Urban Forestry & Urban Greening 101 (2024) 128530

Contents lists available at ScienceDirect

Urban Forestry & Urban Greening

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

Street tree communities reflect socioeconomic inequalities and legacy effects of colonial planning in Nairobi, Kenya

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A R T I C L E I N F O

Handling Editor: Dr Cecil Konijnendijk van den Bosch

Keywords:
Community composition
Environmental justice
Green space
Urban biodiversity
Urban forests

A B S T R A C T

Street trees provide ecological and social benefits that sustain urban life, yet their distribution frequently mirrors socioeconomic inequalities, meaning underprivileged social groups also have fewer trees. The distribution patterns of urban trees are least documented in the cities of rapidly evolving middle-income countries. We assessed whether street tree abundance, size, condition, diversity, and composition vary across census-derived socioeconomic strata in Nairobi, Kenya. We sampled 2047 trees across 12 neighborhoods, covering 24 km of street, then used linear regression to understand how socioeconomic strata relate to street tree characteristics. We found substantial disparities in tree abundance, with affluent areas harboring 91.5 % of the trees sampled. Low-income areas and informal settlements had comparably few trees. Mean diameter and condition did not vary across socioeconomic strata, but high-income, formerly European neighborhoods had a higher proportion of small trees, indicating a bias in recent urban greening investments further benefiting these areas. Species diversity followed a similar pattern of inequality. High-income neighborhoods had over 30 % higher species richness and diversity than low-income areas. Even so, lower income neighborhoods exhibited greater differences in street tree community composition, and a higher proportion of trees that bear edible fruit or are used in traditional medicine. Overall, our results reveal pronounced spatial inequality in the distribution of street trees in Nairobi, reflecting not only socioeconomic differences but the enduring legacies of colonial planning.

1. Introduction

Urban trees are largely recognized as critical to enhancing urban sustainability, climate resilience, and amenity (Venter 2020). Street trees, despite accounting for a fraction of trees present in public parks and private gardens, are the most common form of urban trees people encounter, as a function of their placement in the cityscape (Heynen et al., 2006). Becoming a standard feature in European cities by the mid-17th century (Woudstra and Allen, 2022), as well as in cities colonized by European settlers (Nagendra and Gopal, 2010), street trees remain a fixture of present-day urban tree planting initiatives (Myers et al., 2023).

In recent decades, street trees have gained prominence in urban forest research for their contributions to social and ecological well-being (Coleman et al., 2022). Although street trees are regarded as one layer of the urban forest rather than an ecosystem in their own right (Bolund and Hunhammar, 1999), a substantial body of evidence has emerged on the ecosystem services street trees provide. Among these, particular emphasis has been placed on the ability of street trees to regulate specific ecosystem functions (Coleman et al., 2022). These regulating functions include cooling (Salmond et al., 2016; Oke et al., 1989), reducing stormwater runoff (Gonzalez-Sosa et al., 2017), and sequestering carbon (Boukili et al., 2017), with mixed evidence regarding air quality (Eisenman et al., 2019). Street trees also deliver cultural

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https://doi.org/10.1016/j.ufug.2024.128530
Received 12 June 2024; Received in revised form 21 September 2024; Accepted 27 September 2024
Available online 2 October 2024

ecosystem services, or less tangible benefits linked to human wellbeing, with reports of positive associations between street trees and physical (Reid et al., 2017) and psychological health (Taylor et al., 2015), as well as traffic safety (Reid and Dumbaugh, 2009). The benefits urban populations derive from street trees vary across contexts. In low- and middle-income countries, provisioning services, or direct material outputs like traditional medicine, food, and fuelwood, tend to carry more importance, particularly for poorer households (Adeyemi and Shackleton 2024b). A subset of the literature focuses on the valuation of these ecosystem services, quantifying the benefits and associated costs of street trees in monetary terms. Economic benefits related to property value, carbon abatement, and energy savings tend to be the most studied, and outweigh the cost of planting and maintenance (Song et al., 2018; Wang et al., 2018). Benefits in terms of water regulation and air quality appear more modest, while benefits related to biodiversity and provisioning services, both particularly relevant to cities in the tropics, have received little attention (Song et al., 2018). Notwithstanding their numerous benefits, street trees can also bring disservices, often due to insufficient environmental and social safeguards (Shah et al., 2022), and may have unintended outcomes like gentrification (L. Li, 2023).

As a shared public amenity, street trees can remedy more structural inequity in access to parks and private gardens, yet their distribution is often uneven across different social groups (Ferguson et al., 2018). Inequalities related to income have been most documented, with a strong positive association between street tree abundance and affluence (Brooks et al., 2016; Kirkpatrick et al., 2011; Landry and Chakraborty, 2009; J. Lin et al., 2021; E. C. Anderson et al., 2023). Inequalities have also been revealed based on other socioeconomic variables like housing tenure (Landry and Chakraborty, 2009; W. Lin and Güneralp, 2024) and education (X. Li et al., 2015; Kendal et al., 2012), as well as demographic characteristics like ethnicity (Landry and Chakraborty, 2009; J. Lin et al., 2021) and householder age (Pena et al., 2024; X. Li et al., 2015). Research shows that underprivileged populations (e.g., low-income residents, renters, or minorities) are typically associated with lower street tree cover (J. Lin et al., 2021). This implies diminished access to the benefits street trees provide, rendering disparities in their distribution an issue of environmental justice (Landry and Chakraborty, 2009). Recent studies contend that inequities in street tree distribution are more strongly a product of historical processes than of current socioeconomic stratification (Roman et al., 2018). Several analyses highlight the legacy effects of institutionalized social injustice on street tree communities, like redlining (Burghardt et al., 2023) and colonial planning (Shackleton and Gwedla, 2021; Venter et al., 2020).

Disparities in street tree abundance and diversity are most extensively documented in North America, where the largest number of studies at the nexus of urban forestry and environmental justice have taken place (Krajter Ostoić and Konijnendijk van den Bosch, 2015; Calderón-Argelich et al., 2021). Cities in low- and middle-income countries are comparatively under-represented in the literature, despite experiencing higher rates of urbanization (IPCC, 2023), levels of urban inequality (Zhou et al., 2022), including greenspace exposure inequality (Chen et al., 2022) and climate-change induced risks (Chaudhry, 2024), but report similar patterns of unevenness. Studies conducted in Bogota, Colombia (Brown, 2012), Karachi, Pakistan (Shams et al., 2020), Lagos, Nigeria (Adeyemi and Shackleton 2024a), Belo Horizonte, Brazil (Pena et al., 2024), and the Eastern Cape in South Africa (Kuruneri-Chitepo and Shackleton, 2011; Gwedla and Shackleton, 2017) highlight the prevalence of lower street tree densities in lower income areas compared to their wealthier counterparts. In some cities, disparities in street tree abundance are more pronounced, with many streets in poorer neighborhoods having no trees at all (Brown, 2012; Shams et al., 2020; Gwedla and Shackleton, 2017). Whereas income is typically positively correlated with street tree density, findings for species diversity vary. While higher species richness was reported in affluent areas for Lagos, Nigeria, and Eastern Cape towns in South Africa (Adeyemi and Shackleton 2024a; Shackleton and Gwedla, 2021; Kuruneri-Chitepo and Shackleton, 2011), the opposite was true in Karachi, Pakistan (Shams et al., 2020). In Belo Horizonte, Brazil, no relationship between species richness and income was evident (Pena et al., 2024).

Nairobi comes from the Maasai phrase 'Enkare Nyirobi', which translates to "place of cool waters", owing to the rivers running through the area from which the city developed. Nairobi also became known as "the green city in the sun," in reference to the city's blend of forest, wetlands, and savanna (Cherotich and Maamun, 2022). From 2001–2023, Nairobi lost 450 ha of tree cover, or nearly a tenth of its forest (Global Forest Watch, 2024). Urban development is the primary driver of forest conversion, although land grabbing also played a hand (Makworo and Mireri, 2011; Ndungu, 2004). Perhaps driven by a history of mobilization against these forces (Manji, 2017; Njeru, 2010), research on Nairobi's urban forest has primarily focused on forested natural areas, with an emphasis on their management and conservation status (Manji, 2017; Olooet al., 2020; Binyanya et al., 2022; Chisika and Yeom, 2023), as well as their conservation value (Furukawa et al., 2016; Nyambane et al., 2016; Furukawa, 2011). While these provide critical baseline information amidst rapid urban development (Mundia and Aniya, 2006), information remains scarce concerning the state of Nairobi's urban forest, and street trees in particular. One notable exception is a comparative study that measures street tree features using satellite imagery (Liang et al., 2023). Moreover, the issue of distribution and access to urban trees has not been raised, despite noticeable spatial disparities in Nairobi (Maganga, 2021).

