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ARTICLE

Socio-Ecological Systems

Just street trees? Street trees increase local biodiversity and biomass in higher income, denser neighborhoods

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Abstract

Tree canopy cover is a critical component of the urban environment that supports ecosystem services at multiple spatial and temporal scales. Increasing tree canopy across a matrix of public and private land is challenging. As such, municipalities often plant trees along streets in public rights-of-way where there are fewer barriers to establishment, and composition and biomass of street trees are inextricably linked to human decisions, management, and care. In this study, we investigated the contributions of street trees to the broader urban forest, inclusive of tree canopy distributed across both public and private parcels in Baltimore, MD, USA. We assess how species composition, biodiversity, and biomass of street trees specifically augment the urban forest at local and city-wide scales. Furthermore, we evaluate how street tree contributions to the urban forest vary with social and demographic characteristics of local residential communities. Our analyses demonstrate that street trees significantly enhanced citywide metrics of the urban forests' richness and tree biomass, adding an average six unique species per site. However, street tree contributions did not ameliorate low tree canopy locations, and more street tree biomass was generally aligned with higher urban forest cover. Furthermore, species richness, abundance, and biomass added by street trees were all positively related to local household income and population density. Our results corroborate previous findings that wealthier urban neighborhoods often have greater tree abundance and canopy cover and, additionally, suggest that investment in municipally managed street trees may be reinforcing inequities in distribution and function of the urban forest. This suggests a need for greater attention to where and why street tree plantings occur, what species are selected, and how planted tree survival is maintained by and for residents in different neighborhoods.

KEYWORDS
Baltimore, Baltimore Ecosystem Study, community composition, social–ecological systems, urban forest

INTRODUCTION

Urbanization is a key component of global change that dramatically influences biological and environmental patterns, affecting human well-being at all spatial scales (Haase et al., 2018). Reduced vegetation cover is a hallmark of urbanization, and urban areas are on the cusp of having greater impervious surface than tree cover worldwide (Nowak & Greenfield, 2020). This trend toward increasing imperviousness has dramatic consequences for the environmental conditions experienced by urban residents locally and at regional-to-global scales. Tree canopy is related to better air quality and correspondingly lower cardiovascular and respiratory disease (Mitchell, 2007; Namin et al., 2020; Nowak et al., 2014). Neighborhoods with more tree canopy cover may have lower rates of violent crime (Donovan & Prestemon, 2010; Gilstad-Hayden et al., 2015; Troy et al., 2012; Wolfe & Mennis, 2012). Furthermore, higher tree cover has been associated with positive mental health outcomes (Willis & Petrokofsky, 2017) and improved resident perceptions of the environment (Suchy et al., 2023). From a biophysical perspective, tree shade can effectively cool low-albedo cityscapes (Alonzo et al., 2021), which reduces experienced temperatures (Ziter et al., 2019) and heat-related mortality (Hoffman et al., 2020; McDonald et al., 2019; Sinha et al., 2021). Trees and vegetation also serve as primary habitats for urban wildlife (Le Roux et al., 2018) and intercept rainwater to ameliorate flooding and water quality issues (Phillips et al., 2019). As climate change drives rising summer temperatures in cities worldwide, the value of urban trees will only increase, and it is particularly important to understand nuances between different types of urban trees to best support and manage equitable urban environments.

There are numerous interacting ecological and social factors that influence distribution and composition of the urban forest, defined here as all trees on both public and private land, including individual parcels and forested patches (e.g., Baltimore Sustainability Plan, 2019; Nguyen et al., 2017; Young & McPherson, 2013). The establishment and survival of individual trees in urban landscapes occur across both public and private lands. Likewise, legacies of land ownership and both political and cultural decision-making persist in tree canopies, which can take many decades to mature (Locke et al., 2021; Roman et al., 2018). Urban foresters may have coherent plans for publicly owned land but have little power to influence private yards, and for this reason, municipalities frequently fund street tree planting focused on the publicly owned rights-of-way in order to initiate tree canopy establishment in residential and business districts. Divided and often conflicting responsibilities for planning, installation, care, and removal of trees mean that effective, sustainable, and equitable planting of street trees is a persistent coordination challenge (Lin et al., 2021; Schroeder et al., 2006). Planting street trees is a financial investment in terms of tree stock and labor, often assumed by the municipality, and that investment is threatened by urban biophysical stressors that contribute to high sapling mortality (Mullaney et al., 2015; Nowak et al., 2004). Newly planted trees require regular watering to support root establishment, which can take up to 1.5–9 years for a 1–6 in. diameter sapling (Zuzek, 2018). This care is often not budgeted for at the municipal level, requiring neighborhood groups and residents with different capacities and resources to assume this common responsibility (de Guzman et al., 2018; Wilkerson et al., 2018). Through time, these processes and challenges coalesce to create a spatially distributed street tree community in which the presence, identity, and life cycle of trees across a city reflect fine-scale decisions, resources, and action or inaction by individual people and institutions. Furthermore, unlike the processes that support community diversity in a naturally regenerating forest, recruitment of street trees is based on commercial species availability, cost, and resident choices (Roman et al., 2018; Roy et al., 2017). Likewise, individuals, nonprofit, civic, and municipal groups facilitate removal of urban canopies when trees are in poor condition, at risk, perceived to be a problem, or actively causing damage. Such decisions have long-lasting effects on tree canopy and can generate complex spatial patterns of canopy cover and composition across the urban landscape (Johnson et al., 2021).

