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Plant communities differ along socioeconomic gradients in Baltimore, MD, and Washington, D.C.

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Plants on residential urban properties can provide valuable ecosystem services or produce harmful disservices depending on fine-scale characteristics tied to plant species identity, such as growth habit or native status. The composition of plant identities on a given block is often influenced by socioeconomic factors, leading to variable green space function across a city. We surveyed residential plants on Baltimore, MD, and Washington, D.C. blocks, documenting differences in structure, richness, and community composition along an income gradient and between abandoned and neighboring occupied properties. Both canopy and ground vegetation on low-income residential properties covered less area and were more likely to contain non-native species than their higher-income counterparts, with different tree, vine, and non-native communities present. Abandoned properties had more canopy cover and higher tree richness than occupied neighbors but similar community composition including five common vines, four of which were non-native. These differences have important implications for ecosystem services, and such fine-scale knowledge could better inform the management of green space to benefit urban residents.

KEYWORDS

abandonment, canopy, cover, ground, income, richness, urban, vegetation

Introduction

Urban plants provide many environmental and cultural services that make them valuable components of city ecosystems. Urban residents surrounded by green space may benefit from reduced air pollution (e.g., Yang et al., 2005; Nowak et al., 2006, 2018), improved stormwater management (e.g., Berland et al., 2017; Ponte et al., 2021; Selbig et al., 2022), cooler summer temperatures (e.g., Tan et al., 2016; Wang and Akbari, 2016; Gunawardena et al., 2017), lower crime levels (e.g., Donovan and Prestemon, 2012; Troy et al., 2012; Lin et al., 2021), and improved mental and physical health (e.g., Chawla, 2015; Twohig-Bennett and Jones, 2018; Song et al., 2022). While urban forests and parks are obvious sources of greenery in cities, vegetation on residential land has the potential to contribute even more to city-wide ecosystem services due to its extensive area. Residential property often represents half or more of a city's total zoned land, e.g., 75% in New York City, NY (NYCPlanning, 2023) and 52% in Los Angeles, CA (Menendian et al., 2022). The vegetation immediately surrounding a resident's home is also arguably the most influential green space in their daily life; these are the plants that can, for example, lower their home's heating and cooling bills (Ko, 2018) or create a green view outside their windows to boost self-esteem and happiness (Soga et al., 2021).

While environmental variables such as climate or soil type may have historically dictated where and how well plants can grow, vegetation in urban ecosystems is additionally affected by social conditions, economic factors, and cultural preferences. Among a multitude of socioeconomic variables, household income appears to be particularly influential, possibly because wealth dictates the amount of disposable income that residents can spend on landscaping and/or because higher-income residents can afford properties in neighborhoods with already-desirable vegetation (Leong et al., 2018). Either way, the few reported surveys of vegetation on residential properties in cities across the United States have consistently found differences in total plant abundance along income gradients, most often reporting higher total area of vegetation on higher-income city blocks (e.g., Jenerette et al., 2013; Spotswood et al., 2021; Heo and Bell, 2023).

Though this positive, linear relationship between plant cover and income is common nationwide, it is not ubiquitous. Studies in Philadelphia, PA (Pearsall and Christman, 2012); Baltimore, MD (Little et al., 2017; Biehler et al., 2025); and Detroit, MI (Endsley et al., 2018), all found quadratic relationships with similarly high levels of greenery in both low-income and high-income neighborhoods and less in medium-income neighborhoods. In each case, the authors noted the prevalence of vacant lots (properties without a principal building) and/or abandonment (properties with uninhabited buildings, often in a state of decay) in low-income neighborhoods as significant contributors to total vegetative cover. It has been suggested that while high-income blocks may attain high levels of greenery through consistent care, low-income blocks with high rates of unoccupied property may reach similar levels of greenery through neglect (Gulachenski et al., 2016; Riley and Gardiner, 2020; Biehler et al., 2025). Consequently, two blocks with equally abundant vegetation may have very different plant communities present, in terms of structure (i.e. open-canopy *vs*. closed-canopy, vegetated *vs*. bare ground), richness (i.e. monoculture *vs*. diversity), and composition (i.e. species identities).