Nairobi's spatial heterogeneity can be traced back to the city's origins as a railway outpost. To control the town's growth, ethnic composition, and, officially, prevent disease outbreaks, the British colonial administration sanctioned racial segregation. Europeans occupied the western areas of higher elevation and land values, while Asians, mostly Indians recruited to build the railway, predominantly resided in the northern areas, and Africans were relegated to the densely populated areas southeast of the Central Business District (CBD). (K'Akumu and Olima, 2007; Martin and Bezemer, 2020; Wanjiru-Mwita and Giraut, 2020). Public housing being insufficient and residence for Africans being contingent on employment status, informal settlements, or unplanned, improvised human settlements, developed in the eastern fringes. After independence in 1963 and the lifting of movement restrictions, informal settlements proliferated under the impetus of newcomers in search of employment (Martin and Bezemer, 2020; K'Akumu and Olima, 2007). Within the first decade of independence, residential segregation based on race was reclassified based on income, with implications on urban form. Still today, western areas are characterized as affluent and low-density, and eastern locations as lower income and higher density (Abascal et al., 2022; K'Akumu and Olima, 2007).

In this study, we examine the relationship between contemporary social strata and current urban street tree abundance, diversity, and community composition to quantify and qualify street tree inequalities in Nairobi. Specifically, we ask whether areas in different socioeconomic strata exhibit comparable levels of street tree abundance, size, and condition. Tree size, a proxy for age, serves as a temporal marker reflecting investment in urban greening (Burghardt et al., 2023; Roman et al., 2018). We also compare alpha (α) diversity, or species richness and relative abundance within specific socioeconomic strata, and beta (β) diversity, or compositional differences between strata (Whittaker, 1972). We predict that street trees in lower socioeconomic strata will exhibit: (1) a lower abundance of both small and large trees, (2) lower α-diversity, (3) and distinct species composition that more heavily weighs provisioning ecosystem services compared with more affluent neighborhoods.

2. Methods

2.1. Site description

The area Nairobi (-1.2863°S, 36.8172°E) currently occupies in Kenya (Fig. 1a) formed around a wetland. Situated between 1500 and 1850 m above sea level (Situma et al., 2007), Nairobi has a subtropical highland climate (Kenya Ministry of Environment, 2013), with a mean temperature of 18.9° C and mean annual precipitation of 913 mm (World Bank, 2021). Similar to much of the country, Nairobi has two rainy seasons, the "long rains," typically occurring between March and May, and the "short rains" between October and December (Camberlin and Wairoto, 1997).

The area was inhabited by a complex tribal and ethnic mix including Akamba, Kikuyu, Maasai, and other Bantu and Nilotic peoples (Ogot and Ogot, 2020) before British colonial authorities established Nairobi in 1899 as a railway depot on the line connecting Uganda to the port city of Mombasa. Nairobi became a strategic enclave of the East Africa Protectorate, eventually supplanting Mombasa as its capital in the early 1900s (Wolff, 1974). Nairobi's growth was spurred by administrative and trade activities and remained the center of governance after Kenya's independence. Today, Nairobi City County, as officially renamed in 2010 (henceforth referred to as Nairobi or city), extends over 696 km^2 and is one of the largest cities in East Africa, with over 4.4 million people (Kenya National Bureau of Statistics, 2019). The city's population grows approximately 4 % per annum, and three quarters of new residents are absorbed into informal settlements, where over half of Nairobi's population already lives (Da Cruz, 2006).

2.2. Sampling design and measurements

Street trees were inventoried between June and August 2023, with physical visits to 120 locations across 12 neighborhoods, evenly distributed in four socioeconomic strata. Nine neighborhoods were selected based on a living conditions score (LCS) developed by the Global Development Institute for the Kenyan government (Fig. 1b). The LCS is a proxy for income derived from the 2009 Kenya Census, with information measured at the household level on physical living conditions and services (e.g., dwelling materials, water and energy source), assets and livestock, and household composition and human capital (e.g., education, employment, health). The LCS also takes in geographic information, measured at the smallest administrative unit level, including population, death and birth rates, and climatic variables, for a total of 177 variables. Using principal component analysis, a score was calculated based on the correlations among the variables, weighted based on their contribution to the final score, on a scale 0 – 100 specific to Nairobi, with 0 denoting the poorest household and 100 the wealthiest (Villa, 2016). Neighborhoods developed over comparable timelines, and only residential areas were retained. Neighborhoods with an LCS $\geq$ 90 were categorized as high-income (Karen, Spring Valley, Upper Parklands), 70 – 89 as middle-income (Kenyatta, Highridge, Kilimani), and below 70 as low-income (Donholm, Umoja, Eastleigh), following field visits conducted to visually corroborate the classification based on the 2009 census. Informal settlements (Kibera, Mathare, Mukuru) were selected separately, based on partnerships developed with community groups working in these areas.

The selection of streets for sampling followed a spatially stratified random sampling approach. 10 transects were sampled in each of the 12 neighborhoods, which were nested in four strata. To determine transect locations, 10 random points were generated in each neighborhood, and the nearest road intersection identified as the starting point for a transect (Nagendra and Gopal, 2010; Kuruneri-Chitepo and Shackleton, 2011). From each intersection, a transect of 200 m in length was established in a randomly chosen direction (Nagendra and Gopal, 2010), resulting in 24 km of street sampled. If the road did not extend for 200 m from the intersection, the road 90 degrees in the next cardinal direction from the same point was selected.

In each 200 m transect, we enumerated street trees on both sides of the road. We defined street trees as trees and tree-like monocots (including palms and banana trees) above 30 cm in height (Ramirez et al., 2006) within 5 m of the road. This distance accounted for the width of sewage and stormwater systems, sidewalks, and planting distance from the sidewalk (ITDP et al., 2022). This definition included trees planted and maintained by public or private actors, as well as self-seeded trees (Gwedla and Shackleton, 2017). For each tree, we recorded species, diameter, and condition. This structural information carries implications for the magnitudes of tree-derived ecosystem services and the allocation of tree management resources (J. Lin et al., 2021). We measured diameter at breast height (1.3 m) (Magarik et al., 2020), and in cases where the tree did not reach 1.3 m or forked below, we measured at 30 or 10 cm above the ground, depending on the height of the fork, and made note of the diameter measurement height. Tree condition was coded into one of five classes: (1) trees in near-perfect health with no visible damage were considered in excellent condition; (2) trees missing up to half the crown, or exhibiting minor trunk damage, but still relatively healthy, were considered in fair condition; (3) trees missing over half the crown, or presenting severe trunk damage, vine overgrowth, or suffering from a severe infection or infestation were considered in poor condition; (4) dead trees and (5) stumps were assigned separate classes. Unknown species were assigned a field name, and specimens were collected with photographs for later identification at the Museums of Kenya herbarium. All identified trees were assigned to the species level (except for trees belonging to the non-native genus Eucalyptus). To our final species list, we appended information on whether the species is indigenous to Kenya or non-native, and if it bears edible fruit or properties used in traditional medicine (POWO, 2023; Dharani, 2019).

2.3. Statistical analyses

All analyses were conducted in R version 4.4.0 (R Core Team, 2021) via RStudio. To assess potential differences in street tree diameter (for live trees), and condition across socioeconomic strata, we employed linear mixed models with the lmer() function in the lme4 package (Bates et al., 2015) to accommodate the nested study design, with random effects for neighborhood and strata as a fixed effect. For the abundance and species richness models, we used a generalized linear model with a Poisson distribution, using the glmer() function in lme4. If strata was a significant predictor, post hoc pairwise comparisons between strata were performed using the emmeans() function in the emmeans package for least-squares means, with Tukey adjustments for multiple comparisons (Russell, 2024). We ran two sensitivity analyses, one including only trees measured at 1.3 m (excluding smaller trees, and forking or multi-stemmed trees), and another including only trees larger than 10 cm, to ensure consistent results.

