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# Comparison of Species Composition, Species Diversity, and Structural Distribution of Urban Trees in Three Types of Urban Greenspaces

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## ABSTRACT

*Urban greenspaces are considered as biodiversity hotspots in urban areas due to its limited presence and prevalent threat of land use conversion. The species composition and structural complexity of urban greenspaces influence the ecosystem services it can provide, and ability to adapt to environmental stresses. Hence, this study was conducted to understand how urban greenspaces were shaped by the different land uses associated to management. Fourteen urban greenspaces in the vicinity of Metropolitan Manila were selected and classified into three types – commercial greenspaces, recreational parks, and wildlife parks. These were compared based on tree species composition and diversity metrics estimated using species identification and abundance data. Tree measurement data were also used to compare the structural patterns of trees in different types of urban greenspace. Results of chi-square tests (α=0.05) showed that the proportion of native and exotic tree species and the relative abundance of threatened and non-threatened trees in both local and global scales were significantly associated with the type of urban greenspace. Significant associations also existed between the type of urban greenspace and the distribution of trees into diameter, height, and crown spread classes. Species diversity metrics and tree measurements were also significantly different across the three types of urban greenspace based on Kruskal-Wallis Test and Dunn's Multiple Pairwise Comparison. The study concluded that different types of urban greenspace are composed of distinct tree communities that may require different management strategies.*

***Keywords***: *species composition, species diversity, urban greenspace, urban tree structure*

## INTRODUCTION

The variety of tree species in urban greenspaces influences the biological, physical, and social services that the urban forest can provide. Each tree species is uniquely associated with certain organisms to which it provides food, shelter, and breeding ground. Thus, an urban greenspace that is composed of diverse species of trees can support a wide variety of flora and fauna. Moreover, the structural complexity of the urban greenspace forms various niches and habitats within the urban forest. The vertical distribution of trees into different canopy layers form distinct micro habitat for different organisms. The horizontal distribution of trees into different size classes also form different levels of shade where various understory species grow.

Each tree species possesses unique growth attributes that influence how it performs its several environmental functions. The extent of crown spread and density of a tree influence its climatic functions, while the depth and spread of a tree's root system determine the range of land form and soil environment that it can preserve and enrich. The size and density of a tree's woody parts affect its carbon sequestration potential, and the amount of litter produced by trees affect soil organic matter and nutrient cycling in urban greenspaces. Different combinations of tree species, each with unique and dynamic architectural attributes, also create diverse urban landscapes that provide various social functions. Trees with ornamental leaves, flowers, fruits, and stem provide aesthetic benefits in a landscape. Trees of various growth forms also perform different architectural uses. Century-old heritage trees in urban greenspaces are preserved to provide historical and cultural assets to the community. Thus, the selection of tree species to be planted and retained in urban areas should be guided by the target benefits from each greenspace.

Greenspace management plans should be tailored based on the species composition and structure of trees. Different tree species and tree sizes have distinct cultural requirements, and may pose various positive and negative externalities. Trees in urban greenspaces should be intensively managed to control the risk to urban dwellers and infrastructures, while maximizing the benefits that the trees can provide to the urban ecosystem.

Urban greenspaces are typically classified based on the land-use type where they are integrated (Nowak 1994). In the Peel Region, Canada, urban trees were distinguished across eight land use classes including agriculture, commercial, golf, institutional, parkland, residential, transportation, and vacant land uses (Bourne & Conway 2013). In Melbourne, Australia, the major types of urban greenspaces found were golf courses,

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local urban parks, residential neighborhoods with private gardens, and patches of remnant vegetation (Threlfall *et al*. 2016). In Guangzhou city, South China, Jim and Liu (2001) identified three main urban green space types, including roadside niches, urban parks, and institutional grounds. In Boston, street trees and park trees were recognized as the two components of public urban forests (Welch 1994).

This study compared urban greenspace patterns, such as tree species composition, species diversity, and tree structure, in different types of urban greenspace in the vicinity of Metropolitan Manila. This was to understand if the different land uses could significantly influence the characteristics of urban greenspaces. The findings of this study could be used as guide in designing management strategies that are appropriate to the type of urban greenspace.

## METHODOLOGY

### Selected Urban Greenspaces

Fourteen urban greenspaces were selected in the vicinity of Metropolitan Manila (**Figure 1**), and classified into three types – commercial greenspaces (CGs), recreational parks (RPs), and wildlife parks (WPs) (**Table 1).** CGs are those integrated in commercial establishments such as malls, courtyards, and private office buildings, while RPs include public, private, and residential parks that are intended for outdoor activities. The type of urban greenspace that may nearly approximate forest structure are WPs, which are primarily established for conservation of wildlife species, alongside various floral species.

**Table 1.** List of urban greenspaces selected for the study.

| Type or urban greenspace | Greenspace identification | Location | Land area (ha) | Year surveyed |
|---|---|---|---|---|
| Commercial greenspaces | Alabang Town Center[1] | Muntinlupa City | 17.0 | 2017 |
| | Ayala Triangle Gardens[1] | Makati City | 2.0 | 2016 |
| | Bonifacio High Street[1] | Taguig City[8] | 3.0 | 2017 |
| | Kasaysayan sa bawat oras[1] | Taguig City[8] | 0.5 | 2017 |
| | Trinoma[2] | Quezon City | 3.0 | 2017 |
| Recreational parks | AAV[3] Cuenca park[1] | Muntinlupa City | 2.0 | 2016 |
| | Burgos Circle[1] | Taguig City[8] | 0.3 | 2017 |
| | Kasalikasan[1] | Taguig City[8] | 0.3 | 2017 |
| | Luneta park[4] | City of Manila | 6.3 | 2016 |
| | Terra 28[th1] | Taguig City[8] | 0.8 | 2017 |
| | Track 30th[1] | Taguig City[8] | 0.8 | 2017 |
| Wildlife parks | Avilon zoo[5] | Rodriguez, Rizal | 7.5 | 2018 |
| | Manila zoo[6] | City of Manila | 5.5 | 2018 |
| | NAPWC[5,7] | Quezon City | 23.8 | 2016 |

[1]Tree inventory covered all trees in the site
[2]Tree inventory only covered parking areas and property boundary
[3]AAV is the acronym for Ayala Alabang Village
[4]Tree inventory restricted in the public central park
[5]Tree inventory include all trees in the park except for trees inside animal enclosures
[6]Tree inventory limited to playground area and avian dome
[7]NAPWC is the acronym for Ninoy Aquino Parks and Wildlife Center
[8]Shown in the map within the old political boundary of Makati City

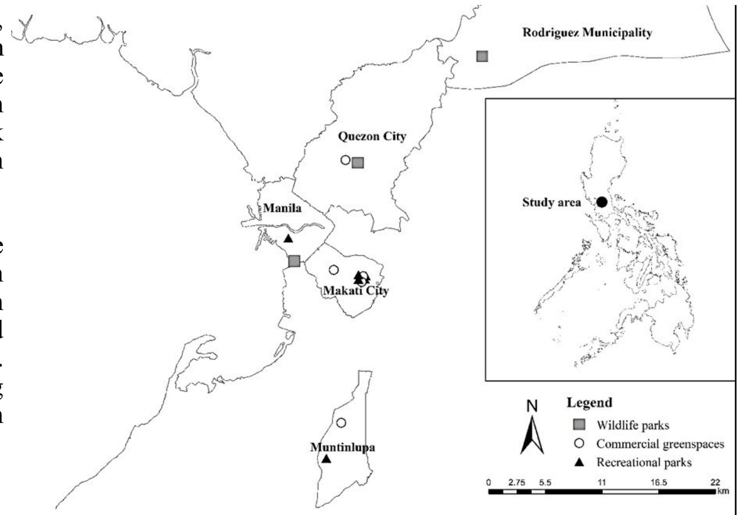



**Figure 1.** Location of study sites in the vicinity of Metropolitan Manila.

### Inventory Method

Tree inventory activities were conducted from 2016 to 2018. All trees with a diameter of 10 cm and above were identified and measured. Tree measurements included diameter-at-breast height (DBH), total height, and crown diameter in two cardinal directions.

### Inventory Data Analysis

#### *Species composition*

Species composition was analyzed based on: (1) relative species abundance, (2) relative dominance, (3) proportion of native species, and (4) proportion of threatened species. The relative species abundance of each tree species in the sampled sites were determined as the fraction of the number of trees per species and the total number of trees recorded. Relative dominance, on the other hand, was determined using the basal area of all trees of a species expressed as a percentage of the total basal area of all species.

#### *Urban greenspaces as habitat for native and threatened tree species*

To determine which urban greenspaces were serving as habitat for native tree species, each tree species was categorized into native or exotic based on their published natural distribution. Moreover, tree species were also classified as threatened or not threatened based on their local and global conservation status. The local list of threatened tree species was sourced from DENR Administrative Order 11 Series of 2017, which classifies threatened species into four categories – *critically endangered, endangered, vulnerable*, and *other threatened species*. The global conservation status of tree species was gathered from IUCN Red List of Threatened Species, with eight categories of threatened species.