Plant community structure, richness, and composition should be considered in studies describing urban vegetation, as equal abundance alone does not necessarily indicate equal desirability or utility (Biehler et al., 2025). Surveys of plant communities in Toledo, OH, revealed that abandoned properties, which had greater canopy cover than neighboring occupied properties, were more frequently documented as urban blight due to the overgrown nature of the plants present (Berland et al., 2020). Replacing overgrown vegetation on vacant lots with regularly mown grass and a small number of trees in Philadelphia, PA, was associated with lowered heart rates (South et al., 2015) and reduced feelings of depression (South et al., 2018) in nearby residents, as well as fewer gun assaults (Branas et al., 2011). Thus, it is essential to consider the characteristics of the plant community and not merely its cover area. A study in Bradford, United Kingdom found that residents' mental health benefit from spending time in urban green spaces increased with increasing plant diversity (Wood et al., 2018). Yet, as with vegetation cover, the desirability of high species diversity (e.g., richness) is not universal and may depend upon other fine-scale plant community characteristics such as composition. Surveys of resident attitudes toward plants in their yard or neighborhood demonstrate the importance of species identity, with specific traits like flower size or place of origin affecting resident opinion (Kendal et al., 2012). Thus, while increased diversity may be more beneficial in the abstract, it seems logical to assume that a yard rich with species/traits less-preferred by humans such as mulberry trees (which drop messy fruits), poison ivy (which can cause rashes), and saw greenbrier (which has thorns) may be less desirable to residents than a yard with monoculture turf.

Plant growth habit (e.g., herb, shrub, tree, vine) and native status are important traits that can mediate resident perceptions of local vegetation. A survey of people living near vacant lots in Detroit, MI, found that neighbors preferred mown turf or low-growing shrubs with prominent flowers over weedy vegetation or trees, as the former were perceived signs of care (Nassauer et al., 2021). Regarding the influence of native status, a global review of urban wildlife diversity found that 43% of studies reported that native plants support wildlife diversity better than non-native plants, while only 8% reported the opposite (Berthon et al., 2021).

Despite the apparent importance of community characteristics on the ecosystem services or disservices (negative impacts) provided by urban vegetation, few studies have evaluated how urban plant structure, richness, and/or composition are associated with socioeconomically diverse residential properties (Rothman et al., 2026). Structure was the most common characteristic documented, with results often showing more canopy (e.g., Clarke et al., 2013; Avolio et al., 2015, 2020) and ground (e.g., Clarke et al., 2013; Lewis et al., 2017; Wheeler et al., 2022) cover with increasing income. Studies on residential canopy richness across income gradients also showed a positive relationship (Avolio et al., 2018, 2020), although ground species richness focused on herbaceous and 'weedy' species, largely decreased with increasing income (Lowenstein and Minor, 2016; Wheeler et al., 2017; Blanchette et al., 2021). Few papers reported differences in composition across income gradients, making it difficult to draw broader conclusions about whether low- and high-income urban properties often differ significantly (as with flowering communities in Avolio et al., 2020) or are similar (as with total yard and tree communities in Cubino et al., 2019; Avolio et al., 2020) given that they draw on the same regional species pool.

The main goal of this study was to evaluate associations between wealth and plant structure, richness, and composition in residential urban spaces Baltimore, MD, and the Washington, D.C. metropolitan area. Prior studies often note income as a driver of plant communities, but rarely examine multiple aspects of vegetation or do so across a relevant and defined range of income. Nor do any studies exist, to our knowledge, that compare plant communities by occupation status in these cities at a fine scale. Thus, our inclusion of abandoned properties, which have no income, is also novel. We hypothesize that vegetation cover and species richness vary systematically with income and occupation status, and that low-income and abandoned properties harbor distinct plant communities characterized by more vines and fewer native species. By comparing canopy and ground structure, richness, and composition across these gradients, our study aims to fill a key knowledge gap urban ecology. These fine-resolution data are necessary to properly assess differences in residential urban green space along socioeconomic gradients, and the first step to provide further insight into the potential services or disservices plant communities offer across heterogeneous urban landscapes, though direct measurements of impact were beyond the scope of this study.