For all analyses related to diversity, we included only live, identified trees. To visualize α-diversity across socioeconomic strata, we constructed individual-based species rarefaction and extrapolation curves using the iNext package (Chao et al., 2014; Burghardt et al., 2023). Curves were generated for species richness, Shannon's diversity index, and Simpson's inverse diversity index, which are commonly used to characterize two aspects of α-diversity: number of species and relative abundance of species (Magurran 2004). To account for potential differences in diversity stemming from variations in tree abundance, we standardized comparisons across strata with rarefaction to the smallest number of individual street trees observed within a strata ($n = 65$).

To assess differences in species composition between strata, or β-diversity, we first conducted nonmetric multidimensional scaling (NMDS) using the metaMDS() function from the vegan R package (Oksanen et al., 2022). We used NMDS, a distance-based ordination technique, to graphically summarize compositional differences between neighborhoods in a two-dimensional plane (Bakker, 2024). Next, we ran

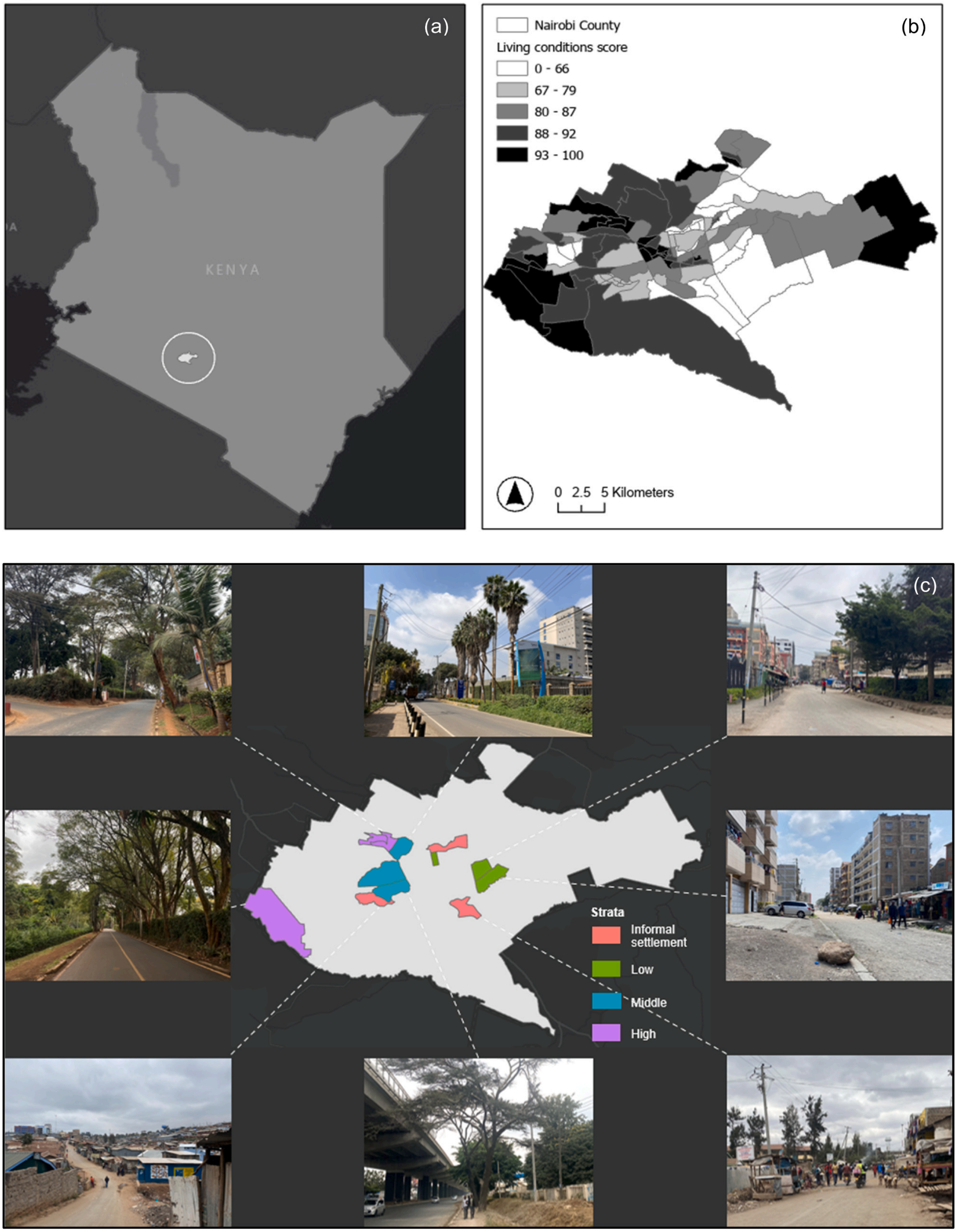

Fig. 1. : Site. (a) Location of Nairobi within Kenya. (b) Nairobi neighborhoods with living conditions scores, based on the 2009 Kenya Census (Villa, 2016). (c) Study area comprising 12 neighborhoods categorized into four socioeconomic strata, with photographs from sample transects.

a permutational multivariate analysis of variance (PERMANOVA) using the adonis() function in the vegan package (M. J. Anderson, 2017; Stevens and Oksanen, 2022), to test whether compositional differences between neighborhoods within strata were greater than differences between neighborhoods across strata. For the NMDS and PERMANOVA, we used Bray-Curtis dissimilarity, which factors in both the presence or absence of species and their relative abundance (Magurran 2004). To identify which components of β-diversity were driving compositional differences, we examined species turnover, or changes in species presence, and reordering, or changes in species ranks, between neighborhoods within each strata (Avolio et al., 2019). To visualize differences in species dominance, we generated rank abundance curves for each strata, with the five most abundant species labelled. To quantify turnover and reordering, we compared rank abundance curves for each neighborhood in pairs using RAC_difference() from the codyn package (Hallett et al., 2016). Finally, to test for differences across strata, we ran a one-way ANOVA with strata as a fixed effect (Burghardt et al., 2023), and compared the proportion of fruit and medicinal trees across strata using Fisher's exact test in the fmsb package (Nakazawa, 2024).

3. Results

3.1. Abundance, size, and condition

Street tree abundance markedly differed between socioeconomic strata. Of the 2047 trees enumerated, 91.5 % were located within the two more affluent residential areas (middle- or high-income neighborhoods), with the remainder distributed between the two lower income areas (low-income areas and informal settlements). All transects in affluent areas contained trees, compared to only half in lower income areas. The mean count of street trees per transect differed across strata (conditional $R^2 = 0.939$, $p < 0.001$), with progressively sparser transects from high-income neighborhoods to informal settlements (Fig. 2a). Pairwise comparisons revealed differences between all strata ($p < 0.0001$ for each pair), except between informal settlements and low-income areas, and between middle- and high-income areas.

Although there was no significant difference in the mean diameter of live trees, several observations in diameter range and distribution are worth noting. First, the upper diameter limit was 71 cm for low-income and 77 cm for informal settlements, while higher income areas boasted trees exceeding 200 cm in diameter (Fig. 2b). Second, most trees in lower income areas fell in the mid-size diameter range of 10–50 cm, whereas higher income areas both harbored approximately 46 % of

