Albizia julibrissin	Leguminosae	Albizia	native
Caragana sinica	Leguminosae	Caragana	native
Cercis chinensis	Leguminosae	Cercis	native
Lespedeza thunbergii	Leguminosae	Lespedeza	native
Styphnolobium japonicum	Leguminosae	Styphnolobium	native
Wisteria floribunda	Leguminosae	Wisteria	exotic
Lagerstroemia indica	Lythraceae	Lagerstroemia	exotic
Lagerstroemia subcostata	Lythraceae	Lagerstroemia	exotic
Punica granatum	Lythraceae	Punica	native
Liriodendron tulipifera	Magnoliaceae	Liriodendron	native
Magnolia kobus	Magnoliaceae	Magnolia	exotic
Magnolia liliiflora	Magnoliaceae	Magnolia	exotic
Magnolia grandiflora	Magnoliaceae	Magnolia	native
Magnolia denudata	Magnoliaceae	Magnolia	native
Abutilon megapotamicum	Malvaceae	Abutilon	native
Hibiscus syriacus	Malvaceae	Hibiscus	native
Hibiscus mutabilis	Malvaceae	Hibiscus	native
Paris polyphylla	Melanthiaceae	Paris	native
Tibouchina urvilleana	Melastomataceae	Tibouchina	native
Melia azedarach	Meliaceae	Melia	exotic
Toona sinensis	Meliaceae	Toona	native

Species	Family	Genus	Provenance
Broussonetia papyrifera	Moraceae	Broussonetia	exotic
Ficus erecta	Moraceae	Ficus	exotic
Ficus microcarpa	Moraceae	Ficus	exotic
Ficus carica	Moraceae	Ficus	native
Morus alba	Moraceae	Morus	native
Myrica rubra	Myricaceae	Myrica	exotic
Callistemon citrinus	Myrtaceae	Callistemon	native
Eucalyptus cinerea	Myrtaceae	Eucalyptus	native
Eucalyptus globulus	Myrtaceae	Eucalyptus	native
Myrtus communis	Myrtaceae	Myrtus	native
Mirabilis jalapa	Nyctaginaceae	Mirabilis	native
Forsythia viridissima	Oleaceae	Forsythia	native
Fraxinus griffithii	Oleaceae	Fraxinus	native
Jasminum mesnyi	Oleaceae	Jasminum	native
Jasminum nudiflorum	Oleaceae	Jasminum	native
Ligustrum japonicum	Oleaceae	Ligustrum	exotic
Ligustrum lucidum	Oleaceae	Ligustrum	native
Ligustrum sinense	Oleaceae	Ligustrum	native
Olea europaea	Oleaceae	Olea	native
Osmanthus heterophyllus	Oleaceae	Osmanthus	exotic
Osmanthus fragrans	Oleaceae	Osmanthus	exotic
Syringa reticulata	Oleaceae	Syringa	native
Paeonia suffruticosa	Paeoniaceae	Paeonia	native
Paeonia lactiflora	Paeoniaceae	Paeonia	native
Cleyera japonica	Pentaphylacaceae	Cleyera	exotic
Eurya emarginata	Pentaphylacaceae	Eurya	exotic
Eurya japonica	Pentaphylacaceae	Eurya	exotic
Ternstroemia gymnanthera	Pentaphylacaceae	Ternstroemia	exotic
Phytolacca japonica	Phytolaccaceae	Phytolacca	exotic
Abies firma	Pinaceae	Abies	exotic
Cedrus deodara	Pinaceae	Cedrus	native
Picea glehnii	Pinaceae	Picea	exotic
Pinus densiflora	Pinaceae	Pinus	exotic
Pinus thunbergii	Pinaceae	Pinus	exotic
Tsuga sieboldii	Pinaceae	Tsuga	exotic
Pittosporum tobira	Pittosporaceae	Pittosporum	exotic
Ceratostigma plumbaginoides	Plumbaginaceae	Ceratostigma	native
Podocarpus macrophyllus	Podocarpaceae	Podocarpus	exotic
Muehlenbeckia axillaris	Polygonaceae	Muehlenbeckia	native
Reynoutria japonica	Polygonaceae	Reynoutria	exotic
Ardisia crenata	Primulaceae	Ardisia	exotic
Cerasus jamasakura	Rosaceae	Cerasus	exotic