Materials and methods

Study area

Our study took place in the mid-Atlantic region of the eastern United States, a temperate area with a humid subtropical climate and four distinct seasons (Bigsby et al., 2014; Jiang et al., 2022). Average temperature is 0.8°C in winter and 26.0°C in summer with approximately 120 cm of rain annually (Anderson et al., 2021), which is relatively evenly distributed throughout the year (Woods et al., 1999). The annual growing season is 160–225 days (Woods et al., 1999). The ecoregion is characterized as Eastern Temperate Forest: Southeastern Plains; the sections of Baltimore, MD, and the Washington, D.C. metropolitan area that we surveyed are further categorized as Chesapeake Rolling Coastal Plain (EPA, 2024). The Chesapeake Rolling Coastal Plain ecoregion is a hilly upland with elevations below 122 m and local relief of 7.6–69 m (Woods et al., 1999). Soils are typically well-drained, nutrient-poor, loamy soils that support predominantly Oak-Hickory-Pine and Appalachian Oak Forests in natural areas (Woods et al., 1999).

We surveyed plant communities on nine city blocks in each of two watersheds in the Washington-Baltimore metropolitan area: Watershed 263 and the Watts Branch watershed (Figure 1). Watershed 263 includes several neighborhoods in West Baltimore, MD, including Franklin Square, Harlem Park, Hollins Market, and Union Square. The Watts Branch watershed covers portions of Capitol Heights, MD, and southeast Washington, D.C. The watersheds, while located in different cities, are close enough for plants to be from the same regional species pool.

Given that our study takes place exclusively on private residential land, there is no municipal management of the plant communities we surveyed. While all three cities have ordinances