#### *Species richness and diversity*

The online program, *SpadeR*, was used in estimating species richness, and diversity indices in each study site. The empirical species richness ($S_{obs}$) is the total number of species represented in a sample. However, this study used the theoretical species richness ($S_{est}$), or the statistically-estimated number of species in each urban greenspace as represented by the sample. This was to account for other unobserved species that were not covered by the inventory. Out of the nine different species richness estimators generated by *SpadeR,* the *Chao1* species richness estimate ($S_{Chao1}$) was used. This estimate considered the number of observed species, as well as the number of undetected species, with the assumption that the information on undetected species was concentrated on the low frequency counts such as species with only one (singletons) and two (doubletons) representatives (Chao 1984). Furthermore, the species richness and abundance data were used to estimate the two most common species diversity indices – Shannon-Wiener index (H) or Shannon's entropy and the Simpson's index of diversity (Simpson1/D). Both indices express higher diversity with higher values. The empirical index of Shannon's entropy assumed that the samples were randomly selected and these samples represented all species in an area. On the other hand, the empirical Simpson's index gave more weight to dominant species, which discounted the presence of rare species with few representatives in the estimation of diversity. To reduce error from these biases, the theoretical estimates of these diversity indices were used in this study. The best possible estimator of Shannon's entropy by Chao *et al*. (2013), and the minimum variance unbiased estimator of Simpson's diversity by Magurran (1988) were obtained from *SpadeR*'s species diversity analysis. For each estimate, the standard error and 95% confidence interval were determined based on *SpadeR* bootstrap method.

The beta diversity or presence of similar species across different types of urban greenspace was also determined using Sorensen's and Jaccard's similarity coefficients derived from *SpadeR*'s multiple community measures. Pairwise similarity coefficients were also computed for study sites of the same type of urban greenspace. The percentage agreement of study sites was illustrated using a dendrogram, generated through the agglomerative hierarchical clustering function of *XLSTAT*.

### Structure of Urban Greenspaces

The structure of urban greenspaces was characterized based on the distribution of trees into six DBH, height, and crown spread classes (**Table 2**). The relative number of individuals per class was used in testing the association of structural patterns with the type of urban greenspace. The indices of species composition and diversity and the relative proportions of structural classes were analyzed to test if there were significant differences between types of urban greenspace. To identify the appropriate statistical analysis to be used for the comparison, data were subjected to normality tests, including Shapiro-Wilk test, Anderson-Darling test, Lilliefors test, and Jarque-Bera test. All parameters returned a normality p-value <0.0001 in at least one normality test, and thus were analyzed using Kruskal-Wallis nonparametric test. The parameters with significant differences across types of urban greenspaces were further analyzed using Dunn's multiple pairwise comparison to determine which types of urban greenspace were significantly different. For categorical parameters, such as native distribution, conservation status, diameter class, height class, and crown spread class, the Chi-Square test of independence was used to determine which parameters were associated with the type of greenspace. The program XLSTAT was used to run the normality, Kruskal-Wallis, Dunn's multiple pairwise comparison, and Chi-Square tests. Mosaic plots of the Chi-square test was generated from SAS JMP.

**Table 2**. Pre-determined structural classes for tree DBH, height, and crown spread.

| Class No. | DBH classes | Height classes | Crown spread classes |
|---|---|---|---|
| 1 | <15 cm | < 3 m | < 4 m |
| 2 | 15 to 30 cm | 3 to 6 m | 4 to 8 m |
| 3 | 30 to 45 cm | 6 to 9 m | 8 to 12 m |
| 4 | 45 to 60 cm | 9 to 12 m | 12 to 16 m |
| 5 | 60 to 75 cm | 12 to 15 m | 16 to 20 m |
| 6 | 75 cm and above | 15 m and above | 20 m and above |

## RESULTS AND DISCUSSION

### Tree Species Composition of Urban Greenspaces

A total of 146 species of trees from 36 botanical families were identified in fourteen urban greenspaces. Each type of urban greenspace was composed of varying species of trees; however, few species dominate each site. In general, the five most abundant tree species compose more than 50% of trees in each site. At least 54% of trees in commercial greenspaces and recreational parks were dominated by five species and below. For wildlife parks, five species constitute 51% to 62% of trees in each site. Same results were reported from urban parks in Bangalore, India where almost half of the tree population were represented by the five most common species (Nagendra & Gopal 2010). In Guangzhou, Taipei and Hong Kong, the five most abundance species compose 40% to 55% of all trees (Jim 2008). Nowak (1994) also cited that five to six species composed more than half of urban tree population in Chicago and Athens.

The 36 botanical families detected in the study sites were led by Fabaceae and Moraceae, which represented 21% and 14% of all trees (**Table 3**). Fabaceae substantially dominated all types of urban greenspaces with 31 species, mean relative abundance of 43%, and mean relative dominance of 49%. Recreational parks recorded the highest mean relative abundance and relative dominance of Fabaceae at 55% and 59%, respectively. Four out of the 10 most common, abundant, and dominant species in the study sites were from Fabaceae family including *Acacia saman, Cassia fistula, Delonix regia*, and *Pterocarpus. indicus* forma *indicus*. Similarly, six out of the 14 most common street trees in Hong Kong were from Fabaceae family, the most abundant of which were *D. regia, C. siamea, C. surattensis, Acacia confusa, Bauhinia blakeana,* and *Albizzia lebbeck* (Jim 2008).

Family Moraceae, represented by 21 species dominated by *Ficus benjamina* in both abundance and basal area, has the second highest mean relative dominance (12%), and third highest mean relative abundance (9%). The mean relative abundance of Moraceae was exceeded by Meliaceae (12%), with only nine species, due to profused abundance of *Swietenia macrophylla* in WPs. Myrtaceae occupied the fourth place with eight species, and mean relative dominance of 14% in CGs, mainly due to high basal area of *Eucalyptus globulus.*

Taipei streets were also dominated by Fabaceae species, including *P. pterocarpum, E. indica* and *P. pinnata*, as well as Moraceae species, such as *F. microcarpa, F. religiosa,* and *F. elastica* (Jim 2008). Moraceae, Myrtaceae, and Caesalpiniaceae (Fabaceae) were also the most abundant botanical families in the urban forests of Guangzhou, China with respective relative abundances of 16% (20 species), 13% (19 species), and 12% (14 species) (Jim & Liu 2000). The abundances of these families in Guangzhou were mainly contributed by *Ficus virens* (Moraceae), *Melalueca leucandendra* (Myrtaceae), and *Bauhinia purpurea,* and *B. variegata* (Caesalpiniaceae) (Jim & Liu 2000).

The 10 common species identified in at least 50% of study sites were four native species, namely *Alstonia scholaris, P. indicus* forma *indicus, Terminalia. cattappa,* and *V. parviflora;* and six

**Table 3.** Most abundant and dominant botanical families in the study sites.

| Family | Species count | | | | % of all trees | | | | Mean rel. abundance (%) | | | | Mean rel. dominance (%) | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | T | CG | RP | WP | Tot | CG | RP | WP | Tot | CG | RP | WP | Tot | CG | RP | WP |
| Fabaceae | 31 | 19 | 12 | 26 | **21.2** | **34.5** | **29.3** | **21.0** | **42.9** | **37.3** | **55.1** | **27.7** | **48.9** | **42.9** | **58.5** | **39.6** |
| Moraceae | 21 | 6 | 3 | 17 | **14.4** | **10.9** | 7.3 | **13.7** | 8.9 | **11.8** | 5.9 | **10.0** | **12.1** | **14.6** | **12.7** | 6.7 |
| Meliaceae | 9 | 4 | 6 | 7 | 6.2 | 7.3 | **14.6** | 5.6 | **12.0** | 7.8 | **10.3** | **22.2** | 7.1 | 3.7 | 5.7 | **15.5** |
| Myrtaceae | 8 | 2 | 1 | 7 | 5.5 | 3.6 | 2.4 | 5.6 | 3.7 | 8.0 | 0.2 | 3.8 | 5.8 | **13.6** | 0.1 | 4.3 |
| Malvaceae | 7 | 3 | 1 | 5 | 4.8 | 5.5 | 2.4 | 4.0 | 1.9 | 3.5 | 0.9 | 1.4 | 3.2 | 4.2 | 0.5 | 7.0 |
| Anacardiaceae | 6 | 2 | 1 | 6 | 4.1 | 3.6 | 2.4 | 4.8 | 3.0 | 1.1 | 1.8 | 8.4 | 2.6 | 1.1 | 1.6 | 7.1 |
| Annonaceae | 5 | 1 | 1 | 5 | 3.4 | 1.8 | 2.4 | 4.0 | 1.1 | 0.3 | 0.2 | 4.5 | 0.4 | - | - | 1.9 |
| Euphorbiaceae | 5 | 3 | 2 | 2 | 3.4 | 5.5 | 4.9 | 1.6 | 2.8 | 1.4 | 5.3 | 0.3 | 2.4 | 1.7 | 4.2 | 0.1 |
| Lamiaceae | 5 | 1 | 3 | 5 | 3.4 | 1.8 | 7.3 | 4.0 | 4.4 | 0.2 | 6.5 | 7.5 | 3.3 | - | 4.1 | 7.0 |
| Sapindaceae | 5 | 2 | 1 | 3 | 3.4 | 3.6 | 2.4 | 2.4 | 0.8 | 0.4 | 1.4 | 0.2 | 1.8 | 0.2 | 4.1 | - |

exotic species, such as *A. saman, C. fistula, D. regia, F. benjamina, M. indica,* and *S. macrophylla. A. saman* and *T. catappa* were widely used for their extensive crown that provides shade and ameliorate temperature in urban areas. *C. fistula* and *D. regia* were typically planted in urban areas for their striking ornamental flowers; while *F. benjamina* was commonly used for its aesthetic aerial roots and ornamental leaves that can be shaped into a topiary. The abundance of *A. saman, P. indicus* forma *indicus,* and *S. macrophylla* could be attributed to rapid growth, low maintenance requirements, and high transplanting survival. *M. indica* in urban areas were typically retained from previous land use, and some were intentionally planted for the production of fruits, or randomly dispersed by urban dwellers. *A. scholaris* and *V. parviflora* were commonly planted for their cultural values as native species, alongside its uses as ornamental and shade-providing trees.