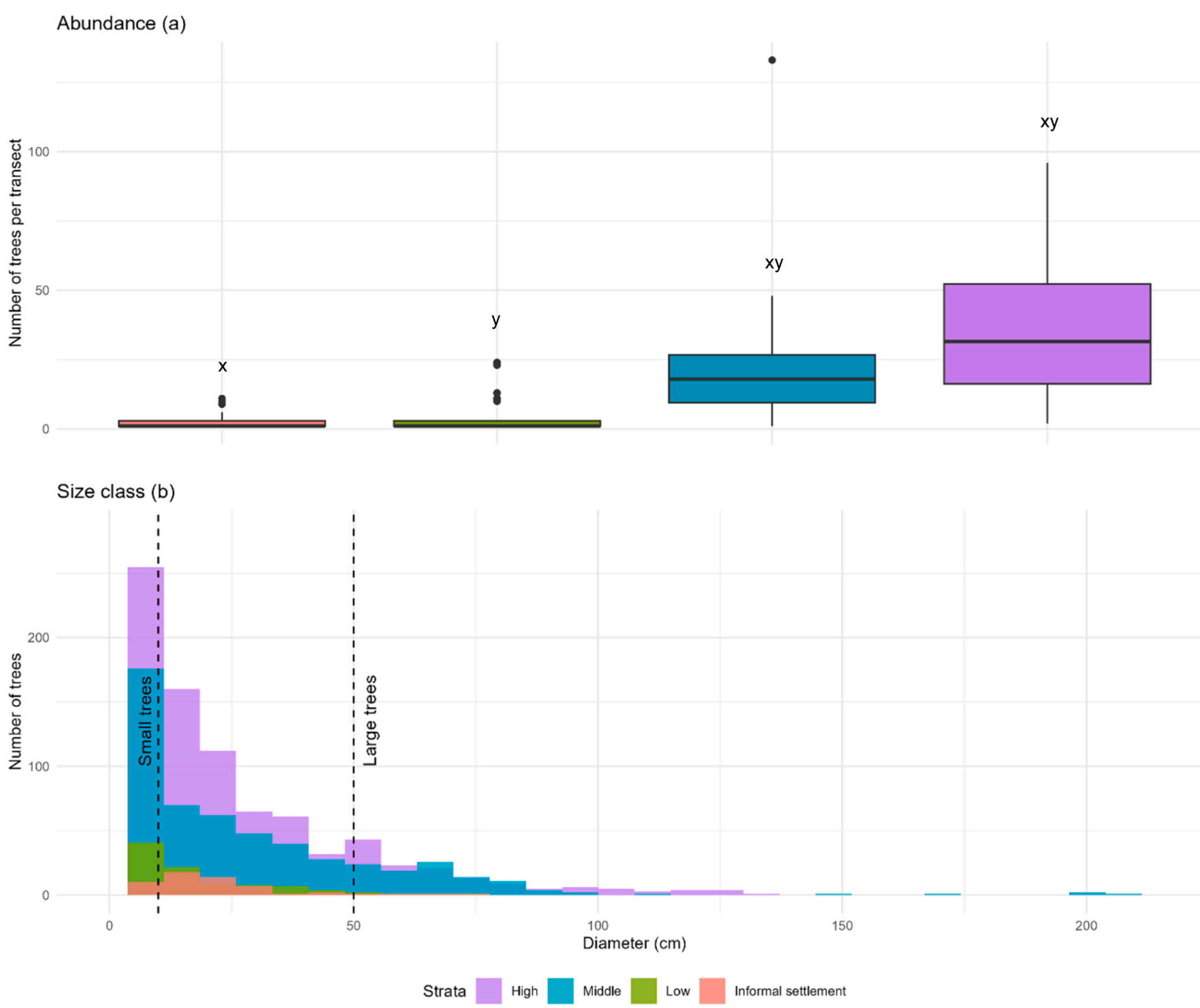

Fig. 2. : Street tree abundance and size class across 12 neighborhoods in Nairobi, Kenya in 2023. (a) Boxplot of number of street trees per transect for each socioeconomic strata. Differences (p < 0.05) are denoted by letters based on linear mixed models; (b) Frequency distribution of tree diameters for each strata. The dotted lines indicate the diameter cutoffs used to classify "small" and "large" trees.

small (young) trees with a diameter ≤ 10 cm, along with over 10 % large trees with a diameter > 50 cm (Table 1). No significant difference was observed in the mean condition of street trees across socioeconomic strata. High-income neighborhoods had the highest proportion of trees in excellent condition (48 %), as well as the highest proportion of dead trees and stumps (5 %), both likely related to having more young and old trees, whereas informal settlements had the highest proportion of trees in poor condition (17 %), more likely due to environmental factors (Table 2).

3.2. Gamma (γ) diversity: total diversity at city scale

A total of 142 species were identified from 46 families (Appendix A). The species distribution was strongly right skewed, with the most prevalent species, *Jacaranda mimosifolia*, constituting 7 % of the total street tree population, and the 10 most common species collectively representing 41 % of the entire tree sample (Table 3). Conversely, 25 % of the species were represented by only one stem.

Of the total species, 65 % were non-native, collectively accounting for 69 % of stems. Alongside *Jacaranda*, the most common non-native species were *Chrysalidocarpus lutescens* (golden cane palm) and *Persea americana* (avocado), while the most common indigenous species were *Filicium decipiens* (fern-leaf tree or thika palm) and *Croton megalocarpus* (silver-leaved croton).

3.3. Alpha (α) diversity: street tree diversity within socioeconomic strata

Mean species richness per transect differed across strata (conditional $R^2 = 0.651$, $p < 0.001$), based on a generalized linear mixed model. Pairwise comparisons revealed differences only between inconsecutive strata, i.e., between informal settlements and middle- and high-income areas ($p = 0.001$ and $p < 0.0001$), and between low-income and high-income areas ($p < 0.0001$). Extrapolating diversity metrics to an equivalent number of live, identified street trees, individual-based accumulation curves depicted highest diversity in high-income areas across diversity metrics (Fig. 3a). Moreover, the absence of an asymptote in species richness for high-income areas suggests that more extensive sampling would likely reveal additional species. When rarified to the minimum observed strata sample size (n = 65), high-income areas had 31 % higher species richness, 35 % higher Shannon's effective number of species, and 37 % higher Simpson's effective number of species relative to low-income areas, which had lowest species richness for an equivalent number of trees, and lowest diversity for an equivalent number of species.

3.4. Beta (β) diversity: street tree diversity between socioeconomic strata

Differences in species composition between socioeconomic strata, or β-diversity, also differed significantly ($p = 0.001$), based on a PERMANOVA. Compositional differences were predominantly driven by reordering, or changes in species ranks, rather than turnover, or changes in species, which suggests lower income areas harbor a subset of the species present in higher income areas. Rank abundance curves revealed distinct dominant species across strata (Fig. 3c). In high- and middle-income strata, *Jacaranda mimosifolia* and *Chrysalidocarpus lutescens*, both exotic ornamentals, dominated the landscape. In low-income areas, *Leucaena leucocephala,* another non-native species, prevailed. Considered an invasive species in many countries, including Kenya (Bakewell-Stone, 2023), *Leucaena* is also an important alternative protein source to fodder for feeding livestock (De Angelis et al., 2021). In informal settlements, *Markhamia lutea*, a common indigenous tree used in traditional medicine, was dominant. Moreover, the proportion of trees bearing fruit or used in traditional medicine differed across strata ($p = 0.01$ and $p < 0.001$ respectively), based on Fisher's Exact Tests. Informal settlements had the highest proportion of fruit trees (29 %), and low-income areas had the highest proportion of medicinal trees (38 %). Within strata, street tree communities in higher income areas exhibited greater resemblance to one another than those in lower income areas. This was visually confirmed by the NMDS ordination plot, where high-income neighborhoods formed more tightly clustered groups than lower income neighborhoods (Fig. 3b). Furthermore, reordering within strata significantly differed ($p < 0.01$) based on an ANOVA, with less reordering across neighborhoods in higher income areas. In contrast, each neighborhood in the low-income strata had a different ordering of dominant and rare species. We found no differences in species turnover ($p = 0.08$), based on the Jaccard index, which suggests that within strata, street tree communities had similar species.

Table 1
Diameter of live street trees across socioeconomic strata.

Strata	≤ 10 cm	10.01 – 50 cm	≥ 50 cm
Informal settlement (n = 65)	17 (26.2 %)	44 (67.7 %)	4 (6.2 %)
Low (n = 105)	41 (39.0 %)	60 (57.1 %)	4 (3.8 %)
Middle (n = 687)	313 (45.6 %)	272 (39.6 %)	102 (14.9 %)
High (n = 1109)	509 (45.9 %)	471 (42.5 %)	129 (11.6 %)

Table 2
Condition of street trees across socioeconomic strata.

Strata	Excellent	Good	Poor	Dead or stump
Informal settlement (n = 65)	21 (32.3 %)	33 (50.8 %)	11 (16.9 %)	0 (0.0 %)
Low (n = 108)	25 (21.1 %)	64 (59.3 %)	16 (14.8 %)	3 (2.8 %)
Middle (n = 702)	256 (36.5 %)	347 (49.4 %)	84 (12.0 %)	15 (2.1 %)
High (n = 1172)	560 (47.8 %)	431 (36.8 %)	118 (10.1 %)	63 (5.4 %)

Table 3
Attributes of the 10 most frequently encountered species across all socioeconomic strata, based on a sub-sample survey of live, identified street trees in Nairobi (n = 1966).