The historical legacies and contemporary decisions that shape the composition and extent of urban tree canopy can also reinforce a broader array of environmental injustices evident in US cities (Boone et al., 2009; Grove et al., 2018; Lane et al., 2022; Schell et al., 2020). Redlining, the New Deal-era policy of disinvestment in communities with large minority populations (Zenou & Boccard, 2000), is one example of a historical policy with contemporary effects on the individual, social, economic, and environmental health of urban communities, including the urban forest (Grove et al., 2015; Locke et al., 2021; Namin et al., 2020; Roman et al., 2018). One legacy of redlining policies is that many US neighborhoods with lower income and/or majority black or African American populations receive disproportionately fewer ecosystem services associated with tree canopy cover relative to higher income or predominantly white neighborhoods (Gerrish & Watkins, 2018; Watkins & Gerrish, 2018). While less frequently examined, differences in the species composition and richness or diversity of tree species also contribute to variation in ecosystem services provisioning. Higher tree diversity at local scales lowers the exposure of residents to highly allergenic pollen (Sousa-Silva et al., 2021) and supports resilience to pest outbreaks at local and citywide scales (Mogenroth et al., 2016). Both the reduction of allergenic pollen and the increase in pest resilience have important public health implications. Allergy-induced asthma is a primary cause of hospitalization and missed school and work, and tree loss due to pests is related to increases in violent crime (Kondo et al., 2017) and incidences of cardiac and respiratory diseases (Donovan et al., 2013).

Many global cities, including Baltimore, MD, actively invest in the urban forest. Although biodiversity is valued by many urban foresters and agencies (Konijnendijk, 2021; Subburayalu & Sydnor, 2012), it is infrequently evaluated as a desired metric of the urban forest. In this study, we leverage long-term data to examine how street trees add to the current biomass and species composition of Baltimore's urban forest, both at the scale of city and at finer spatial scales associated with different residential communities and neighborhoods. Our overarching aim is to characterize how street trees augment the broader urban forest and with what added function across the diverse socioeconomic landscape of Baltimore, which we believe is representative of many other postindustrial cities. To do this, we focus on three objectives. First, we evaluate the distribution of street tree species richness and tree size to examine how street trees augment diversity and biomass metrics of the urban forest at the city scale. Second, we use a paired design to evaluate how street trees differ from any tree cover present on adjacent, randomly distributed plots. Lastly, we evaluate how street trees and their corresponding benefits are distributed among the social, economic, and race-based mosaic of Baltimore City.

METHODS

Study area

Baltimore, MD, USA, is a postindustrial city on the Atlantic seaboard and is home to approximately 575,000 people, with 62.3% of residents identifying as black and 30.46% identifying as white (worldpopulationreview.com; Figure 1). Baltimore has a temperate climate with hot, humid summers (average 26.0°C with 10 cm of rainfall monthly) and mild winters (average 0.8°C). Since 1998, Baltimore City and surrounding counties have been the focus of the Baltimore Ecosystem Study (BES, baltimoreecosystemstudy.org). From the beginning, BES has sought to integrate a social–ecological understanding of patterns and processes across the metropolitan region (Pickett et al., 2008, 2020), and this focus has specifically included advances in understanding the urban forest (Anderson et al., 2021; Grove et al., 2006; Sonti, 2020; Templeton et al., 2019; Zhou et al., 2011). Baltimore's urban forest looks and functions differently in different parts of the city, ranging from sparse vegetation in multifamily attached housing neighborhoods to closed, large crown tree canopies in historic townhome neighborhoods (Grove et al., 2006).

Data and analysis

Data from a 2017 inventory of all street trees, park trees in actively managed and mown areas, and vacant tree pits has been published by the City of Baltimore, Tree Baltimore, and Baltimore City Recreation and Parks (https://baltimore.maps.arcgis.com/apps/webappviewer/index.html?id=d2cfbbe9a24b4d988de127852e6c26c8). From this database, we extracted species identifications and diameter at breast height (dbh) measurements from 93,185 live trees planted along streets in Baltimore City (accessed November 2019; Anderson et al., 2019).

We assessed these street trees in combination with trees from 202 randomly located, 11.3-m-radius plots, which were sampled in 2014 using the i-Tree protocol to capture a representative sample of the urban forest in Baltimore City (Nowak, 2020; Figure 1a). This dataset contains relevant species identification, size, location, and condition data.

For analyses comparing street trees with composition and abundance of other trees in the urban forest at both city and local scales, we constructed 100-m-radius buffers from the center of each of the random plots. We used a 100-m radius to encompass the street trees on approximately two adjacent city blocks, as Baltimore City blocks are approximately 100 m long on average. We excluded the area covered by the random plot ($\pi \times (11.3^2$ m)) and identified the subset of live street trees in this buffer ring (Figure 1b). Some trees in the random plot dataset were also street trees, but they represented a small fraction of the total tree data (2.1% of total trees at 6.9% plots). Because our street tree samples already excluded any trees within the randomized plots, we retained these few within-plot street trees as part of the randomized plot dataset.

Objective 1: City-scale urban forest analysis

To assess how the ecological community and biomass provided by street trees augment the urban forest *writ large* (i.e., all trees across the urban landscape), we evaluated specific contributions of street trees as compared to the 202 randomized plots from the i-Tree dataset. To assess the degree to which street trees complement the diversity and biomass distribution across Baltimore's urban forest, we assessed species richness, tree abundance,