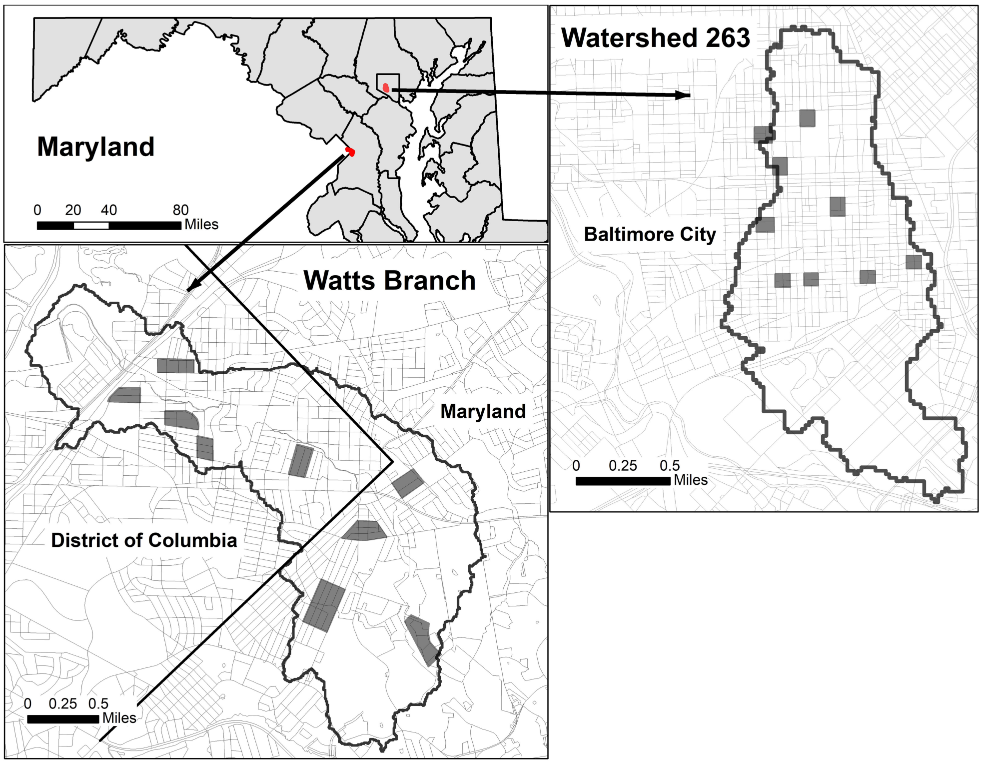

FIGURE 1
We surveyed nine blocks, shown as gray polygons, in each of two watersheds: Watershed 263, in West Baltimore, MD, and the Watts Branch Watershed, spanning southeast Washington, D.C. and Capitol Heights, MD.

regarding private vegetation (DCDOB, 2023; BDHCD, 2024; CHPSA, n.d), including limiting grass, weed, and untended plant growth to below 8 or ten inches and prohibiting noxious species (e.g., poison ivy), we observed such violations without witnessing any municipal intervention in response over the time of the study. Thus, while policies pertaining to private land vegetation exist, they do not appear to be heavily enforced. Even publicly-owned lots are often unmanaged due to insufficient funds for monitoring and maintenance (LaDeau et al., 2013). Large, managed public green spaces (Carroll, Druid Hill, and Leakin Parks in Watershed 263; Fort Circle and Kenilworth Parks in Watts Branch Watershed) were a minimum of 0.51 km (median: 1.64 km) from our study blocks and therefore unlikely to substantially influence the private plant communities we surveyed.

The 18 blocks surveyed represent a range of median annual household incomes in each city based on 2019 census data at the block group scale, which designates a median income for all city blocks within a given group (U.S. Census Bureau, 2020a) (Table 1). Our sample size of 18 blocks is consistent with other fine-scale urban vegetation studies (Lewis et al., 2017; Avolio et al., 2020; Blanchette et al., 2021) and was designed to capture a wide socioeconomic gradient while maintaining feasibility for detailed surveys. The specific blocks were chosen as part of longer-term studies focused on urban mosquito ecology (e.g., LaDeau et al., 2013; Little et al., 2017; Leisnham et al., 2021) and green infrastructure (Biehler et al., 2025). Overall, Baltimore study blocks represent lower median annual household incomes ($12,915–50,736) in this study and compared to the 2015–2019 City-wide median annual household income of $50,379 (U.S. Census Bureau, 2020b). Capitol Heights study blocks represent higher median annual household incomes ($62,443–85,781) in this study, though the values are generally below Prince George's County's 2015–2019 median annual household income of $84,920 (U.S. Census Bureau, 2020b). The D.C. study blocks' median annual household incomes ($31,307–107,188) have the widest range, overlapping the other two cities in this study as well as representing values both below and above D.C.'s 2015–2019 median annual household income of $86,420 (U.S. Census Bureau, 2020b). We categorized each block into a relative income group of low (around or below $30,000), medium ($31,000–60,000), or high (above $61,000).

TABLE 1 The 18 study blocks, listed by city (B=Baltimore, DC= District of Columbia, CH=Capitol Heights) and block code, can be categorized into three relative groups based on the median annual household income of the census block group in which they are located (U.S. Census Bureau, 2020a).