Scientific name	Proportion of trees (across all strata)	Common names in English	Origin	Uses
Jacaranda mimosifolia	7.0 %	Blue jacaranda	Non-native	Ornamental, timber, fuel
Filicium decipiens	4.8 %	Fern tree, thika palm	Indigenous	Ornamental, fuel
Chrysalidocarpus lutescens	4.6 %	Golden cane palm, areca palm	Non-native	Ornamental
Croton megalocarpus	4.6 %	Silvery leaved croton	Indigenous	Ornamental, medicinal, timber, fuel, animal fodder
Persea americana	4.0 %	Avocado	Non-native	Edible fruit
Grevillea robusta	3.8 %	Silky oak	Non-native	Ornamental, timber, fuel, animal fodder
Eriobotrya japonica	3.1 %	Loquat	Non-native	Ornamental, edible fruit, fuel
Dracaena steudneri	3.0 %	Dragon tree	Indigenous	Ornamental, medicinal, fuel
Archontophoenix alexandrae	3.0 %	Alexander palm	Non-native	Ornamental
Cupressus arizonica	2.9 %	Arizona cypress	Non-native	Ornamental

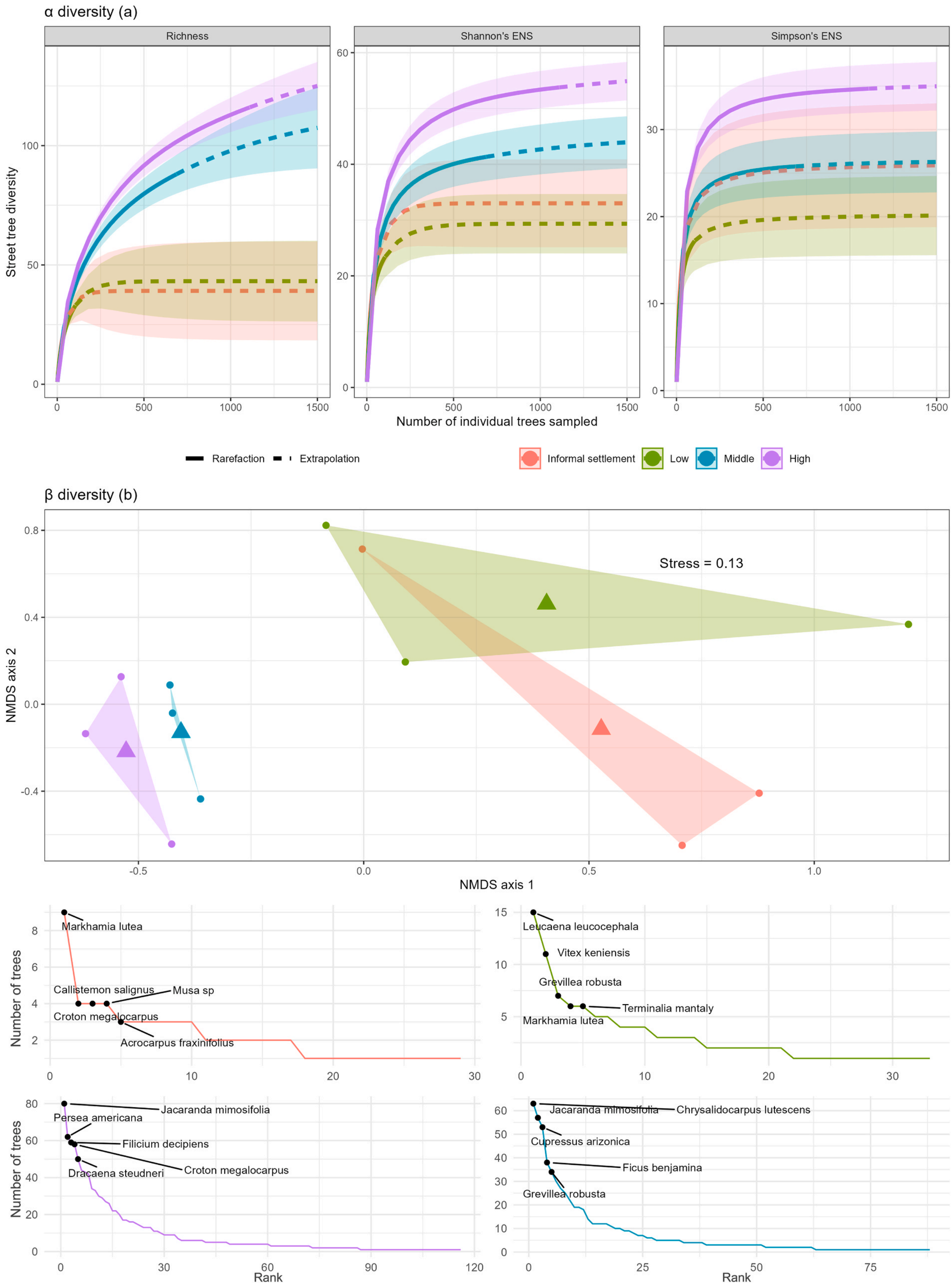

Fig. 3. : Street tree diversity and community composition across 12 neighborhoods in Nairobi, Kenya in 2023. (a) Individual-based species accumulation curves extrapolated to 1500 trees, based on diversity metrics that vary in the weight attributed to number of species and relative abundance. (b) Nonmetric multidimensional scaling (NMDS) of street tree communities in neighborhoods represented by points. Community composition of points closer to one another are likely to be more similar than those further apart. (c) Rank abundance curves of street trees for each socioeconomic strata, with the five most abundant species labeled.

4. Discussion

Given the social and environmental benefits street trees provide, we set out to examine how the distribution and diversity of street trees relate to socioeconomic strata. Our findings substantiated our hypothesis that street trees are disproportionately more abundant in higher income relative to lower income areas, and more diverse. We found that street tree communities differed in composition, driven by changes in dominant species, and that lower income areas had proportionally more trees delivering provisioning services.

The areas surveyed exhibited pronounced spatial inequality in the street tree abundance, which increased with affluence, adding to global findings that street tree cover mirrors social disparities. The severity in inequality resembled that reported in other African (Kuruneri-Chitepo and Shackleton, 2011; Adeyemi and Shackleton 2024a), South Asian (Shams et al., 2020), and South American cities (Pena et al., 2024), where most streets in poorer areas have no trees. Low-income areas and informal settlements had a comparable paucity of trees, which was surprising given their differences in density and infrastructure.

Several theories have been proposed to explain the mirroring effect of street trees on socioeconomic conditions (J. Lin et al., 2021; Kendal et al., 2012). These theories may however insufficiently account for the lag time between tree planting and maturation, that is, overlook the impact of decisions made in the past on present-day street tree communities. In recent years, environmental justice research has contributed a historical counterweight, by casting current street tree disparities in the light of past systematic social injustice (Roman et al., 2018), including colonial planning (Shackleton and Gwedla, 2021; Kuruneri-Chitepo and Shackleton, 2011) and institutionalized segregation (Burghardt et al., 2023). Studies from South Africa (Shackleton and Gwedla, 2021; Kuruneri-Chitepo and Shackleton, 2011) and the United States (Burghardt et al., 2023) have explored how historically White residential areas have more street trees than predominantly Black neighborhoods. In our survey of Nairobi, Karen, Spring Valley, and Kilimani had the most trees. All were former European enclaves. Present-day Karen and Spring Valley, developed in the late 1920s from farmland and estates, are two of the wealthiest neighborhoods, with a low-density population, and a high proportion of homeowners and expats. Kilimani was also a low-density residential area until the 1990s, when an influx of high rise development and commercial activity transformed it into a mixed-use neighborhood (Rutto, 2009). Upper Parklands and Highridge had approximately half the number of trees. Developed in the 1930s, these neighborhoods formed part of the Asian area north of the CBD. Today, they make up a lively, well-heeled residential and commercial district, and retain a significant population of people of Asian descent. We observed the smallest street tree population in Eastleigh. One of the oldest neighborhoods in the city, Eastleigh has a rich history of habitation by different groups. Originally a European estate, it became an Asian and African settlement, and is presently a predominantly immigrant, low-income neighborhood (Rahbaran and Herz, 2014). Built on a grid, Eastleigh has the highest building and population density of the low-income neighborhoods surveyed. Informal settlements also had very few trees. Kibera is in the west and developed in the 1910s, whereas Mathare originated in the northeast in the 1920s, and Mukuru emerged after independence in the southwest. Existing in pockets on both sides of Nairobi's west-east divide, on the outskirts of the city and of high-income residential areas, informal settlements vary considerably, but generally share very high population and building densities, few streets, and limited public infrastructure (Ren et al., 2020). Street trees in these areas were primarily planted by social institutions like schools, health centers, and police stations around their premises.