The common species discussed above were not necessarily abundant and dominant in all types of urban greenspaces (**Figure 2**). Same results were reported from urban greenspaces in China, Canada, Arizona, and Florida where dominant species varied by land use (Jim & Liu 2000; Bourne & Conway 2013; Kim 2016; Escobedo *et al.* 2018).

The most abundant species found in each type of urban greenspace were *D. regia* (18%) in CGs, *C. fistula* (22%) in RPs, and *S. macrophylla* (20%) in WPs (**Table 4**). The mean relative abundance of these species were comparatively higher than the proportion of the most abundant species in urban greenspaces in Chicago (12% to 15%) but similar with those in Guangzhou, China (6% to 20%) and in Peel Region, Canada (10% to 44%) (Nowak, 1994; Jim & Liu 2000; Bourne & Conway 2013). These relative abundance values were also far from meeting the 10% maximum abundance threshold proposed by Santamour (1990) as a preventive measure against pest epidemics in urban areas. In terms of basal area, *A. saman* was the most dominant species in all types of urban greenspaces with mean relative dominance of 23.5% in CGs, 13.7% in RPss, and 9% in WPs (**Table 5**). Overall, exotic species composed a significant proportion of trees, both in abundance and dominance, in all types of greenspaces.

Among the 10 most common species, only *S. macrophylla* has significantly different ($\alpha$=0.05) relative abundances (p-value=0.046) and relative dominances (p-value=0.027) across the three types of urban greenspace. Pairwise comparisons showed that *S. macrophylla* has significantly higher relative abundance in WPs than CGs (p-value = 0.016) and RPs (p-value = 0.048). However, there was no significant difference in the relative abundance of *S. macrophylla* in CGs and RPs (p-value = 0.616). Likewise, the relative dominance of *S. macrophylla* was significantly higher in WPs than CGs (p-value=0.013) and RPs (p-value=0.018). There was no significant difference in the species' relative dominance between CGs and RPs (p-value=0.821).

**Proportion of Native Tree Species in Urban Greenspaces**

Out of 146 species identified in urban greenspaces, 76 species or 52% were native species. The same proportion of native species was recorded in Guangzhou, China where 52% of 254 urban tree

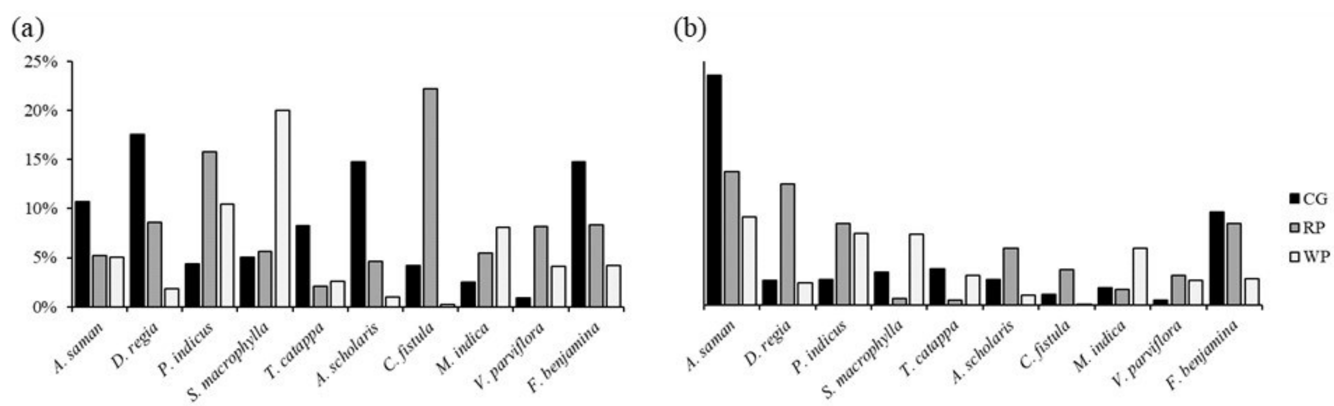



**Figure 2.** Average (a) relative abundance and (b) relative dominance of tree species in three types of urban greenspaces.

**Table 4.** Relative abundance of tree species in the study sites.

| Species ID | Commercial greenspaces | | | | | Residential parks | | | | | Wildlife parks | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 1 | 2 | 3 | 4 | 5 | 1 | 2 | 3 |
| *A. auriculiformis* | - | - | 3.3 | - | - | 4.0 | - | - | - | - | 4.7 | - | - |
| *A. bunius* | - | - | 2.2 | - | - | 2.0 | - | - | - | - | - | 2.8 | 0.6 |
| ***A. indica*** | 3.6 | 0.9 | - | **21.7** | - | 5.1 | - | 6.7 | - | - | - | - | 1.1 |
| *A. mangium* | - | 1.8 | - | - | - | - | - | - | - | - | 0.3 | - | 1.8 |
| ***A. saman*** | **20.1** | 5.5 | - | 6.5 | - | 7.1 | 6.3 | 3.3 | 5.9 | 3.6 | 9.8 | 1.9 | 3.6 |
| ***A. scholaris*** | **10.3** | - | 8.7 | - | **25.3** | 3.0 | 6.3 | - | - | - | - | 1.9 | 0.0 |
| *B. acerifolius* | - | - | 8.7 | - | - | - | - | - | 1.0 | 4.5 | - | - | - |
| *B. malabarica* | - | - | - | 4.3 | - | 1.0 | - | 1.1 | - | - | - | 0.9 | 0.2 |
| ***B. monandra*** | **16.6** | - | - | - | - | - | - | - | - | - | 0.6 | 0.9 | 2.9 |
| *C. cainito* | 0.2 | - | - | - | - | - | - | - | - | - | 0.1 | 4.7 | 1.1 |
| ***C. fistula*** | 0.7 | 5.5 | - | 6.5 | - | 1.0 | - | **43.3** | - | - | 0.2 | - | 0.2 |
| ***C. inophyllum*** | - | - | - | - | **34.2** | - | 3.1 | - | - | 9.8 | - | 1.9 | - |
| ***C. javanica*** | - | - | - | **15.2** | - | - | 6.3 | - | - | - | - | - | - |
| *C. odorata* | - | - | - | - | - | 1.0 | - | - | - | - | 0.3 | - | - |
| *C. pulcherrima* | - | 4.5 | - | - | - | - | - | - | - | - | 0.2 | - | - |
| *C. ramiflora* | - | - | 2.7 | - | - | - | - | - | - | - | 0.1 | - | 0.3 |
| *D. gaudichaudianum* | - | - | - | - | - | - | 6.3 | - | 8.9 | - | - | - | 0.1 |
| *D. philippinensis* | - | 2.7 | - | - | - | 1.0 | - | - | - | - | 0.3 | - | - |
| ***D. regia*** | 0.7 | **12.7** | - | - | **39.2** | 1.0 | - | **25.6** | 5.9 | 1.8 | 1.3 | 0.9 | 3.3 |
| ***E. deglupta*** | - | - | - | **15.2** | - | - | - | - | - | - | 0.5 | - | 0.7 |
| ***E. globulus*** | - | **20.0** | 4.9 | - | - | - | - | - | - | - | 7.9 | - | - |
| ***F. benjamina*** | 1.3 | - | - | **28.3** | - | 1.0 | **15.6** | - | - | - | 9.6 | 2.8 | 0.2 |
| *F. elastica* | - | 2.7 | - | - | - | 5.1 | - | - | 5.0 | - | 0.6 | - | 1.1 |
| *F. lyrata* | - | - | 9.3 | - | - | - | - | - | 1.0 | 8.0 | - | - | - |
| *F. simplex* | - | 6.4 | - | - | - | - | - | - | - | - | - | 1.9 | - |
| ***G. arborea*** | - | - | - | - | - | - | 6.3 | - | - | - | 0.5 | - | 7.0 |
| *H. brasiliensis* | - | - | 3.3 | - | - | - | - | - | - | 7.1 | - | - | - |
| *I. bijuga* | - | 3.6 | 0.5 | - | - | - | - | - | 5.0 | 2.7 | - | - | 0.2 |
| ***L. ferrea*** | - | - | 1.1 | - | - | - | - | - | 6.9 | **15.2** | - | 2.8 | - |
| *L. leucocephala* | - | - | - | - | - | - | - | - | - | 2.7 | - | 2.8 | 2.0 |
| ***L. speciosa*** | **12.3** | 0.9 | - | - | - | - | - | - | - | - | - | - | 2.7 |
| ***M. indica*** | 2.2 | 2.7 | - | - | - | 1.0 | - | **10.0** | - | - | 4.1 | **15.0** | 5.1 |
| ***M. multiglandulosa*** | 2.5 | - | - | - | - | - | - | - | **12.9** | **11.6** | - | - | - |
| *P. dulce* | 0.4 | 3.6 | - | - | - | - | - | - | - | - | 0.1 | - | 1.3 |
| ***P. indicus*** | 2.7 | 9.1 | - | - | 1.3 | **24.2** | **21.9** | 1.1 | - | - | 2.3 | **14.0** | **15.1** |
| *P. longifolia* | 1.3 | - | - | - | - | - | - | - | - | - | 6.1 | 6.5 | 0.3 |
| *P. odorata* | - | - | - | - | - | 8.1 | - | - | - | - | 0.2 | 0.9 | 0.1 |
| *P. pinnata* | - | - | 2.2 | - | - | - | - | - | - | - | - | - | 0.9 |
| *P. pterocarpum* | - | - | - | - | - | - | - | - | 5.9 | 9.8 | - | - | 0.0 |
| *Plumeria sp.* | 6.7 | - | - | - | - | 2.0 | - | - | - | - | 0.9 | - | 0.4 |
| *S. actinophylla* | - | - | - | - | - | - | - | - | 4.0 | - | - | - | 0.1 |
| ***S. campanulata*** | 2.2 | - | - | 2.2 | - | **10.1** | - | - | - | - | 0.2 | - | 1.4 |
| *S. koetjape* | 0.7 | - | - | - | - | - | 3.1 | - | - | - | 0.5 | 3.7 | 0.5 |
| ***S. macrophylla*** | 6.5 | 3.6 | - | - | - | 2.0 | 9.4 | 5.6 | - | - | **28.9** | 7.5 | **23.7** |
| *S. purpurea* | 0.7 | - | - | - | - | - | - | - | - | - | 0.2 | - | - |
| *S. saponaria* | - | - | - | - | - | - | - | - | 5.0 | 3.6 | 0.1 | - | 0.0 |
| ***T. catappa*** | 0.4 | 4.5 | **19.7** | - | - | 2.0 | - | 2.2 | - | - | 2.2 | 3.7 | 2.0 |
| *T. cumingiana* | 0.4 | - | - | - | - | - | - | - | - | - | - | - | 4.4 |
| *T. indica* | 0.2 | - | - | - | - | - | - | - | - | - | 2.0 | - | 0.8 |
| ***Tabebuia sp.*** | - | - | 7.1 | - | - | - | - | - | **16.8** | 2.7 | - | - | - |
| ***V. parviflora*** | 0.9 | - | - | - | - | **17.2** | 6.3 | 1.1 | - | - | 0.1 | 8.4 | 3.9 |