Species	Family	Genus	Provenance
Chaenomeles speciosa	Rosaceae	Chaenomeles	native
Chaenomeles sinensis	Rosaceae	Chaenomeles	native
Eriobotrya japonica	Rosaceae	Eriobotrya	exotic
Kerria japonica	Rosaceae	Kerria	exotic
Photinia glabra	Rosaceae	Photinia	exotic
Prunus spachiana	Rosaceae	Prunus	exotic
Prunus subhirtella	Rosaceae	Prunus	exotic
Prunus mume	Rosaceae	Prunus	native
Prunus armeniaca	Rosaceae	Prunus	native
Prunus persica	Rosaceae	Prunus	native
Pyracantha coccinea	Rosaceae	Pyracantha	native
Rhaphiolepis indica	Rosaceae	Rhaphiolepis	native
Rhodotypos scandens	Rosaceae	Rhodotypos	exotic
Rosa chinensis	Rosaceae	Rosa	native
Rosa banksiae	Rosaceae	Rosa	native
Rosa hybrida	Rosaceae	Rosa	native
Rubus trifidus	Rosaceae	Rubus	exotic
Rubus spectabilis	Rosaceae	Rubus	native
Sanguisorba officinalis	Rosaceae	Sanguisorba	exotic
Spiraea thunbergii	Rosaceae	Spiraea	exotic
Spiraea japonica	Rosaceae	Spiraea	exotic
Spiraea cantoniensis	Rosaceae	Spiraea	native
Gardenia jasminoides	Rubiaceae	Gardenia	exotic
Ixora chinensis	Rubiaceae	Ixora	native
Mussaenda pubescens	Rubiaceae	Mussaenda	native
Serissa japonica	Rubiaceae	Serissa	native
Citrus limon	Rutaceae	Citrus	exotic
Citrus junos	Rutaceae	Citrus	native
Citrus reticulata	Rutaceae	Citrus	native
Citrus japonica	Rutaceae	Citrus	native
Zanthoxylum bungeanum	Rutaceae	Zanthoxylum	exotic
Salix futura	Salicaceae	Salix	exotic
Salix babylonica	Salicaceae	Salix	native
Acer palmatum	Sapindaceae	Acer	exotic
Acer pictum	Sapindaceae	Acer	exotic
Acer japonicum	Sapindaceae	Acer	exotic
Acer buergerianum	Sapindaceae	Acer	native
Aesculus turbinata	Sapindaceae	Aesculus	exotic
Brucea javanica	Simaroubaceae	Brucea	native
Brunfelsia australis	Solanaceae	Brunfelsia	native
Styrax japonicus	Styracaceae	Styrax	exotic
Symplocos sumuntia	Symplocaceae	Symplocos	exotic

Species	Family	Genus	Provenance
Taxus wallichiana	Taxaceae	Taxus	native
Camellia japonica	Theaceae	Camellia	exotic
Camellia sasanqua	Theaceae	Camellia	exotic
Camellia sinensis	Theaceae	Camellia	exotic
Camellia rusticana	Theaceae	Camellia	exotic
Stewartia pseudocamellia	Theaceae	Stewartia	exotic
Daphne odora	Thymelaeaceae	Daphne	exotic
Ulmus parvifolia	Ulmaceae	Ulmus	exotic
Zelkova serrata	Ulmaceae	Zelkova	exotic
Boehmeria nivea	Urticaceae	Boehmeria	native
Pilea cadierei	Urticaceae	Pilea	native
Duranta erecta	Verbenaceae	Duranta	native
Lantana camara	Verbenaceae	Lantana	native
Vitis vinifera	Vitaceae	Vitis	native
Prunus × yedoensis	-	-	exotic
Osmanthus × fortunei	-	-	native
Citrus × aurantium	-	-	native
Abelia × grandiflora	-	-	native
Pelargonium × hortorum	-	-	native

Table A2. Pairwise product-moment correlations between measures of plant density and species diversity indexes in the plots.

Significance is indicated as: *p < 0.05, ** p < 0.01, *** p < 0.001.