The street tree community within the neighborhoods sampled is highly diverse. Broad-leaved trees largely dominated our sample, a finding consistent with Liang et al.'s analysis in which broad-leaved trees made up 86 % of the street canopy measured in 2022 using satellite imagery (Liang et al., 2023). We found the second and third most frequent functional groups were palms, which are common for residential and commercial complexes in high-income neighborhoods, and conifers. Liang et al. found "banana-like trees" to be second most abundant, followed by palms and conifers. The larger sample size and area, consisting in 12,229 trees covering 684 km^2, resolution of the GSV images, and classification based on leaf shape, may explain these differences. With 142 species found from 2047 trees, Nairobi's street tree population has higher species richness compared to most major cities in middle-income countries discussed in the literature. In Bangalore, Nagendra and Gopal (2010) recorded 108 species from 2399 street trees, and in Karachi, 62 species were counted from 6507 street trees (Shams et al., 2020). In Lagos, Adeyemi and Shackleton (2024) enumerated 46 species from 4017 street trees, and in Kumasi, 70 species were identified from 1101 street trees (Uka and Belford, 2016). A larger database of 250, 000 street trees contained 559 species in Belo Horizonte, Brazil (Pena et al., 2024).

These intercity differences in street tree diversity may stem from differences in research methods. Sampling criteria for street trees vary widely in size and mode of establishment. Apart from Pena (2024), where a city database of planted trees was available, all trees on either side of the street were sampled, with varying minima for height (none to $\geq$ 5 m) and diameter (none to $\geq$ 10 cm). There are also biophysical and sociocultural drivers for intercity differences in street tree diversity (Galle et al., 2021), including nursery supply and resident preferences (Avolio et al., 2018), both of which may favor high species diversity in Nairobi. The city is famous for its roadside nurseries, whose vendors pay a fee to the County to use the public green spaces and waterways, which they also help to conserve (Patinkin, 2013). In addition, although a permit from the County is required to cut down a street tree (Nairobi City County, 2019), many groups are involved in planting them. In contrast with the dominant North American model in which the city assumes planting and maintenance responsibilities for the street tree layer (Eisenman et al., 2020), in Nairobi, private actors, including real estate developers, contracted landscapers, social institutions, and residents, complement public agencies by planting trees surrounding their properties, on sidewalk strips, and in road dividers. These groups largely act independently, and may contribute to Nairobi's streets reaching private garden levels of species diversity (Hutt-Taylor and Ziter, 2022). Government-informed handbooks for street tree planting (Kilongasi et al., 2020; ITDP et al., 2022) exist but are directed at road developers and lack recommendations for suitable species. Similarly, species-site matching tools (Chisika and Yeom, 2023) are designed for restoration and agroforestry projects, rather than urban greening.

The disparity in street tree diversity mirrored abundance, despite controlling for differences in the latter, with lower income neighborhoods exhibiting lower street tree α-diversity than wealthier neighborhoods. For the same number of live trees, low-income areas had the least species richness, and for the same number of species, they had the least even pool. Despite having lower species richness, street tree communities in lower income areas had lower resemblance. Because of how few trees these communities contain however, their uniqueness may result from their propensity to be dominated by fewer tree species. This in turn may reflect resident preferences for trees delivering provisioning services, or nursery surplus donated to social institutions. Given that street tree communities with higher diversity indices are more resilient to pest and disease outbreaks and better able to withstand sudden environmental change (Morgenroth et al., 2017), low α-diversity, coupled with low tree stock, may jeopardize the continued supply of tree-based ecosystem services to residents in underprivileged areas in the future.

Over two-thirds of street trees encountered were non-native. The predominance of non-native street tree species, many introduced during the colonial period (Shackleton and Gwedla, 2021), is common across many former colonized countries, including Brazil (Pena et al., 2024), Pakistan (Shams et al., 2020), South Africa (Gwedla and Shackleton, 2017), Ghana (Uka and Belford, 2016), and Nigeria (Adeyemi and

Shackleton 2024a). Europeans transferred species from other colonies to create a sense of familiarity and meet aesthetic expectations. Genera favored for street planting (e.g., *Grevillea, Jacaranda, Eucalyptus*) were fast-growing and tolerant of challenging urban growing conditions (Shackleton and Gwedla, 2021). While some defend that non-native, non-invasive species can increase diversity and contribute to ecosystem resilience in adverse urban environments (Chalker-Scott, 2015), contemporary urban greening guidelines tend to advocate for planting indigenous species due to concerns of invasion and biodiversity loss from non-native species (Sjöman et al., 2016). Chalker-Scott (2015) Non-native trees also raise questions of suitability by reflecting colonial standards (Shackleton and Gwedla, 2021). While non-native species are also common in former colonizing countries of Europe, their presence often stems from the desire to augment a smaller indigenous species pool rather than an external imposition (Sjöman et al., 2016).

The dominance of non-native species in the street tree population contrasts with studies on Nairobi's parks and natural forested areas. Research conducted in three of Nairobi's largest green spaces – Karura Forest, Ngong Road Forest Reserve, and City Park – reported that over 80 % of trees were indigenous (Furukawa et al., 2016; Nyambane et al., 2016). These are largely remnants of indigenous forest, and despite undergoing various human interventions like selective logging in Ngong Forest, landscaping in City Park, and partial conversion to monoculture in Karura, the parks have largely retained the original composition of the forests that once extended over much of Nairobi.

Increasing tree cover, mandated to be at least 10 % by the 2010 Constitution (Kevins, 2022) has been on the Kenyan national agenda for decades. Most recently, the national government declared November 13th, 2023, a one-off holiday dedicated to tree planting as part of a campaign to plant 15 billion trees to achieve 30 % tree cover in Kenya by 2032 (Koskei, 2022). The prevalence of small (young) trees in the city may partially reflect this objective. However, small trees were concentrated in higher income areas, where recent road widening projects may induce compensatory tree planting, and higher rates of home ownership may incentivize voluntary tree planting. In areas characterized by narrow roads, limited sidewalks, and majority renters, public investment in tree pits, site-specific species selection informed by public input, protection of root zones and foliage susceptible to browsing, and consistent post-planting maintenance could mitigate these feedbacks (Doherty et al., 2003). In areas with an established street tree canopy, safeguarding mature, large canopy trees that provide greater ecosystem services is crucial (Park et al., 2019). Nagendra and Gopal (2010) a spatial database of trees in the city is essential for monitoring tree felling, planting, and growth over time, as well as sustaining diversity in age class and species composition.

5. Conclusion

This paper contributes to the growing body of evidence that documents socioeconomic inequalities in urban trees and the ecological and social value they provide, from an under-studied region. Our findings revealed striking inequalities in street tree abundance, with higher-income neighborhoods boasting significantly more street trees, and lower income areas being nearly devoid of them. Concurrently, we observed lower street tree diversity and distinct compositional differences in lower income neighborhoods, characterized by a higher prevalence of trees delivering provisioning services. These inequalities may reflect more than socioeconomic stratification. The most dense and diverse street tree comunities sampled were in Nairobi's affluent, former European core in the west, while the least street trees were found in poorer locations in the east historically designated for Africans. As such, the inequalities Nairobi exhibits in street tree distribution between socioeconomic strata can also be interpreted through the lens of legacy effects as inequalities between neighborhoods designated for different racial groups during the colonial period. This suggests there are enduring marks of colonial planning and racial segregation on access to public amenities, including street trees, in Nairobi. Moreover, our findings suggest that present-day environmental conditions and planting practices may perpetuate past injustices, exemplified by the larger proportion of small, young trees in higher-income neighborhoods. To address these environmental legacies while safeguarding the existing street tree canopy, a better understanding of the causes underlying canopy imbalances, including infrastructural and land tenure barriers, and resident perceptions, is warranted. In addition, a comprehensive spatial inventory of Nairobi's tree cover could support the formulation of objectives for ecological resilience and equitable access. In turn, these would guide investments in site rehabilitation, site-specific tree planting, maintenance to increase tree survival, and social programs to avoid the unintended consequences of tree provision.