**Table 5.** Relative dominance of tree species in the study sites.

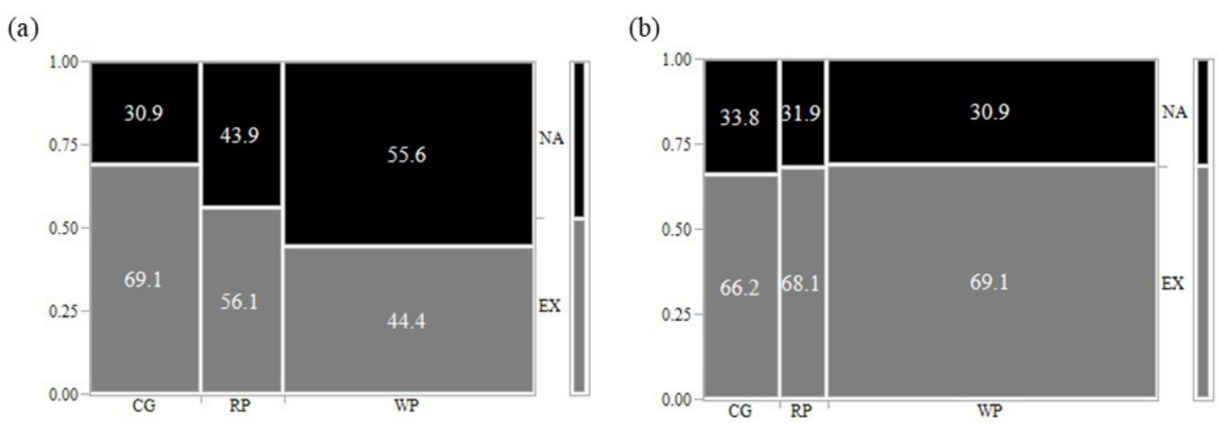
| Species ID | Commercial greenspaces | | | | | Recreational parks | | | | | Wildlife parks | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 1 | 2 | 3 | 4 | 5 | 1 | 2 | 3 |
| *A. auriculiformis* | - | - | 4.9 | - | - | 4.9 | - | - | - | - | - | 6.1 | - |
| *A. bunius* | - | - | 1.5 | - | - | 0.4 | - | - | - | - | - | - | 1.1 |
| *A. indica* | 3.8 | 0.2 | - | 9.4 | - | 2.6 | - | - | 8.0 | - | - | - | - |
| *A. mangium* | - | 0.7 | - | - | - | - | - | - | - | - | - | 1.1 | - |
| ***A. saman*** | **62.6** | **10.9** | - | **24.7** | - | **20.4** | - | **11.7** | **23.5** | **26.6** | 2.1 | **23.9** | 1.1 |
| ***A. scholaris*** | 2.1 | - | 9.8 | - | **27.1** | 5.7 | - | 1.9 | - | - | - | - | 2.9 |
| ***B. acerifolius*** | - | - | **11.4** | - | - | - | - | - | - | 0.2 | 2.6 | - | - |
| *B. malabarica* | - | - | - | 1.6 | - | 1.9 | - | - | 0.3 | - | - | - | 0.6 |
| *B. monandra* | 3.2 | - | - | - | - | - | - | - | - | - | - | 0.3 | 0.1 |
| *C. cainito* | 0.0 | - | - | - | - | - | - | - | - | - | - | 0.4 | 1.9 |
| ***C. fistula*** | 0.2 | 1.1 | - | 3.7 | - | 0.2 | - | - | **21.3** | - | - | 0.1 | - |
| ***C. inophyllum*** | - | - | - | - | **21.1** | - | - | 0.2 | - | - | 10.0 | - | 3.0 |
| ***C. javanica*** | - | - | - | **10.7** | - | - | - | 0.3 | - | - | - | - | - |
| *C. odorata* | - | - | - | - | - | 0.2 | - | - | - | - | - | 0.2 | - |
| *C. pulcherrima* | - | 0.8 | - | - | - | - | - | - | - | - | - | 0.0 | - |
| *C. ramiflora* | - | - | 1.6 | - | - | - | - | - | - | - | - | 0.0 | - |
| *D. gaudichaudianum* | - | - | - | - | - | - | - | 2.2 | - | 5.9 | - | - | - |
| *D. philippinensis* | - | 0.1 | - | - | - | 0.4 | - | - | - | - | - | 0.2 | - |
| ***D. regia*** | 0.2 | 7.3 | - | - | **44.4** | 0.5 | - | - | **26.0** | 3.3 | 3.8 | 2.4 | 0.5 |
| *E. deglupta* | - | - | - | 5.4 | - | - | - | - | - | - | - | 0.1 | - |
| ***E. globulus*** | - | **55.0** | 7.6 | - | - | - | - | - | - | - | - | **10.9** | - |
| ***F. benjamina*** | 0.9 | - | - | **43.8** | - | 0.1 | - | **50.1** | - | - | - | 7.8 | 0.2 |
| ***F. elastica*** | - | **11.4** | - | - | - | **18.3** | - | - | - | 1.3 | - | 3.6 | - |
| *F. lyrata* | - | - | 8.4 | - | - | - | - | - | - | 0.1 | 6.2 | - | - |
| ***F. simplex*** | - | 4.5 | - | - | - | - | - | - | - | - | - | - | **14.6** |
| *G. arborea* | - | - | - | - | - | - | - | 2.3 | - | - | - | 0.6 | - |
| *H. brasiliensis* | - | - | 4.8 | - | - | - | - | - | - | - | 5.8 | - | - |
| *I. bijuga* | - | 0.5 | 0.3 | - | - | - | - | - | - | 1.5 | 3.6 | - | - |
| *L. ferrea* | - | - | 0.5 | - | - | - | - | - | - | 5.3 | 9.5 | - | 3.0 |
| *L. leucocephala* | - | - | - | - | - | - | - | - | - | - | 3.1 | - | 1.3 |
| *L. speciosa* | 3.1 | 0.0 | - | - | - | - | - | - | - | - | - | - | - |
| ***M. indica*** | 4.3 | 0.9 | - | - | - | 0.1 | - | - | 9.6 | - | - | 4.7 | **12.7** |
| ***M. multiglandulosa*** | 2.9 | - | - | - | - | - | - | - | - | 5.3 | **13.9** | - | - |
| *P. dulce* | 0.1 | 3.4 | - | - | - | - | - | - | - | - | - | 0.3 | - |
| ***P. indicus*** | 1.5 | 1.1 | - | - | 7.4 | **14.7** | - | **26.6** | 1.2 | - | - | 2.4 | **19.6** |
| *P. longifolia* | 0.2 | - | - | - | - | - | - | - | - | - | - | 1.3 | 4.0 |
| *P. odorata* | - | - | - | - | - | 4.6 | - | - | - | - | - | 0.0 | 0.4 |
| *P. pinnata* | - | - | 1.2 | - | - | - | - | - | - | - | - | - | - |
| ***P. pterocarpum*** | - | - | - | - | - | - | - | - | - | 3.4 | **18.0** | - | - |
| *Plumeria sp.* | 0.7 | - | - | - | - | 0.7 | - | - | - | - | - | 0.1 | - |
| *S. actinophylla* | - | - | - | - | - | - | - | - | - | 1.1 | - | - | - |
| ***S. campanulata*** | 1.6 | - | - | 0.7 | - | **11.3** | - | - | - | - | - | 0.1 | - |
| *S. koetjape* | 1.1 | - | - | - | - | - | - | 0.2 | - | - | - | 0.2 | 2.2 |
| ***S. macrophylla*** | 2.8 | 0.4 | - | - | - | 1.1 | - | 0.8 | 2.0 | - | - | **17.5** | 4.3 |
| *S. purpurea* | 0.3 | - | - | - | - | - | - | - | - | - | - | 0.2 | - |
| ***S. saponaria*** | - | - | - | - | - | - | - | - | - | **21.6** | 2.8 | 0.0 | - |
| ***T. catappa*** | 1.6 | 0.3 | **14.2** | - | - | 0.9 | - | - | 2.1 | - | - | 1.6 | 7.4 |
| *T. cumingiana* | 0.2 | - | - | - | - | - | - | - | - | - | - | - | - |
| *T. indica* | 0.0 | - | - | - | - | - | - | - | - | - | - | 1.3 | - |
| ***Tabebuia sp.*** | - | - | 5.2 | - | - | - | - | - | - | **12.3** | 2.2 | - | - |
| ***V. parviflora*** | 0.2 | - | - | - | - | **10.5** | - | 1.4 | 6.0 | - | - | 0.1 | 7.3 |