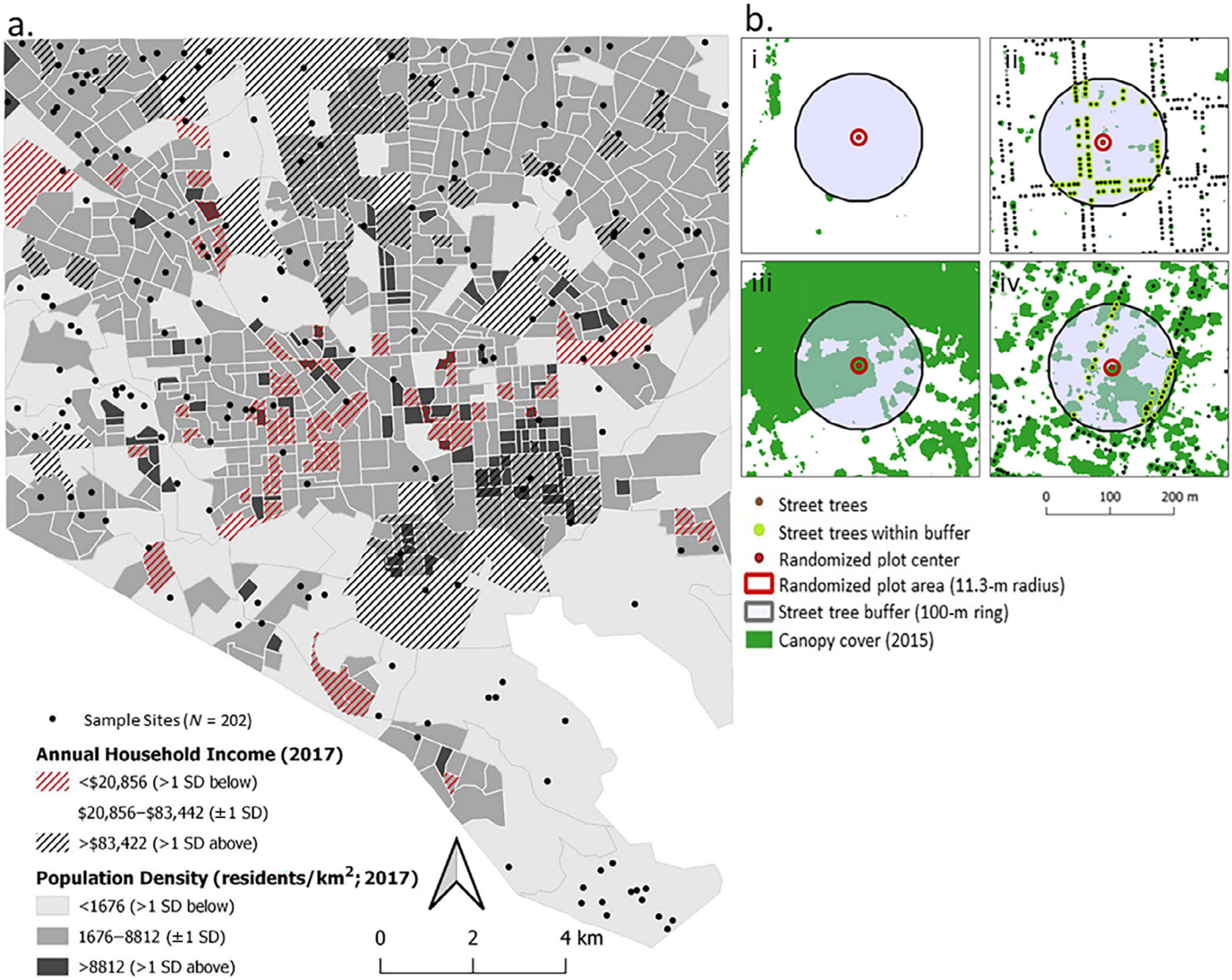

FIGURE 1 (a) Map of 202 randomly distributed sample sites across Baltimore, MD, USA. Block groups (census 2017) are shaded in gray based on population density (low, medium, and high) and hashed based on annual household income (low, red hashing; high, black hashing). (b) Examples of canopy cover (background green polygons) from trees in randomized plots (red circles) and street trees (black points). Across the 202 sites, we documented 41 sites without trees (i), 48 sites with no trees in the randomized plots but with street trees in the ring buffers (ii), 24 sites with random trees in the randomized plots but no street trees in the ring buffers (iii), and 89 sites with both street trees and random trees (iv).

and size of all randomized plots and across all street tree buffers (Figure 1b). Hereafter, we refer to the combination of randomized plots and the surrounding buffers where street trees were measured as sampled "sites" and differentiate between random plots and street tree buffers when appropriate.

To generate a city-scale view, we visualized data across Baltimore with scaled point maps using the QGIS graduated symbology with five classes per tree type. We used standard scales based on the natural breaks in the street tree dataset to compare the trees in random plots and the trees in street tree buffers. We visualize sample sites based on comparisons of tree species richness, tree abundance (irrespective of area differences), mean dbh (1.37 m above the ground for all trees >2.54 cm) of all trees, and basal area of trees per hectare as a standardized metric between different plot areas. Comparing the differences in street tree versus random tree contributions across the city allows us to visually assess the relative landscape-scale contributions of street trees to the broader urban forest and estimate the variation that exists across the urban landscape. In short, these maps highlight where street trees add diversity, abundance, and biomass across Baltimore.

To deepen our understanding of city-scale tree diversity from these two datasets, we consolidated community matrices of the street tree species within the buffer ring and for the tree species sampled within the randomized plots. From these community matrices, we calculated the rank abundance of species of street trees and randomized plot trees across the city using the "rankabundance" function in the "BiodiversityR" package (Kindt, 2021; Kindt & Coe, 2005; RStudio Team, 2020). We report the abundance of each species, the frequency of sites at which each species was found, and the percentage of the total number of trees for the top 20 species of each tree community—defined here as the composition of species at a site or within a plot or buffer—which collectively comprised at least 70% of sampled trees in both randomized plots and street trees.

Objective 2: Local-scale paired analysis of street trees and randomized plot trees

To address our second objective, we examine the differences in biodiversity and biomass metrics between randomized plots and adjacent street tree buffers (as described above). By pairing these analyses, we directly compare street tree physical characteristics and species compositions with those of other trees in direct proximity (Figure 1b).

Using the paired dataset, we calculated the total number of species in each of the randomized plots and in the surrounding buffers. These species counts are independent of abundance and are hereafter referred to as species richness. From these species lists, we further identified any species that were found in both the plot and the buffer at a given site. In sites that had trees in both the randomized plot and the street tree buffer, we conducted paired-sample t tests to assess differences in street tree versus randomized plot tree species richness, mean tree dbh, and basal area per hectare. We do not compare tree abundance in this pairwise fashion due to the incongruent areas of the 11.3-m-radius randomized plot and the 100-m-long street tree buffers.

Objective 3: Sociodemographic and landscape patterns

To understand how the species composition, abundance, and size of street trees vary across the social–ecological landscape of Baltimore, we analyzed spatial patterns in conjunction with data from the American Community Survey (ACS) at the block group level (U.S. Census Bureau, 2017; Figure 1a). The ACS estimates demographic parameters between decennial censuses. We assigned block group identity based on which block group contained the center of the randomized tree plot. For each block group, we extracted median household income, proportion of residents identifying as white, and population count. We selected these variables because money, power, and population density are all known correlates of tree canopy cover and reflect the complex roles of humans in urban systems (Troy et al., 2007). We used QGIS 3.10.13 to calculate population density (in per square kilometers) for each block group and further calculated the proportion of built, paved, and other impervious surfaces from the 2013 Chesapeake Bay High-Resolution Land Cover Project (Pallai & Wesson, 2017; Robinson et al., 2019) by summing the proportions of built, paved, and other impervious surfaces using the Tabulate Area tool and dividing by the total area.