CRediT authorship contribution statement

Alice Gerow: Writing – review & editing, Writing – original draft, Visualization, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization, Data curation. **Dexter Locke:** Writing – review & editing, Validation, Formal analysis, Data curation. **Vivian Kathambi:** Resources, Investigation. **Mark Ashton:** Supervision, Writing – review & editing. **Craig Brodersen:** Writing – review & editing, Supervision, Resources, Methodology.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data Availability

The Nairobi street tree data, collected during a field survey in 2023, is publicly available to download and was processed as described in the Methods section with reproducible code provided at https://doi.org/10.5281/zenodo.13893349.

Acknowledgements

Thank you to CIFOR-ICRAF at World Agroforestry, particularly Cathy Watson, for hosting the Urban Forestry Research Fellowship. Thank you to Derrick Orero at Greenhavens, Jack Campbell-Clause and James Kirika at Kounkuey Design Initiative, Dorice Bosibiri at Muungano wa Wanavijiji, and Wangui Kimari and Wangari at Mathare Social Justice Centre for supporting field data collection. Thank you to the Tropical Resources Institute and the Hixon Center for Urban Sustainability at the Yale School of the Environment for providing financial support. Thank you to Liza Comita, Luke Sanford, Charlie Shackleton, Juan Villa, Dave Nowak, Morgan Grove, and two peer reviewers for the shapefile contributions and constructive feedback. The findings and conclusions in this publication are those of the authors and should not be construed to represent any official USDA or U.S. Government determination or policy.

Appendix A. Species list for live, identified trees (n = 1966), in alphabetical order

Family	%	Genus	%	Species	%	Origin
1. Anacardiaceae	1.12 %	1. Mangifera	0.81 %	*1. Mangifera indica*	0.81 %	Non-native
		2. Schinus	0.31 %	*2. Schinus molle*	0.20 %	Non-native
				3. Schinus terebinthifolius	0.10 %	Non-native
2. Annonaceae	0.71 %	3. Annona	0.10 %	*4. Annona sp*	0.10 %	Non-native
		4. Polyalthia	0.61 %	*5. Polyalthia longifolia*	0.61 %	Non-native
3. Apocynaceae	1.58 %	5. Acokanthera	0.15 %	*6. Acokanthera oppositifolia*	0.15 %	Indigenous
		6. Cascabela	0.25 %	*7. Cascabela thevetia*	0.25 %	Non-native
		7. Plumeria	1.17 %	*8. Plumeria alba*	0.31 %	Non-native
				9. Plumeria rubra	0.66 %	Non-native
				10. Plumeria sp	0.20 %	Non-native
4. Araliaceae	1.22 %	8. Schefflera	1.22 %	*11. Schefflera actinophylla*	1.22 %	Non-native
5. Araucariaceae	1.27 %	9. Araucaria	1.27 %	*12. Araucaria columnaris*	0.36 %	Non-native
				13. Araucaria heterophylla	0.92 %	Non-native
6. Arecaceae	10.58 %	10. Archontophoenix	2.95 %	*14. Archontophoenix alexandrae*	2.95 %	Non-native
		11. Chrysalidocarpus	4.63 %	*15. Chrysalidocarpus lutescens*	4.63 %	Non-native
		12. Hyophorbe	0.31 %	*16. Hyophorbe verschaffeltii*	0.31 %	Non-native
		13. Phoenix	0.15 %	*17. Phoenix canariensis*	0.05 %	Non-native
				18. Phoenix reclinata	0.10 %	Indigenous
		14. Roystonea	1.27 %	*19. Roystonea regia*	1.27 %	Non-native
		15. Syagrus	0.76 %	*20. Syagrus romanzoffiana*	0.76 %	Non-native
		16. Washingtonia	0.51 %	*21. Washingtonia filifera*	0.51 %	Non-native
7. Asparagaceae	6.87 %	17. Dracaena	4.48 %	*22. Dracaena afromontana*	0.36 %	Indigenous
				23. Dracaena ellenbeckiana	0.76 %	Indigenous
				24. Dracaena fragrans	0.05 %	Non-native
				25. Dracaena reflexa	0.31 %	Non-native
				26. Dracaena steudneri	3.00 %	Indigenous
		18. Yucca	2.39 %	*27. Yucca gigantea*	2.39 %	Non-native
8. Asteraceae	0.05 %	19. Brachylaena	0.05 %	*28. Brachylaena huillensis*	0.05 %	Indigenous
9. Bignoniaceae	12.05 %	20. Jacaranda	6.97 %	*29. Jacaranda mimosifolia*	6.97 %	Non-native
		21. Markhamia	2.75 %	*30. Markhamia lutea*	2.75 %	Indigenous
		22. Spathodea	2.14 %	*31. Spathodea campanulata*	2.14 %	Indigenous
		23. Tecoma	0.20 %	*32. Tecoma stans*	0.20 %	Non-native
10. Bombacaceae	0.25 %	24. Ceiba	0.25 %	*33. Ceiba speciosa*	0.25 %	Non-native
11. Boraginaceae	2.39 %	25. Cordia	0.25 %	*34. Cordia africana*	0.20 %	Indigenous
				35. Cordia monoica	0.05 %	Indigenous
		26. Ehretia	2.14 %	*36. Ehretia cymosa*	2.14 %	Indigenous
12. Cactaceae	0.10 %	27. Cereus	0.10 %	*37. Cereus repandus*	0.10 %	Non-native
13. Canellaceae	0.25 %	28. Warburgia	0.25 %	*38. Warburgia ugandensis*	0.25 %	Indigenous
14. Caricaceae	0.61 %	29. Carica	0.61 %	*39. Carica papaya*	0.61 %	Non-native
15. Casuarinaceae	0.66 %	30. Casuarina	0.66 %	*40. Casuarina cunninghamiana*	0.10 %	Non-native
				41. Casuarina equisetifolia	0.56 %	Non-native
16. Celastraceae	0.05 %	31. Elaeodendron	0.05 %	*42. Elaeodendron buchananii*	0.05 %	Indigenous
17. Combretaceae	0.81 %	32. Combretum	0.05 %	*43. Combretum molle*	0.05 %	Indigenous
		33. Terminalia	0.76 %	*44. Terminalia mantaly*	0.76 %	Non-native
18. Convolvulaceae	0.46 %	34. Ipomea	0.46 %	*45. Ipomea arborescens*	0.46 %	Non-native
19. Cupressaceae	5.65 %	35. Cupressus	4.83 %	*46. Cupressus arizonica*	2.90 %	Non-native
				47. Cupressus lusitanica	1.93 %	Non-native
		36. Juniperus	0.25 %	*48. Juniperus procera*	0.10 %	Indigenous
				49. Juniperus sp	0.15 %	Non-native
		37. Thuja	0.56 %	*50. Thuja orientalis*	0.56 %	Non-native
20. Cycadaceae	0.10 %	38. Cycas	0.10 %	*51. Cycas thouarsii*	0.10 %	Indigenous
21. Euphorbiaceae	7.88 %	39. Aleurites	0.51 %	*52. Aleurites moluccanus*	0.51 %	Non-native
		40. Bridelia	0.05 %	*53. Bridelia micrantha*	0.05 %	Non-native
		41. Codiaeum	0.71 %	*54. Codiaeum variegatum*	0.71 %	Non-native
		42. Croton	4.88 %	*55. Croton macrostychus*	0.31 %	Indigenous
				56. Croton megalocarpus	4.58 %	Indigenous
		43. Erythrococca	0.05 %	*57. Erythrococca bongensis*	0.05 %	Indigenous
		44. Euphorbia	0.61 %	*58. Euphorbia candelabrum*	0.41 %	Indigenous
				59. Euphorbia cotinifolia	0.15 %	Non-native
				60. Euphorbia tirucalli	0.05 %	Indigenous
		45. Ricinus	0.92 %	*61. Ricinus communis*	0.92 %	Indigenous
		46. Shirakiopsis	0.05 %	*62. Shirakiopsis elliptica*	0.05 %	Indigenous
		47. Synadenium	0.10 %	*63. Synadenium compactum*	0.10 %	Indigenous
22. Fabaceae	7.02 %	48. Acacia	1.68 %	*64. Acacia mearnsii*	0.05 %	Non-native
				65. Acacia sp	0.10 %	Indigenous
				66. Acacia xanthophloea	1.53 %	Indigenous
		49. Acrocarpus	1.78 %	*67. Acrocarpus fraxinifolius*	1.78 %	Non-native
		50. Bauhinia	0.31 %	*68. Bauhinia thonningii*	0.05 %	Indigenous
				69. Bauhinia tomentosa	0.10 %	Indigenous
				70. Bauhinia variegata	0.15 %	Non-native
		51. Calliandra	0.51 %	*71. Calliandra haematocephala*	0.05 %	Non-native
				72. Calliandra surinamensis	0.46 %	Non-native
		52. Erythrina	0.05 %	*73. Erythrina abyssinica*	0.05 %	Indigenous
		53. Leucaena	0.97 %	*74. Leucaena leucocephala*	0.97 %	Non-native