species were native (Jim & Liu 2000). The relative number of native and exotic species was found to have significant association with the type of urban greenspace, $\chi^2(2) = 9.58$, p = 0.0083 (**Figure 3a**). WPs has the highest proportion of native species with a mean proportion of 51%; followed by 43% in CGs. RPs has the lowest mean proportion of native species at 38%. These proportions were comparatively higher than the 34% share of native species in the urban parks of Bangalore, India (Nagendra & Gopal 2010). Different urban habitats in Sahiwal City, Pakistan also had varying proportions of native species that ranged from 30% in institutes and streets to 56% in graveyards (Akbar *et al*. 2014) . In the City of Roanoke, Arizona the proportion of native species ranged between 69% in vacant lands to more than 85% in industrial and commercial lands (Kim 2016).

In terms of abundance, only 31% of more than 5,000 trees observed were native species. Across the three types of urban greenspaces studied, the mean proportion of native trees were almost the same, ranging between 31% in WPs and 34% in CGs.Thus, no significant association was found, $\chi^2(2) = 2.667$, p = 0.2636 (**Figure 3b**). This was relatively higher than the 23% abundance of native species in urban parks of Bangalore, India (Nagendra & Gopal 2010), but comparatively lower than the 56% proportion of native species in Sahiwal City, Pakistan (Akbar *et al*. 2014), 57% in the urban forests of Guangzhou, China (Jim & Liu 2000), and 89% in the city of Gainesville, Florida (Escobedo *et al*. 2018) The native species with the highest relative number of individuals in all study sites were *P. indicus* forma *indicus* (9.7%), *V. parviflora* (2.7%), *L. speciosa* (2.5%), *T. catappa* (2.4%), and *A. scholaris* (1.8%). On the other hand, the most abundant exotic species in all study sites were *S. macrophylla* (18.7%), *A. saman* (6.1%), *M. indica* (4.2%), *G. arborea* (3.8%), and *D. regia* (3.5%). Some of the exotic tree species found in this study were also introduced in the urban parks of Bangalore, India. These included *D. regia, P. longifolia, P. pterocarpum, S. campanulata,* and *T. aurea,* which were commonly planted for their ornamental uses (Nagendra & Gopal 2010). *M. indica* was also one of the most abundant exotic species found in roadside and institutional greenspaces in Guangzhou, China (Jim & Liu 2000), as well as in roadsides in Taipei (Jim 2008). Jim (2008) stated that the use of exotic species in urban greenspaces in Hong Kong, Taipei, and Guangzhou could be due to lack of planting stocks, limited knowledge on native species with ornamental bloom, and inadequate practical experiences in propagating native species in urban areas. These reasons also could possibly hold true in the dominance of exotic trees in the study sites. However, national policies encouraging the use of native species had already started a shift from the use of exotic to native species. Despite limitations in attracting native wildlife, exotic species were found to contribute to increasing tree diversity and thus reducing susceptibility of urban greenspaces to pest infestation. In Oakland, 69% of urban tree species were exotic, but the introduction of exotic species was said to increase its Shannon-Wiener's diversity index from 1.9 to 5.1 (Nowak 1994). In the study of Akbar *et al*. (2014) in Sahiwal City, Pakistan, urban habitats with the highest percentage of exotic species also had the highest species diversity indices. Kim (2016) also stated that the mix of native and exotic species in urban forests often resulted to higher tree species diversity than adjacent native landscapes in Roanoke. However, he added that the dominance of exotic species with invasive characteristics should be controlled to prevent them from displacing native species and to provide sufficient habitat for native wildlife (Kim 2016).

***Conservation Status of Tree Species Used in Urban Greenspaces***

The importance of urban greenspaces for the conservation of threatened species was manifested by the presence of 19 locally threatened species in the study sites. All of these species, except for *T. calantas*, were found in wildlife parks. Only four threatened species were observed in recreational parks and only three in commercial greenspaces. *H. foxworthyi* was the only *critically endangered* species found in one of the wildlife parks. There were five locally *endangered* species observed, including *Afzelia. romboidea, Sindora. supa*, *T. philippinensis, V. parviflora, and Xanthostemon verdugonianum.* There were also 11 vulnerable species and two other threatened species observed in urban greenspaces. The mean proportion of locally threatened species in each type of urban greenspace ranged from 10% in commercial greenspaces to 12% in both recreational and wildlife parks. Chi-square test of independence indicated that the relative number of threatened and non-threatened trees was significantly associated with the type of urban greenspace, $\chi^2(2) = 97.690$, $p < 0.0001$ (**Figure 4b**). However, no significant association was found between proportion of species and the type of urban greenspace, $\chi^2(2) = 3.009$, p = 0.2221 (**Figure 4a).**




| Test | Chi Square | Prob>ChiSq |
|---|---|---|
| Likelihood ratio | 9.766 | 0.0076* |
| Pearson | 9.583 | 0.0083* |

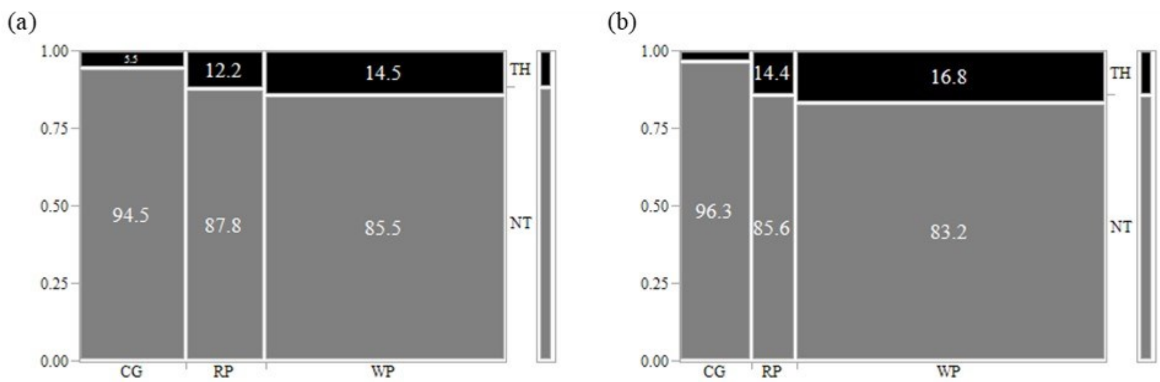
| Test | Chi Square | Prob>ChiSq |
|---|---|---|
| Likelihood ratio | 2.644 | 0.2666ns |
| Pearson | 2.667 | 0.2636ns |

**Figure 3.** Mosaic plots showing the relative number of native and exotic species and the relative number of native and exotic trees across three types of urban greenspace.

Some native tree species abundant in the locality, could also be classified as threatened, in view of its global presence. More than 30% or 47 out of 146 species in the study sites were globally threatened species. Almost all of these species were found in WPs (94%), while less than 40% were observed in CGs (38%), and RPs (32%). The type of urban greenspace had no significant association with the relative number of globally threatened and non-threatened species, $\chi^2(2) = 0.801$, $p = 0.6699$ (**Figure 5a**), but was significantly associated with the relative number of threatened and non-threatened trees, $\chi^2(2) = 182.493$, $p < 0.0001$ (**Figure 5b**). The two critically endangered species, namely *H. plagata* and *T. philippinensis,* were only found in one wildlife park. CGs had the highest mean proportion of globally threatened species, most of which were in the *least concern* category. This shows that there were existing efforts to plant threatened species in urban greenspaces. However, there were few endangered species found, especially in CGs and RPs. This could be due to limited availability of pole-sized trees of these species, which is necessary in providing instantaneous impression in commercial and recreational landscapes.