We used an corrected Akaike information criterion (AIC_c) model averaging approach to assess how street tree investment varied across the city (Burnham & Anderson, 2002). We fit a linear regression model for each of the following responses at the site level ($N = 202$): street tree abundance, street tree richness, species added by street trees, and street tree basal area. Each model included the same sociodemographic predictor variables (median income + proportion built and impervious + proportion white residents + population density) and the corresponding predictor variable for trees in the randomized plot (abundance, richness, number of unique species, and basal area). Model assumptions were tested by plotting residuals versus fitted values and running nonconstant error variance tests with function "nvcTest" in R package "car" (Fox & Weisberg, 2019). All models had p values >0.05, indicating constant variance in residual errors. Although the included variables do not comprehensively represent the neighborhoods or residents of Baltimore, we based our a priori selection of sociodemographic predictors on factors that are known to contribute to politicized power dynamics in American cities, including race (focusing on white people as the dominant social group; McDermott & Samson, 2005), income (Koontz, 2010), and the number of residents in a given area (Lin et al., 2021). These variables have furthermore been investigated in recent work evaluating patterns of street tree equity and environmental justice (Landry & Chakraborty, 2009; Lin et al., 2021). We calculated variance inflation factors (VIFs) for all variables to identify potential effects of multicollinearity on our model outputs using the "vif" function in the "car" library (Fox & Weisberg, 2019) and interpret VIF values between 1 and 5 as indicative of low correlation between variables. All statistical analyses were done using RStudio 1.4.1717 (RStudio Team, 2020).

We extracted and analyzed all possible models from the full contingent of variables based on AIC_c weight using the "dredge" function in the MuMIn package (version 1.43.17; Barton, 2020). From this list of ranked candidate models (see Appendix S1: Table S2), we calculated the coefficients and 95% confidence intervals of a final model comprised of the model-averaged set (all models with AIC_c weight adding to 0.95). This method, like other information-theoretic approaches, assesses the relative importance of all variables established as a priori hypotheses, thus equipping us to describe regions of the city where street trees are larger, more abundant, diverse, and/or distinct from randomized plot trees.

RESULTS

Objective 1: What do street trees add to the urban forest at the city scale?

A majority (68%, 137 sites) of the 202 sites had street trees present, and street trees were the only trees present at 25% of all sites (48 sites). There were 41 sites without any trees, and these sites were most frequent in the southern part of the city (Appendix S1: Figure S1). The abundance of trees per site ranged from 0 to 79 (mean 17.45 ± 18.50), but abundance was generally lower in the 32% of sites that had no street trees (65 sites, 6.99 ± 15.41) and higher in sites where street trees were present (22.43 ± 17.79). While abundance comparisons could be due to the greater area of the street tree buffer rings, mean dbh was also greater (36.83 ± 16.22 cm) at sites with street trees than in sites without (30.3 ± 25.5 cm). At sites where street trees were the only constituents of the urban forest, abundance was comparable to sites with trees in both types of plots (48 sites, 17.06 ± 19.55 stems), and mean dbh was similar (34.20 ± 15.60 cm). Across all sites, inclusion of street trees increased the species richness over richness in the randomized plots by an average six species (6.30 ± 4.77 species added).

Across Baltimore, street trees were typically larger, less dense, and more species-rich than the trees in random plots, but these patterns differed across the landscape. Areas of high street tree basal area per hectare (Figure 2d) were dispersed, and there were areas of low street tree basal area along the south, west, and northeast fringes. Street tree abundance was expectedly higher than that of smaller random plots, and this was particularly prominent in the center of the city, where the central business district is located (Figure 2b). These same plots in the city center had very low mean dbh (Figure 2c), likely indicating areas of new plantings. Interestingly, these areas near the central business district also had fairly high species richness (Figure 2a). Areas, where street tree dbh was high, were primarily in the northern part of the city (Figure 2c). In these areas, street tree dbh was generally higher than dbh in randomized plot trees. Species richness exhibited a distinct pattern (Figure 2a), with moderately high street tree richness in the central business district and the highest richness in the northwest quadrant of the city. Again, this was different from tree species richness in the randomized plots, which was highest in and around a large, forested park on the west side.

We documented a total of 154 species of trees across Baltimore. Of these, 53 were found only in the street tree communities, and 36 were found only in the randomized plots. The remaining 65 species were found in both the randomized plots and the street tree buffers, but the relative abundance and total composition of the communities differed between these subsets of the urban forest (Table 1). Maple species (genus *Acer*) were by far the most common street trees, accounting for 26% of street trees across all sites. By comparison, only 6.8% of trees in the random plots were maples. Of the 20 most frequently recorded species in the street tree buffers and the randomized plots, only 6 species were observed in both types of subplots across the landscape (Table 1).

Objective 2: What do street trees add to the urban forest at the local scale?

Street trees have different patterns of biomass than those of nonstreet trees, and these patterns are detectable through paired assessments between random plots and street tree buffer sites. Individual street trees were significantly larger (by a median of 14 cm) than trees growing in adjacent plots (paired sample t test, $df = 201$, $t = -5.11$, $p < 0.0001$, median ± SD: 22.88 ± 21.81 cm vs. 28.67 ± 20.16 cm for nonstreet trees; Figure 3c). However, the basal area in square meters per hectare contributed by street trees was significantly lower (median ± SD: street trees 0.24 ± 0.72 m^2/ha, randomized plots 0.44 ± 13.18 m^2/ha, paired t test, $df = 201$, $t = 7.81$, $p < 0.0001$; Figure 3d). Overall, while street trees were on average larger than trees in paired randomized plots, street trees are distributed less densely across the site than random trees.