(*continued on next page*)

(*continued*)

Family	%	Genus	%	Species	%	Origin
		54. Senna	1.22 %	*75. Senna siamea*	0.20 %	Non-native
				76. Senna spectabilis	1.02 %	Non-native
		55. Tamarindus	0.15 %	*77. Tamarindus indica*	0.15 %	Non-native
		56. Tipuana	0.31 %	*78. Tipuana tipu*	0.31 %	Non-native
		57. Vachellia	0.05 %	*79. Vachellia farnesiana*	0.05 %	Non-native
23. Hamamelidaceae	0.05 %	58. Trichocladus	0.05 %	*80. Trichocladus ellipticus*	0.05 %	Indigenous
24. Lauraceae	3.97 %	59. Persea	3.97 %	*81. Persea americana*	3.97 %	Non-native
25. Loganiaceae	0.05 %	60. Anthocleista	0.05 %	*82. Anthocleista grandiflora*	0.05 %	Indigenous
26. Magnoliaceae	0.10 %	61. Magnolia	0.10 %	*83. Magnolia champaca*	0.10 %	Non-native
27. Malvaceae	0.31 %	62. Brachychiton	0.25 %	*84. Brachychiton acerifolius*	0.25 %	Non-native
		63. Dombeya	0.05 %	*85. Dombeya burgessiae*	0.05 %	Indigenous
28. Meliaceae	0.46 %	64. Ekebergia	0.05 %	*86. Ekebergia capensis*	0.05 %	Indigenous
		65. Melia	0.36 %	*87. Melia azedarach*	0.36 %	Non-native
		66. Toona	0.05 %	*88. Toona ciliata*	0.05 %	Non-native
29. Moraceae	4.93 %	67. Ficus	4.22 %	*89. Ficus altissima*	0.05 %	Non-native
				90. Ficus benjamina	2.90 %	Non-native
				91. Ficus elastica	0.36 %	Non-native
				92. Ficus natalensis	0.15 %	Non-native
				93. Ficus sur	0.15 %	Indigenous
				94. Ficus sycomorus	0.25 %	Indigenous
				95. Ficus thonningii	0.36 %	Indigenous
		68. Morus	0.71 %	*96. Morus alba*	0.71 %	Non-native
30. Musaceae	1.88 %	69. Musa	1.48 %	*97. Musa sp*	1.48 %	Indigenous
		70. Strelitzia	0.41 %	*98. Strelitzia nicolai*	0.41 %	Non-native
31. Myrtaceae	5.14 %	71. Callistemon	2.14 %	*99. Callistemon citrinus*	0.10 %	Non-native
				100. Callistemon rigidus	0.20 %	Non-native
				101. Callistemon salignus	1.42 %	Non-native
				102. Callistemon viminalis	0.41 %	Non-native
		72. Eucalyptus	1.73 %	*103. Eucalyptus sp*	1.73 %	Non-native
		73. Lophostemon	0.05 %	*104. Lophostemon confertus*	0.05 %	Non-native
		74. Psidium	0.81 %	*105. Psidium guajava*	0.81 %	Non-native
		75. Syzygium	0.41 %	*106. Syzygium cuminii*	0.36 %	Non-native
				107. Syzygium sp	0.05 %	Non-native
32. Oleaceae	1.88 %	76. Chionanthus	0.05 %	*108. Chionanthus battiscombei*	0.05 %	Indigenous
		77. Fraxinus	0.31 %	*109. Fraxinus pennsylvanica*	0.31 %	Non-native
		78. Ligustrum	0.41 %	*110. Ligustrum japonicum*	0.36 %	Non-native
				111. Ligustrum ovalifolium	0.05 %	Non-native
		79. Olea	1.02 %	*112. Olea europaea ssp cuspidata*	0.97 %	Indigenous
				113. Olea welwitschii	0.05 %	Indigenous
		80. Schrebera	0.10 %	*114. Schrebera alata*	0.10 %	Indigenous
33. Papaveraceae	0.15 %	81. Bocconia	0.15 %	*115. Bocconia frutescens*	0.15 %	Non-native
34. Phyllanthaceae	0.15 %	82. Bischofia	0.15 %	*116. Bischofia javanica*	0.15 %	Non-native
35. Phytolaccaceae	0.05 %	83. Phytolacca	0.05 %	*117. Phytolacca dioica*	0.05 %	Non-native
36. Podocarpaceae	0.20 %	84. Podocarpus	0.20 %	*118. Podocarpus falcatus*	0.15 %	Non-native
				119. Podocarpus latifolius	0.05 %	Non-native
37. Proteaceae	3.81 %	85. Grevillea	3.81 %	*120. Grevillea robusta*	3.81 %	Non-native
38. Punicaceae	0.05 %	86. Punica	0.05 %	*121. Punica granatum*	0.05 %	Non-native
39. Rosaceae	3.46 %	87. Eriobotrya	3.10 %	*122. Eriobotrya japonica*	3.10 %	Non-native
		88. Prunus	0.36 %	*123. Prunus africana*	0.25 %	Indigenous
				124. Prunus serotina	0.05 %	Non-native
				125. Prunus sp	0.05 %	Non-native
40. Rutaceae	1.63 %	89. Calodendrum	0.41 %	*126. Calodendrum capense*	0.41 %	Indigenous
		90. Casimiroa	0.20 %	*127. Casimiroa edulis*	0.20 %	Non-native
		91. Citrus	0.20 %	*128. Citrus limon*	0.20 %	Non-native
		92. Clausena	0.10 %	*129. Clausena anisata*	0.10 %	Indigenous
		93. Vepris	0.71 %	*130. Vepris simplicifolia*	0.10 %	Indigenous
				131. Vepris sp	0.56 %	Indigenous
				132. Vepris trichocarpa	0.05 %	Indigenous
41. Salicaceae	0.05 %	94. Dovyalis	0.05 %	*133. Dovyalis caffra*	0.05 %	Non-native
42. Sapindaceae	4.83 %	95. Filicium	4.83 %	*134. Filicium decipiens*	4.83 %	Indigenous
43. Solanaceae	3.31 %	96. Brugmansia	1.12 %	*135. Brugmansia suaveolens*	1.12 %	Non-native
		97. Cyphomandra	0.15 %	*136. Cyphomandra betacea*	0.15 %	Non-native
		98. Solanum	2.03 %	*137. Solanum erianthum*	0.05 %	Non-native
				138. Solanum mauritianum	1.93 %	Non-native
				139. Solanum torvum	0.05 %	Non-native
44. Strelitziaceae	0.86 %	99. Ravenala	0.86 %	*140. Ravenala madagascariensis*	0.86 %	Non-native
45. Ulmaceae	0.10 %	100. Trema	0.10 %	*141. Trema orientalis*	0.10 %	Indigenous
46. Verbenaceae	0.86 %	101. Vitex	0.86 %	*142. Vitex keniensis*	0.86 %	Indigenous

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