Income category	City and block code	2019 Median annual household income ($)	Percent abandonment (%)
Low	B-HP1E	12,825	57
	B-HP3E	14,640	52
	B-US1S	25,391	38
	B-FS4S	26,061	13
	B-HP5S	27,768	53
	B-FS3W	30,441	47
Medium	DC-GAL	31,307	–
	B-HM1W	39,018	–
	B-US2W	43,878	–
	DC-EAS	49,185	–
	B-HM2N	51,736	–
	DC-JAY	51,750	–
High	CH-NOV	62,443	–
	DC-EAP	64,115	–
	CH-DUT	80,402	–
	CH-DAV	80,833	–
	CH-QUR	85,781	–
	DC-BLN	107,188	–

In addition to differences in income, in-person surveys at the block scale reveal that the study blocks display stark differences in percent infrastructural abandonment, even over very short distances (Table 1) (Saunders, 2020). All properties designated as abandoned in this study have been abandoned since at least 2019 (Saunders, 2020). We define percent abandonment as the percent of properties on a given block that have uninhabited, boarded-up buildings; abandoned infrastructure is also often in a state of decay. Most studies of unoccupied urban properties in the literature highlight vacant lots, defined as having no buildings; such lots may have histories including "greening" efforts like planting grass, shrubs, or trees, and possibly even a degree of ongoing maintenance that differentiate them from the abandoned properties in our study (e.g., Branas et al., 2011; Kremer et al., 2013; Nassauer et al., 2021). Baltimore is a post-industrial city with a population that has declined nearly 40% since 1950 (Anderson et al., 2021) and displays uneven patterns of economic investment (e.g., Boone et al., 2009; Grove et al., 2018; Biehler et al., 2025) that have led to large differences in abandonment by neighborhood. In West Baltimore, more than half of the properties on Harlem Park study blocks are abandoned; in contrast, study blocks in the Hollins Market neighborhood a mile away have approximately 10% or less abandonment (Saunders, 2020; Biehler et al., 2025). Population decline has been less severe in Washington, D.C.—approximately 14% since 1950 (EdScape, 2019)—and abandonment on study blocks in the Watts Branch watershed is uncommon (Saunders, 2020).

Vegetation surveys

Vegetation surveys were carried out on the block scale to align with U.S. Census block group income and block-level abandonment data. We conducted vegetation surveys on five occupied properties per block and three abandoned properties on each of the six blocks with the highest levels of abandonment. For property selection, all addresses on a block were randomized, and we knocked on the doors of the first five occupied properties, in each case moving to the next occupied property to the right until a resident granted access to their yard. Similarly, we visited the first three abandoned properties on the list, moving to the right when a property was inaccessible. We aimed to avoid surveying adjacent properties to ensure that vegetation surveys would be spatially representative of a block. Low occupation and/or participation rates required allowances of two neighboring properties on five blocks.

We conducted one to three paired canopy and ground surveys per property, depending on lot size and suitable survey area, for a total of 241 paired surveys. Conducting more surveys on larger properties allowed us to capture a similar proportion of each property and maintain a consistent representation of each block. As part of a more extensive study on mosquito ecology, survey center points were chosen to be level, shaded spots where mosquitoes could breed should a water-holding container be present. Survey center points were also selected to have ≥ 2 m of open space above the ground and to be ≥ 4 m apart to ensure ground surveys would not overlap. All ground and canopy surveys occurred between July 16th and September 2nd, 2020.

For canopy surveys, a photo was taken approximately 1 m above the ground over each survey center point using CanopyApp[1], a tool developed by the University of New Hampshire to assess percent canopy cover. Every individual plant captured in the canopy photo was then identified to species, with few exceptions (Supplementary Table S1). Two individuals could only be reliably identified to genus, and two individuals could not be identified at all due to height and property boundaries making close inspection unfeasible. The USDA PLANTS Database (plants.usda.gov) was used for species name, growth habit, and native status determination, with the exceptions of *Photinia fraseri* and *Salvia yangii*, both observed on one property each. *Photinia fraseri* is a hybrid between *P. glabra* and *P. serrulata*, recognized by many university extension offices but not yet on the PLANTS Database. *Slavia yangii* is the updated name for *Perovskia atriplicifolia*.