***Species Richness and Diversity***

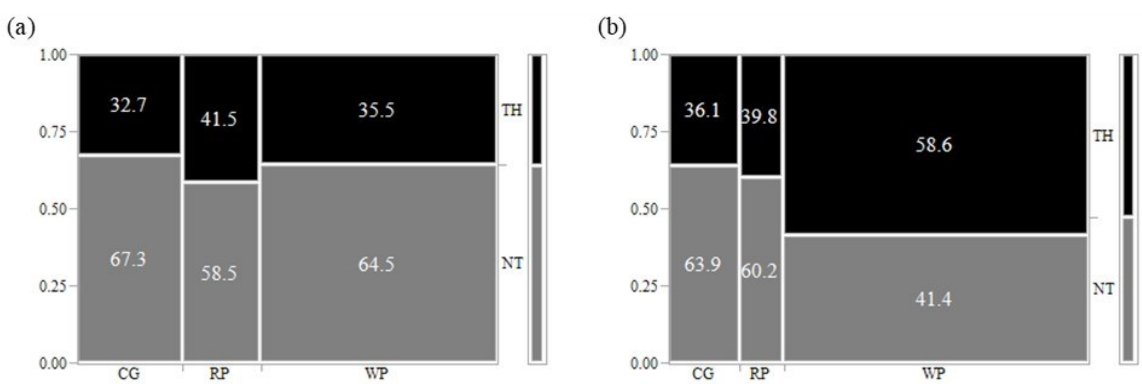
Of the 146 species recorded in all study sites, 124 (85%) could be found in WPs, 55 (38%) in CGs, and 40 (27%) in RPs. The mean species richness estimated from Chao's model (1984) through *SpadeR* were 83 species, 20 species, and 17 species for WP, CG, and RP, respectively. These estimated species richness values were significantly different (p-value = 0.038) across different types of urban greenspace. Species richness in WPs were significantly different from RPs (p-value = 0.012), but were less distinct from CGs (p-value = 0.056) (**Figure 6a**). CGs and RPs had statistically similar species richness estimates (p-value = 0.576). Significant differences in tree species richness were also documented in different urban land uses in Mexico City (Ortega-Alvarez *et al*. 2011). However, results from the said study showed significantly higher species richness in residential (49 species) and commercial (32 species) areas than in green areas (20 species). Higher species richness in commercial and residential areas was attributed to landscaping practices and cultural preferences influencing species selection; while low species richness in green areas were said to be influenced by greening purposes using few species, and retention of original vegetation of limited species (Ortega-Alvarez *et al*. 2011).




| Test | Chi Square | Prob>ChiSq |
|---|---|---|
| Likelihood ratio | 3.428 | 0.1801ns |
| Pearson | 3.009 | 0.2221ns |

| Test | Chi Square | Prob>ChiSq |
|---|---|---|
| Likelihood ratio | 125.336 | <0.0001* |
| Pearson | 97.690 | <0.0001* |

**Figure 4.** Mosaic plots showing the relative number of locally threatened and non-threatened species and the relative number of locally threatened and non-threatened trees across three types of urban greenspace.




| Test | Chi Square | Prob>ChiSq |
|---|---|---|
| Likelihood ratio | 0.794 | 0.6724ns |
| Pearson | 0.801 | 0.6699ns |

| Test | Chi Square | Prob>ChiSq |
|---|---|---|
| Likelihood ratio | 183.497 | <0.0001* |
| Pearson | 182.493 | <0.0001* |

**Figure 5.** Mosaic plots showing the relative number of globally threatened and non-threatened species and the relative number of globally threatened and non-threatened trees across three types of urban greenspace.

Wildlife parks had the highest alpha diversity values, estimated using the Shannon-Wiener diversity index (H'), with an average of 3.1 (**Figure 6b**). The average H' values obtained from CGs and RPs, where almost the same at 2.5 and 2.4, respectively. Species diversity is influenced by the relative proportion of the most abundant species; as well as the number of rare species, which in this study refers to species with only 10 individuals and below. The highest relative abundance of a species recorded from WPs, CGs, and RPs, were 29%, 34%, and 43%, respectively. The average number of rare species found were 48 species in WPs, 13 in CGs, and 12 in RPs. Significant differences in H' values (p-value = 0.029) resulted from the comparison of the different urban greenspaces. WPs had higher H' values than RPs (p-value = 0.008) and CGs (p-value = 0.085), but the difference was only significant with RPs. The H' values of CGs and RPs were not significantly different (p-value = 0.352). The Simpson diversity indices (**Figure 6c**) corroborated the rankings of H', but the estimates did not show significant differences across the three types of urban greenspace (p-value =0.960).

The average pairwise beta diversity value across the three types of urban greenspace was 0.282, which means that only 28% of the species in each type could also be found in other types. The equivalent Sorensen ($\beta_s$) and Jaccard ($\beta_j$) similarity indices estimated through *SpadeR* were 0.420 and 0.194, respectively.

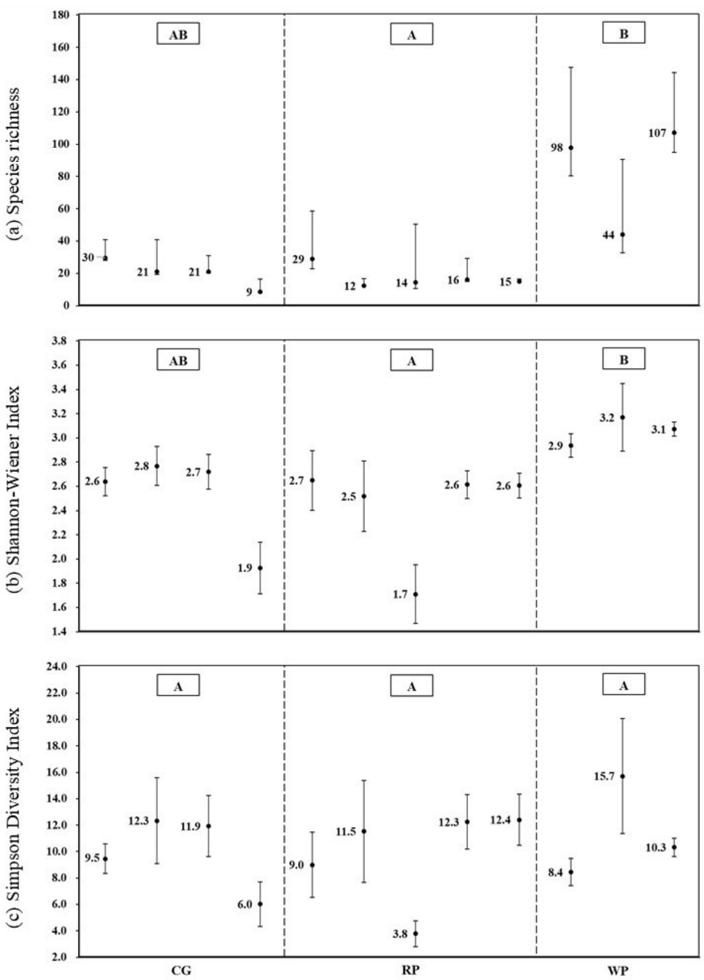



**Figure 6.** Estimates of species richness, Shannon-Wiener diversity index, and Simpson's diversity index.

These low beta diversity values imply that different types or urban greenspaces were composed of different tree species. Based on pairwise comparison, the species composition of CGs and RPs were more similar to each other than when compared with WPs. The $\beta_s$ and $\beta_j$ coefficients for the pairwise comparison of CG and RP species were 0.413 and 0.585, respectively (**Figure 7**). CG and WP had $\beta_s$ = 0.377 and $\beta_s$ = 0.232; while RP and WP had the least beta diversity of $\beta_s$ = 0.335 and $\beta_j$ = 0.201.

Across the three types of urban greenspace, wildlife parks had the highest proportion of shared species among related samples ($\beta_s$ = 0.649, $\beta_j$ = 0.222). This implies that based on Sorensen's coefficient, 65% of species in one wildlife park could also be observed in another wildlife park. There was high similarity in the species composition of different wildlife parks because their management mainly entailed the use of diverse species of trees. The least proportion of common species was among CGs ($\beta_s$ = 0.418, $\beta_j$ = 0.126), which could be attributed to low tree species richness in these areas, as well as to highly variable species preferences of landscape developers. Kruskal-Wallis test results indicated that there were no significant differences between the beta diversities (p-value = 0.105) of the different types of urban greenspace.

Similar findings on tree diversity analysis were reported from southern California's urban forests. Both alpha (H' index) and beta diversities were found to be higher in parks (H' = 0.358, $\beta_w$ = 0.942) than in commercial greenspaces (H' = 0.094, $\beta_w$ = 0.595) (Avolio *et al*. 2015).