To assess the value added to urban forest diversity and biomass by street trees at a local scale, we compared tree size and community composition in each randomized plot with the paired street tree buffer. When street trees were present in the buffer, they always added at least one unique species to the composition of the randomized plot (Figure 3a). The addition of street trees accounts for 67% greater species richness at sites with both types of trees (9.17 ± 4.85 species) compared to sites that have only trees in random plots (5.46 ± 4.42 species; Figure 3a). From a

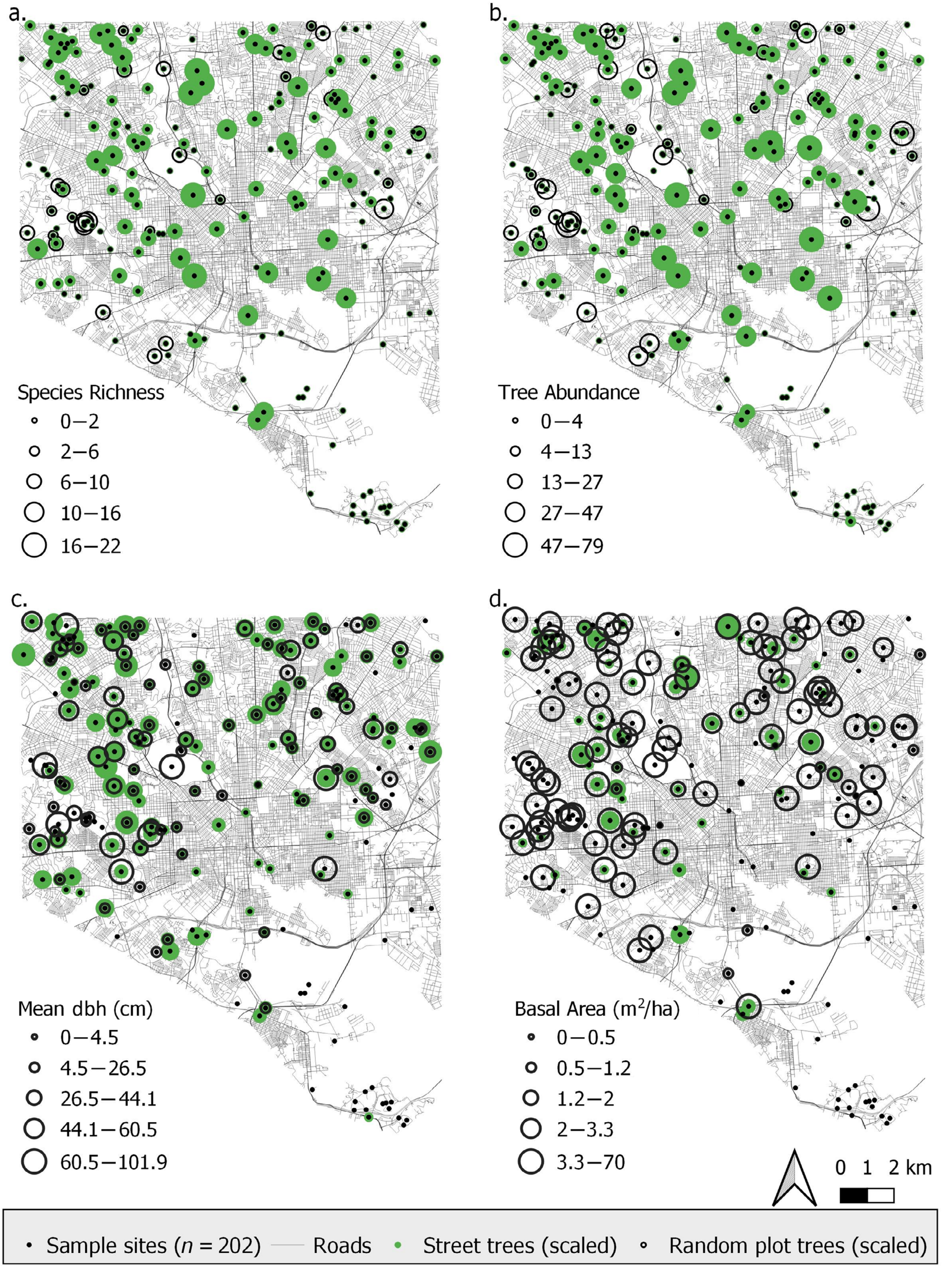

FIGURE 2 Legend on next page.

TABLE 1 The 20 most common species recorded in randomized plots ($N = 202$) and street trees within 100-m buffers.

	Randomized plots			Street tree buffers		
Rank	Species	Abundance (frequency)	Percentage of stems	Species	Abundance (frequency)	Percentage of stems
1	*Fagus grandifolia*	149 (14)	13.4	***Acer rubrum****	355 (71)	14.7
2	*Ailanthus altissima*	59 (18)	5.3	*Zelkova serrata*	211 (40)	8.7
3	***Ulmus americana*****	58 (17)	5.2	*Prunus* spp.	148 (36)	6.1
4	*Prunus serotina*	50 (18)	4.5	*Platanus × acerifolia*	134 (26)	5.5
5	***Quercus rubra******	49 (17)	4.4	*Acer platanoides*	110 (47)	4.5
6	*Morus alba*	46 (16)	4.2	*Pyrus calleryana*	109 (26)	4.5
7	*Robinia pseudoacacia*	44 (13)	4.0	*Tilia cordata*	109 (34)	4.5
8	***Acer rubrum****	39 (22)	3.5	***Acer saccharinum***†††	74 (26)	3.1
9	*Fraxinus pennsylvanica*	39 (14)	3.5	*Gleditsia triacanthos inermis*	66 (16)	2.7
10	***Fraxinus americana***†	38 (16)	3.4	***Quercus phellos***††	58 (9)	2.4
11	*Quercus alba*	35 (12)	3.2	*Acer saccharum*	56 (26)	2.3
12	*Ulmus pumila*	34 (1)	3.1	*Liquidambar styraciflua*	52 (20)	2.1
13	*Liriodendron tulipifera*	33 (13)	3.0	***Quercus rubra******	49 (23)	2.0
14	*Cornus florida*	24 (10)	2.2	***Fraxinus americana***†	46 (25)	1.9
15	***Quercus phellos***††	22 (6)	2.0	***Ulmus americana*****	40 (18)	1.7
16	*Acer negundo*	21 (11)	1.9	*Acer × freemanii*	34 (14)	1.4
17	*Carpinus caroliniana*	19 (5)	1.7	*Lagerstroemia indica*	33 (11)	1.4
18	*Nyssa sylvatica*	19 (6)	1.7	*Pinus strobus*	33 (8)	1.4
19	*Ulmus rubra*	19 (9)	1.7	*Tilia americana*	32 (10)	1.3
20	***Acer saccharinum***†††	16 (4)	1.4	*Ginkgo biloba*	29 (10)	1.2