	Richness	Shannon index	Simpson index	Evenness
Trees				
Number of trees	0.68***	0.53***	0.42***	-0.63***
Richness		0.91***	0.77***	-
Shannon index			0.95***	0.23*
Simpson index				0.44***
Shrubs				
Area of shrubs	0.44***	0.32***	0.27**	-0.15*
Richness		0.89***	0.74***	-
Shannon index			0.96***	0.49***
Simpson index				0.68***

Table A3. Main parameters of i-Tree Eco input for this research.

The superscript: "a": customized in this study; "b": replaced with local parameter value via i-Tree Database).

Ecosystem service	Parameter	Value/ID/Monitor	Data Year	Reference
Carbon Storage/ Sequestration	Social cost of carbon [a]	96 US$/ton carbon	2019	Japan Ministry of the Environment (2019)
Air Pollutant Removal	Leaf-on date [b]	April 4th	1981–2010	Japan Meteorological Agency (2021)
	Leaf-off date [b]	November 18th	1981–2010	Japan Meteorological Agency (2021)
	Upper air monitor ID [a]	47778: Shionomisaki	2015	Earth System Research Laboratory(2020)
	Solar radiation monitor ID [a]	26104060: Mibu	2015	National Institute for Environmental Studies (2021)
	Net radiation monitor ID [a]	28204150: Hamakoushien	2015	National Institute for Environmental Studies (2021)
	Precipitation monitor ID [b]	28214010: Yoriaihiroba	2015	National Institute for Environmental Studies (2021)
	CO concentration monitor ID [b]	26104510: Jihaioomiya	2010–2015	National Institute for Environmental Studies (2021)
		26107510: Jihaiminami	2010–2015	National Institute for Environmental Studies (2021)
	NO_2 concentration monitor ID [b]	26101010: Kita	2010–2015	National Institute for Environmental Studies (2021)
		26102510: Jihaikamigyou	2010–2015	National Institute for Environmental Studies (2021)
		26103010: Sakyou	2010–2015	National Institute for Environmental Studies (2021)
		26104010: Kyoutoshiyakusho	2010–2015	National Institute for Environmental Studies (2021)
		26104060: Mibu	2010–2015	National Institute for Environmental Studies (2021)
		26104510: Jihaioomiya	2010–2015	National Institute for Environmental Studies (2021)
		26107510: Jihaiminami	2010–2015	National Institute for Environmental Studies (2021)
	O_3 concentration monitor ID [b]	26101010: Kita	2010–2015	National Institute for Environmental Studies (2021)
		26103010: Sakyou	2010–2015	National Institute for Environmental Studies (2021)
		26104010: Kyoutoshiyakusho	2010–2015	National Institute for Environmental Studies (2021)
		26104060: Mibu	2010–2015	National Institute for Environmental Studies (2021)
	$PM_{2.5}$ concentration monitor ID [b]	26102510: Jihaikamigyou	2010–2015	National Institute for Environmental Studies (2021)
		26104010: Kyoutoshiyakusho	2010–2015	National Institute for Environmental Studies (2021)
		26104060: Mibu	2010–2015	National Institute for Environmental Studies (2021)
		26104510: Jihaioomiya	2010–2015	National Institute for Environmental Studies (2021)
		26107510: Jihaiminami	2010–2015	National Institute for Environmental Studies (2021)

	SO_2 concentration monitor ID [b]	26104060: Mibu	2010–2015	National Institute for Environmental Studies (2021)
Human health effects	Population [b]	1,474,735	2016	Kyoto City Statistics Portal (2019)
	Medical expense [a]	46% of the US	2018	OECD (2021a)
	Household income [a]	65% of the US	-	OECD (2021b)
	Value of a statistical life [a]	3,909,090.91 US$	1991–2007	Miyazato (2010)
Avoided runoff	Surface weather [b]	477590: Kyoto	2015	National Centers for Environmental Information (2021)
	Precipitation [b]	28214010: Yoriaihiroba	2015	National Institute for Environmental Studies (2021)
	Impervious cover [a]	80.57%	2014–2016	JAXA (2021)
	Stormwater control cost [a]	7 US$/$m^3$	2007	Kawaguchi et al. (2021)

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