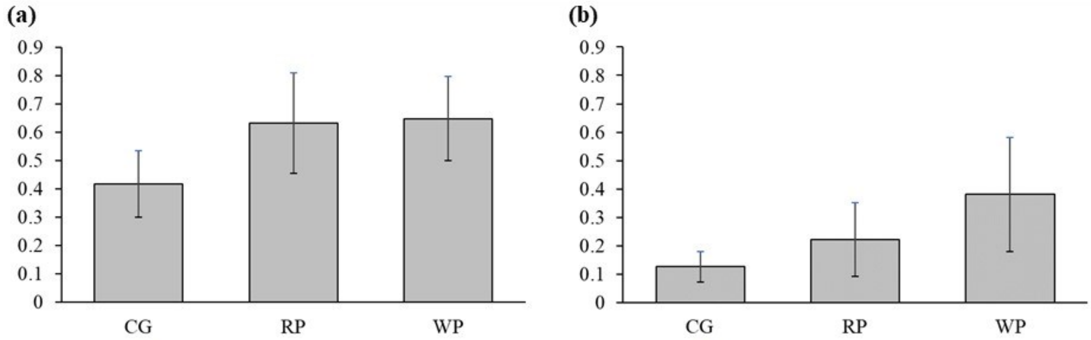
Furthermore, the similarity of urban greenspaces based on species abundance within each greenspace is illustrated in a dendrogram (**Figure 8**). It showed high percentage similarities across CGs and RPs, which ranged from 0.762 to 0.966. Wildlife parks had the least percentage similarity with the other types of urban greenspace with values less than 0.5. Similar results were obtained from the comparison of residential, commercial, and green areas in Mexico City, where residential and commercial areas had more than 0.5 similarity, while green areas only had an average of 0.25 similarity with the other land uses (Ortega-Alvarez *et al.* 2011). In this study, wildlife parks were comparable with green areas that were highly different from the other land uses, and was probably due to higher abundance of few species.

***Structure of Urban Greenspaces***

Chi-square tests showed that there were significant associations between the type of urban greenspace and the distribution of trees into diameter classes, $\chi^2(10)$ = 67.76, p <0.0001, height classes, $\chi^2$ (10) = 1424.22, p <0.0001, and crown spread classes, $\chi^2(10)$ = 307.86, p <0.0001. Similar results were reported by Welch (1994) in Boston where the size class categories of street and park tree populations were not distributed similarly.

The three types of urban greenspaces have the same distribution of DBH classes, which were skewed to the smaller DBH classes of 30 cm and below (**Figure 9a**). The proportion of small trees ranged from 60% in WPs to 64% CGs. The high proportion of small trees in longstanding and less-managed WPs could be attributed to natural regeneration; while the dominance of small trees in recently established CGs and RPs could be due to the abundance of transplanted trees.

Urban greenspaces in temperate countries in the United States were reported to have smaller sizes. In 1994, Nowak stated that majority of urban trees in the United States had less than 15 cm DBH. Specifically, 64% to 79% of urban trees in different urban greenspaces in Chicago have DBH of 15 cm and below (Nowak




| | | | |
|---|---|---|---|
| CG | **55** | 28 | 37 |
| RP | 0.585 | **40** | 30 |
| WP | 0.377 | 0.335 | **124** |

| | | | |
|---|---|---|---|
| CG | **55** | 28 | 37 |
| RP | 0.413 | **40** | 30 |
| WP | 0.232 | 0.201 | **124** |

**Figure 7.** Estimated Sorensen and Jaccard similarities within each type of urban greenspace.

1994). In 2016, the study of Kim revealed that 40% to 65% of trees in different types of urban land uses in Roanoke, Arizona also had DBH of 15 cm and below. Escobedo *et al*. (2018) also found that 49% of urban trees in Gainesville, Florida have less than 20 cm DBH. Nowak explained that the distribution of trees into size classes could be attributed to the intensity of greenspace management and the history of the site (Nowak 1994).

On the other hand, the highest proportion of large trees with more than 60 cm DBH was found in CGs (10.7%). Similar results were reported in Chicago where the highest proportion of large trees can be found in land uses dominated by buildings (Nowak 1994). Although less in abundance, large trees provide greater ecosystem functions and serve as keystone structures for wildlife (Stagoll *et al*. 2011; Kim 2016). These trees also pose higher risk than small trees, and thus entail higher maintenance requirements.

Across all study sites, almost 50% of large trees with DBH of 60 cm and above were composed of *A. saman* (31.7%), *P.indicus* (10.1%), and *S. macrophylla* (7.5%). Both *S. macrophylla* (19.4%) and *P indicus* (9.0%) have the highest proportion of small trees with DBH of 30 cm and below, and thus expected to dominate these urban greenspaces in the future. There may be few *A. saman* in the probable future greenspace as it only accounted to 1.1% of small trees. This future species composition and structure could be altered by the managers based on their preferences and target benefits from these greenspaces.

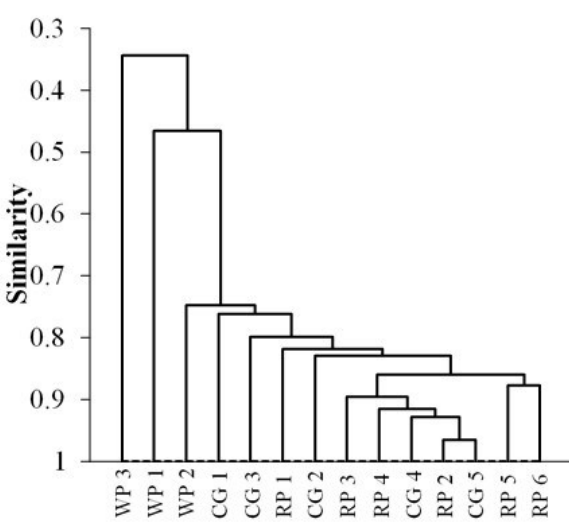



**Figure 8.** Dendrogram of urban greenspace similarity based on species abundance.

Variation in height classes creates vertical strata that serve as niches to support diverse species of wildlife. Each type of urban greenspace was dominated by distinct height classes (**Figure 9b**). More than 40% of trees in CGs were short in stature, having a height of less than 6 m. Due to closer proximity to urban structures, trees in CGs were frequently pruned by reducing or raising the crown to maintain vertical clearance from utility lines or to ensure the visibility of signages. Some trees were also pollarded for utility or aesthetic purposes. RPs were dominated by trees in the median height classes, from 6-9 m (47%) to 9-12 m (21%). The highest proportion of tall trees (51%) that were at least 12 m high (31%), were found in WPs.

The distribution of trees into crown spread classes were similar across the three types of urban greenspaces. More than 70% of trees in all types have small crown spread of less than 8 m (Figure 9c). Various ecosystem functions can be attributed to extensive crown spread such as microclimate amelioration, production of flowers, fruits and seeds, and wildlife habitat, among others. As with the DBH and height classes, CGs have the highest proportion (77%) of trees with small crown spread, as compared with recreational parks (74%) and wildlife parks (71%). Trees in urban areas are regularly thinned on the sides to maintain horizontal clearance between tree crown and infrastructures. Side pruning is also done to maintain adequate space between tree crowns to prevent rubbing of branches, which may result to tree branch failure.

Significant differences in tree measurements were found across the three types of urban greenspaces (**Figure 10**). The DBH of standard trees (DBH = 30 – 60 cm) in RPs were significantly larger than standard trees in CGs, p-value = 0.006 (Figure 10b) On the contrary, veteran trees (DBH = 60 cm and above) in CGs were significantly larger than veteran trees in WPs, p-value < 0.0001 (Figure 10c). Pole-sized trees (DBH = 10-30 cm) were statistically similar (p-value = 0.396) across different types of urban greenspaces (Figure 10a).

Tree height measurements were also significantly different across different types of urban greenspaces. Trees in RPs that were less than 6 m in height were significantly taller than those in CGs, p-value = 0.001 (Figure 10d). For trees with height between 6 to 12 m, WP trees were found to be significantly taller than CG and RP