Note: Species are ranked by collective abundance across all sites, with the frequency of sites where a species was recorded noted in the parentheses. Species in both top 20 lists appear in boldface and are noted with combinations of * or † to help identify overlaps across rank abundance lists.

biodiversity perspective, overall redundancy in species in randomized plots and street tree buffers was quite low—only 40 out of 202 sites had overlapping species. The maximum number of shared species in the paired plots within a site was just three species, and most sites had considerably less overlap (for 89 sites with both: 0.56 ± 0.71 [mean $\pm$ SD]; Figure 3a). Maple species (*Acer* spp.) were the most common links between the community of trees in the randomized plots and the adjacent street tree community. Fourteen sites had cross-plot redundancies in one of three *Acer* species: red maple (*A. rubrum*, 10 sites), Norway maple (*A. platanoides*, 5 sites), or silver maple (*A. saccharinum*, 3 sites). Other species that occasionally overlapped were white mulberry (*Morus alba*, 4 sites), green ash (*Fraxinus pennsylvanica*, 3 sites), and tulip tree (*Liriodendron tulipifera*, 2 sites).

Objective 3: Sociodemographic and landscape patterns in street tree metrics

The results of our model averaging analysis suggest that street tree abundance is positively associated with median household income, population density, and the proportion of white residents (Figure 4; Appendix S1: Table S2). Median household income was positively associated with

FIGURE 2 Scaled point maps showing differences in distribution of high tree species richness (a), tree abundance (b), mean dbh (c), and basal area (d) for the full urban forest sample at sampled sites across Baltimore. Scales are indicated by natural breaks in the street trees dataset for each panel and are comparable between random trees (black outlines) and street trees (green circles) within an individual map. Sites with only a small black point had no trees present.

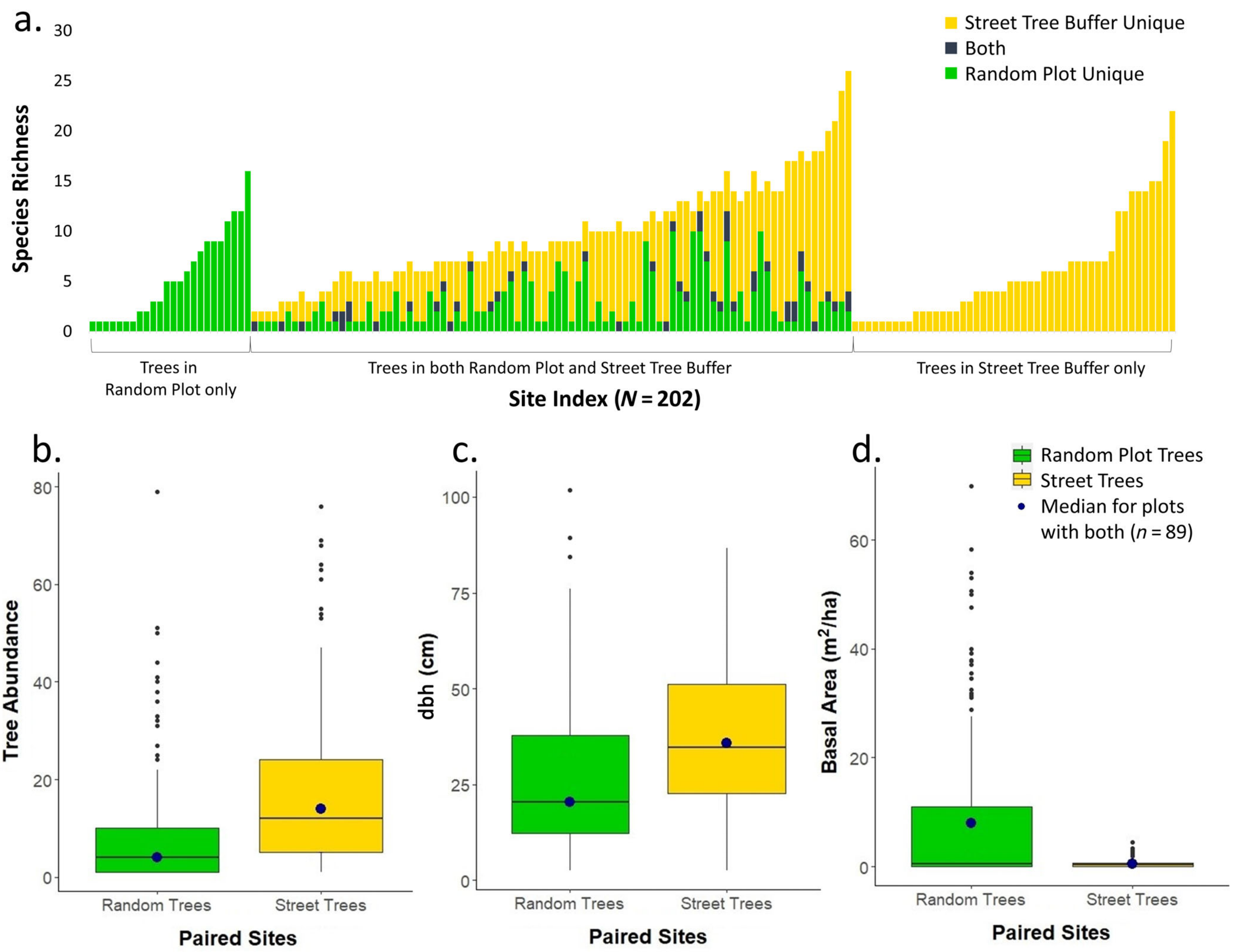

FIGURE 3 Comparison of (a) paired site-level species richness, (b) total tree abundance (11.3-m-radius randomized plots and street tree buffers), (c) site-level mean dbh, and (d) total basal area of trees in randomized plots (green) and in street tree buffers (yellow). All sites are included in paired graphs, regardless of the combination of types of trees present. Site indices (along the *x*-axis) in (a) are sorted first by the combination of types of trees present (random trees only, random and street trees, and street trees only), and within these groups by species richness. Species that were found in both are marked in navy blue. Boxplots delineate the median and interquartile ranges, and the medians for sites that contained both types of trees are further marked with navy blue circles.

each of the other street tree metrics as well, including how many unique species street trees add to urban forest at local scales (Figure 4). Even though median household income is moderately correlated with the proportion of white residents (Appendix S1: Table S1), this did not limit the model inference (VIFs of 1.64 and 1.61, respectively; Appendix S1: Table S2). Given that street trees are by definition associated with built streets, the finding that impervious surface was positively associated with street tree abundance is expected (Figure 4; Appendix S1: Table S2).

Tree abundance and richness in the randomized plots were not significant predictors of any street tree metrics, although there was a negative association between lower street tree abundance in areas with lower tree abundance in the paired randomized plots (Figure 4).

DISCUSSION

In this study, we identified a diverse community of street trees that is different in both structure and species composition from adjacent tree communities. Street trees across our study sites contributed 59 unique species to Baltimore's urban forest, increasing local, site-level tree richness by an average six species. Individual street trees were larger on average than trees in adjacent randomized plots but still comprised a relatively small proportion of the total basal area across the city. Still, street tree abundance, biomass, and species diversity were greater in parts of the city that also have high urban forest cover overall, reflecting greater street tree resources in the higher income neighborhoods that also have greater urban forest cover. Overall, our findings highlight the

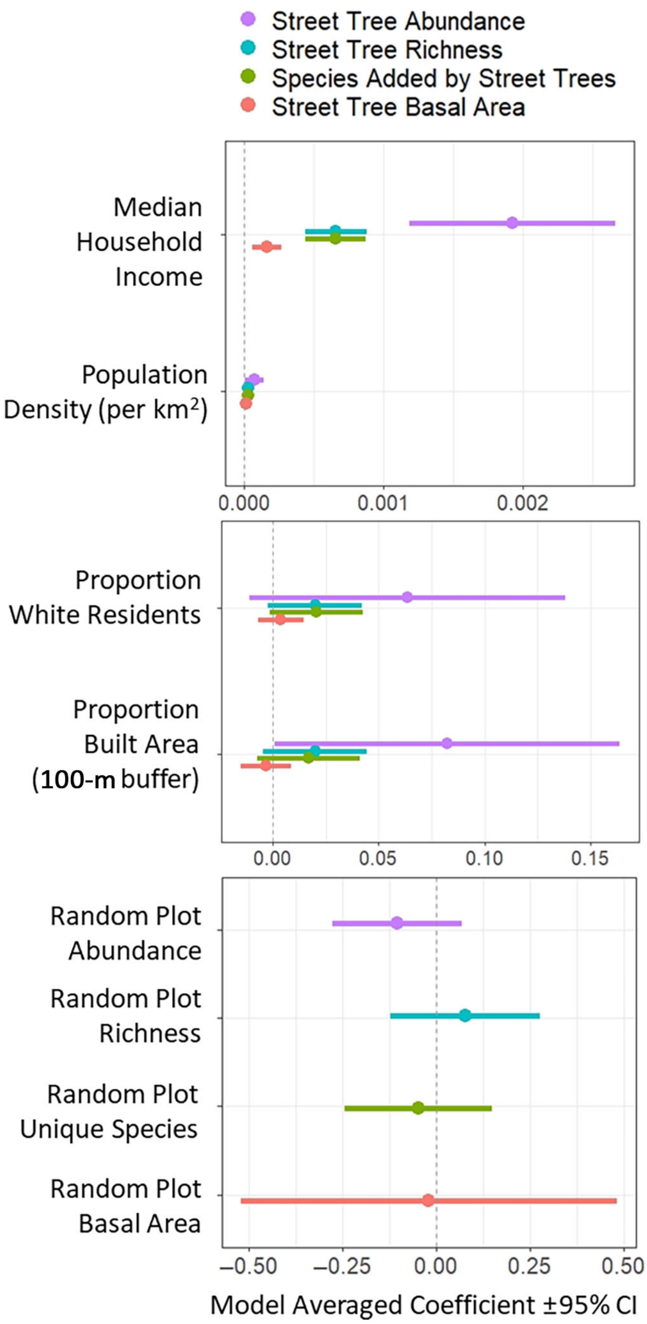

FIGURE 4 Model averaged coefficients (±95% CI) for all variables included in multiple regression analyses. Note the different scales of the *x*-axes; scale was adjusted for groups of predictors to effectively present results.

importance of street trees as a distinct and important subset of the urban forest that augments biomass a little and species richness substantially. Tree composition and size in urban ecosystems are the results of a combination of ecological patterns of recruitment and survival with social mechanisms of planting, care, and maintenance. Our results may reflect biased efforts in planting, inequitable capacities for maintenance, or some combination of the two. Likewise, our findings reflect a legacy of historical investment (Burghardt et al., 2022; Locke et al., 2021) as well as contemporary and ongoing efforts.

The ecosystem services provided by the urban forest are driven by the extent of the tree canopy, which is more a function of tree identity and structure than the number of trees at a site (Kim et al., 2015). Consequently, trees of different species, ages, sizes, and in different states of health contribute differently to the urban forest at both citywide and local scales (Kim et al., 2015; Lin et al., 2021).

The local capacity for street trees to provide ecosystem services such as microhabitat cooling and shading (Salmond et al., 2016) further depends on the urban forest context in which street trees are cultivated. Across Baltimore City, we found that street trees made meaningful contributions to tree abundance, occasionally matching the tree density at sites in forested parts of the city (Figure 2a). Across the landscape, street trees were typically larger than trees in randomized plots (Figure 2c). This pattern was even starker in sites that had both street trees and trees in randomized plots; at such sites, street trees were significantly larger—and likely older—than the trees in the paired randomized plots (Figure 3c). This could be informative of potential mechanisms of street tree investments spurring gentrification and leading to tree plantings on private properties. Further research is needed to investigate this plausible time lag in planting evident in local-scale dbh differences.