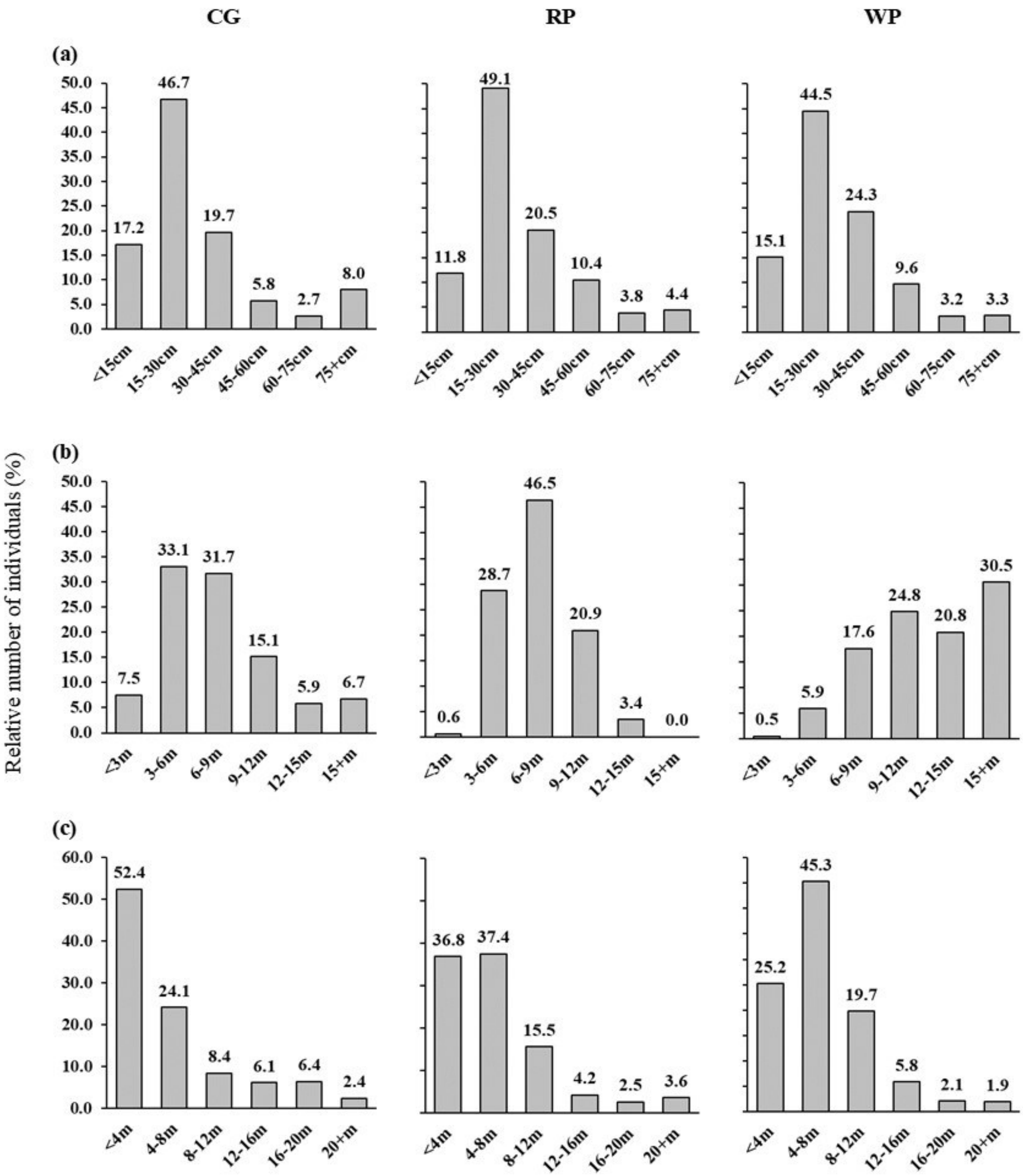



**Figure 9.** Histogram showing the distribution of urban trees into diameter classes, height classes, and crown spread classes across different types of urban greenspaces.

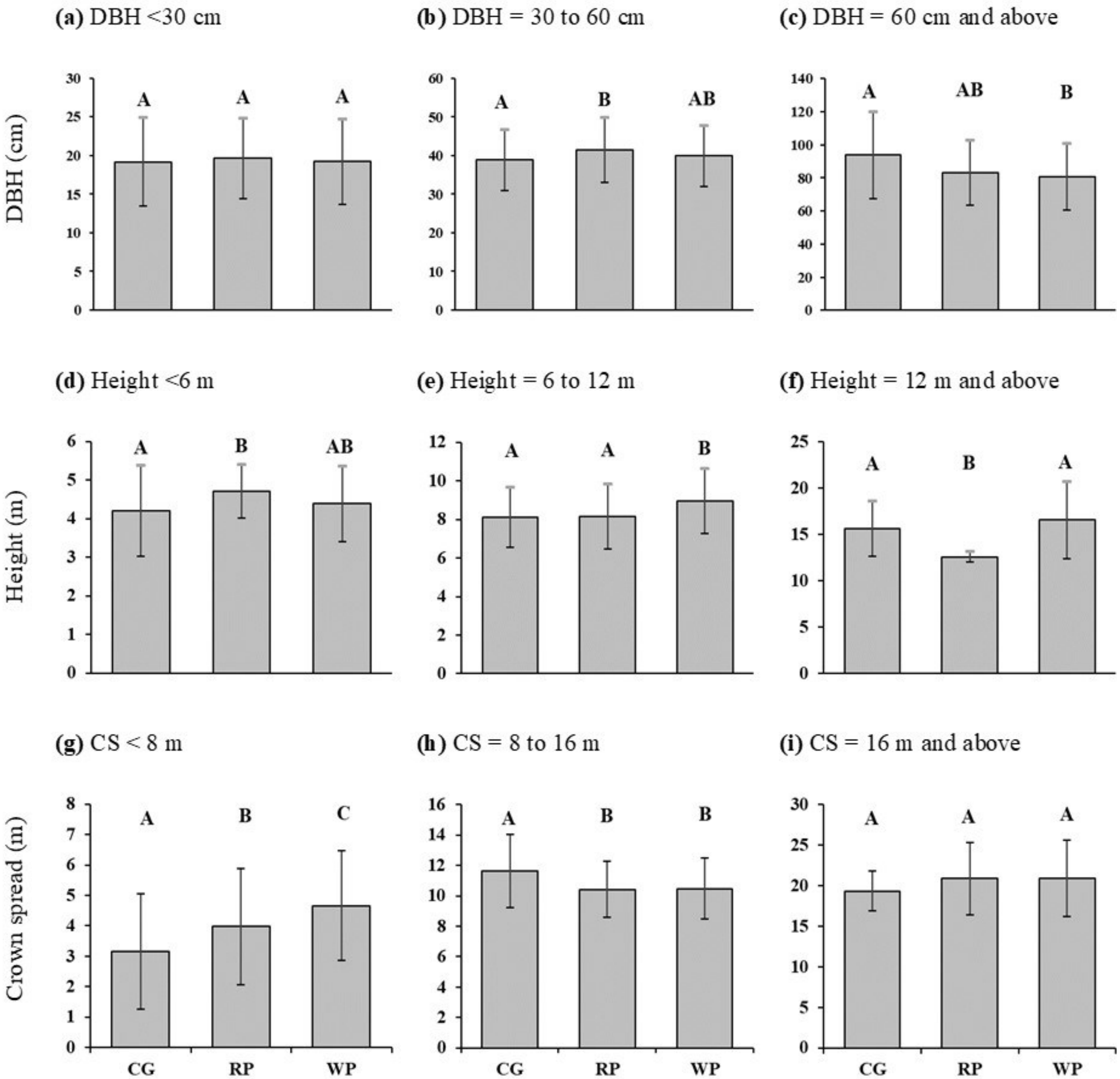



**Figure 10.** Average measurements of tree DBH, height, and crown spread in different types of urban greenspace.

trees, p-value < 0.0001 (Figure 10e). For the tallest height category of 12 m and above, RP trees measured significantly shorter than CG and WP trees, p-value < 0.0001 (Figure 10f).

Crown spread of trees were also significantly different across the three types of urban greenspaces. The comparison of trees with small crown spread (< 8m), showed that WP trees have significantly larger crown spread than CG (p-value < 0.0001) and RP (p-value = 0.001) trees; and that RP trees had significantly larger crown spread than CG trees (p-value < 0.0001) (Figure 10g). Smaller trees in the understorey of WPs tend to develop more branches to capture more light from the dense canopy of larger trees, hence larger crown spread. The larger crown spread of RP trees compared to CG trees could be attributed to species selection and management practices in RPs that ensure ample shade needed for outdoor recreation. Among trees with medium crown spread of 8 m to 16 m, CGs were found to have significantly larger crown spread than WP trees (p-value < 0.0001) and RP trees (p-value = 0.001) (Figure 10h). This could be due to intensive competition in space in the canopy layer of WPs and RPs, which limits crown expansion. There was no significant difference among trees with large crown spread that extend up to 16 m and above (Figure 10i), probably due to absence of competition among overstorey trees that allow full expansion of tree crowns.

## CONCLUSION

The study found that urban greenspaces were dominated by few tree species and botanical families. The most abundant and dominant tree species varied across different types of urban greenspaces. Overall, urban tree species composition was a mix of native and exotic species, some of which have threatened local and global existence. The proportions of native and threatened species were associated with the type of urban greenspace. More native and threatened tree species could be found in greenspaces intended for wildlife conservation.

Species diversity also varied in different types of greenspaces. High alpha and beta diversity indices can be expected from greenspaces where trees are integrated as a primary component (wildlife parks), than in areas where trees are only used to complement urban infrastructures (commercial greenspaces) or support social activities (recreational parks).

The study also found associations between tree structural patterns and type of urban greenspace. The diameter and crown spread of urban trees were generally small, especially in commercial greenspaces. On the contrary, different height distribution patterns were observed across different types of urban greenspace, such that short stature trees are more abundant in commercial greenspaces, while higher proportion of taller trees could be found in wildlife parks.

The heterogeneity of urban greenspaces across different land uses should be recognized by greenspace managers and be used as basis in designing appropriate greenspace management strategies. Urban greenspace composition and structure should be regularly checked to maintain a healthy and stable urban ecosystem that can provide maximum environmental services. The factors that influence urban tree composition, diversity, and structure should be studied further to identify entry points for management interventions. Further studies should also cover other common types of urban greenspaces to fully understand the influence of different urban land uses on urban tree patterns.

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