Our findings clearly demonstrate that street trees augment species richness at both local and city-wide scales. Given the dearth of species that are redundant in the street tree buffer and the randomized plot at a site, it is unlikely that species planted as street trees are recruited into the broader urban forest in any appreciable way, but it should be noted that our data does not include seedling or sapling-sized individuals. It is likely that specific social factors and human decisions translate into high tree species richness across subsets of the urban forest (Grove et al., 2014; Locke & Grove, 2016; Troy et al., 2007). Species in the genus *Acer* may be a valuable case study for investigating social–ecological patterns in diversity across the urban forest, as they were the most common species in both the randomized plots and the street tree buffers at a site. Maple species, particularly red maple (*A. rubrum*), are popular street trees because they are hardy over a wide range of conditions, quickly grow dense crowns, and have attractive fall foliage (Ma et al., 2020). For these reasons, they are also likely attractive to homeowners and private land managers. Examining maples as linkages may help clarify mechanisms of urban forest dynamics by assessing, for example, if street trees are a seed source on private land or if planting decisions are comparable between institutions and private landowners. Additionally, evaluating the interactions between all maple trees across the city could be particularly useful in understanding the vulnerabilities of various urban tree communities to the risks of climate change and pathogen invasion (Reynolds et al., 2020).

The three most common street tree species across the citywide analysis, red maple (*A. rubrum*), Japanese Zelkova (*Zelkova serrata*), and cherries (*Prunus* sp.), were distinct from the species most often found in randomized plots: American beech (*Fagus grandifolia*), tree of heaven (*Ailanthus altissima*), and American elm (*Ulmus americana*). Zelkova and tree of heaven are introduced species, while American beech, American elm, and red maple are native or horticultural cultivars of native species. These distinctions and combinations of native and non-native trees across the landscape may contribute to fine-scale changes in canopy-based ecosystem services overtime, as another study on the same permanent plots identified a link between non-native species and local-scale canopy losses in Baltimore (Anderson et al., 2021). Examining how these patterns extend or change in the street tree subset of the urban forest could provide valuable information about whether and how street tree plantings contribute to urban canopy cover through time. Likewise, further studies investigating seedling recruitment from street trees to private and/or public lands within the urban forest are necessary to fully understand the role street trees may play in sustainability and resilience metrics.

Although increased biodiversity is broadly linked to a variety of ecosystem services (Harrison et al., 2014), the distinct tree communities found in randomized plots and street tree buffers suggest there may also be different ecosystem disservices from different subsets of the urban forest. For example, both American beech and American elm (randomized plot dominants) have moderately high allergenicity hazard (scores 7–8 out of a possible 10; Nowak & Ogren, 2021), while red maple (males) and Zelkova are among the highest risks for allergenic pollen (scores 9 and 10, respectively; Nowak & Ogren, 2021). Across the city, street trees add red maple to 61 sites (30%) and add Zelkova to 40 sites (20%); 22 of these sites add both (11%). While adding individual tree species with high allergenicity risks to an urban landscape certainly poses challenges for public health (Pawankar, 2014; Velasco-Jiménez et al., 2020), little is known about the mechanisms of the interspecific interactions between pollen-producing trees in urban areas (Behrendt & Becker, 2001). More investigation is needed to determine whether changes in fine-scale tree biodiversity result in such challenges as interspecific pollen competition and reduced allergen sensitization or increases in the press of pollen loads throughout the season, which might increase sensitization (Dales et al., 2004).

Street trees are not evenly distributed across the landscape in Baltimore. Notably, there are many sites in south Baltimore that do not have any trees. These are primarily industrial sites, but the dearth of trees of any kind in this part of the city represents an opportunity for meaningful tree plantings. In the more residential parts of the city, we found that higher income and densely populated block groups had more and larger street trees, greater street tree diversity, and also a higher number of unique species compared to adjacent randomized plots. Neighborhoods with a higher human population density likely have higher street density, resulting in more spaces to plant street trees and potentially higher tax revenue (Smith, 1978). Furthermore, it is well-established that higher income neighborhoods in American cities have greater tree canopy cover (Gerrish & Watkins, 2018). Prior studies have also shown similar associations with street tree abundance (Baró et al., 2019; Lin et al., 2021). We demonstrate that the pattern extends further to the biodiversity of municipally managed street trees. It is unsurprising that higher income neighborhoods receive greater investments in street tree plantings, as municipal governments often act in accordance with social power hierarchies, whether explicitly or through implicit, unaddressed biases (Boone et al., 2014; Buckley et al., 2019; Lawrence et al., 2013; Lord & Norquist, 2010). Even as initiatives try to promote environmental equity and justice by greening historically excluded neighborhoods, this does not come at the expense of planting and maintenance of existing large urban trees in wealthier regions (Foster et al., 2022; Pincetl et al., 2013). Indeed, planting many small, new street trees in areas that are lacking canopy cover does not compensate for the ecosystem services provided by large, established canopies (Nowak, 2002), and may drive green gentrification that harms vulnerable residents well-intentioned programs are often trying to help (Donovan et al., 2021). Increasing small tree abundance and diversity through planting efforts without recognizing historic power imbalances that have led to a larger and more diverse community of street trees in higher income neighborhoods, often white neighborhoods, may reinforce the status quo in environmental form and ecosystem service provisioning (Locke et al., 2021). Instead, the pursuit of environmental justice in urban tree canopy in low-income areas may involve simultaneous investments in the preservation and management of existing street trees, creation of high-quality spaces to plant new trees, and capacity to plant a greater diversity of tree species than may be necessary in other areas of the city.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data underlying the work reported here were obtained as follows: iTree data: https://portal.edirepository.org/nis/mapbrowse?packageid=knb-lter-bes.1000.170. street tree data: https://portal.edirepository.org/nis/mapbrowse?packageid=knb-lter-bes.5020.1. Chesapeake Land Conser vancy data: https://lila.science/datasets/chesapeakelandcover. American Community Survey data: https://www.census.gov/newsroom/press-kits/2018/acs-5year.html.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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