For ground surveys, two measuring tapes were laid out to mark up to 16 m^2 around the survey center point. Property boundaries, buildings, or other obstacles sometimes restricted the accessible ground area so that not all ground surveys were a full 16 m^2; the mean ground survey area across all blocks was 12.8 ± 3.4 m^2. Within each survey area, we visually estimated the percentage of the accessible ground area that was vegetated. The USDA categorizes plants broadly into eight growth habits: lichenous, nonvascular, forb/herb, graminoid, subshrub, shrub, tree, and vine (USDA, 2022). All but lichens, which are not true plants, were included in the visual estimate of vegetative cover. Of those included in coverage estimates, only vascular growth habits were counted toward species richness. While herbaceous and graminoid species richness was tallied, species identities were not documented. For plants with woody growth habits (subshrub, shrub, tree, vine), we identified each individual (Supplementary Table S1). Of several thousand woody individuals, 211 could only be reliably identified to genus, and five individuals could not be identified at all; the rest were identified to species. A plant that could only be identified to genus was included in its survey richness count whenever it was the only individual in its genus present, and thus represented a new species for the survey regardless of species identity (e.g., a survey with an unknown oak and no other oaks present). Instances of an individual plant identified only to genus with another representative of the genus documented in the same survey (e.g., an unknown oak in the same survey as a documented *Quercus falcata*) were excluded to avoid artificially inflating richness counts, as it is likely the former was a seedling of the latter. Again, the PLANTS Database was used for species names, growth habit, and native status determination.

1 https://apps.apple.com/us/app/canopyapp/id926943048

Statistical analyses

All statistical analyses were conducted using R Statistical Software (v4.2.2, R Core Team, 2021). Statistical significance was evaluated at $p \leq 0.05$ and 0.1. For each model, we confirmed all data assumptions were met, including spatial structure. Comparisons along an income gradient or between relative income categories included only occupied properties, as no income is associated with an abandoned property. Income is used as our fixed effect due to its well-known association with other socioeconomic variables such as race and ethnicity (e.g., Akee et al., 2019), education (e.g., Blaug, 1972), lifestyle (e.g., Zhou et al., 2009), population and housing density (e.g., Hummel, 2020), and parcel size (e.g., Avolio et al., 2018). Comparisons by occupation status included only occupied properties on the same six blocks as the surveyed abandoned properties.

Structure

Differences in percent canopy and ground cover per survey along an income gradient were analyzed using a generalized linear mixed model with the lmerTest package (v3.1.3, Kuznetsova et al., 2017) with property as a random intercept effect to account for spatial non-independence due to multiple surveys per property. Differences in structure by occupation status were analyzed using a two-sample t-test.

Three surveys were missing a documented percent vegetated ground cover and removed from structure analyses; one of those three surveys was also missing ground survey area and was subsequently removed from richness and community composition analyses.

Richness

Richness—total, by growth habit, and by native status—was analyzed per survey for canopy species (as all canopy surveys were the same area) but standardized per m^2 for ground species (as ground surveys were different areas) to account for differences in sampling effort. Differences in richness along an income gradient were analyzed using a Poisson regression with the lme4 package (v1.1.31, Bates et al., 2015) for canopy data and a generalized linear mixed model with the lmerTest package (v3.1.3, Kuznetsova et al., 2017) for ground data, all with property as a random intercept effect to account for spatial non-independence due to multiple surveys per property. Differences in canopy and ground richness by occupation status were analyzed using two-sample t-tests.

Three surveys (two canopy, one ground) contained individuals that could not be identified even to the genus level and were marked as unknowns. In one canopy survey, the unknown individual was the only plant in the canopy; it therefore represented a novel species for that survey and was included in total richness analyses. As we could not be certain whether the other unknowns represented novel species, we tested for differences in total richness in three ways: without the unknowns, with the unknowns counted as one additional species in the survey, and with the unknowns counted as though each individual belonged to a different species. We found no changes in statistical significance. To be conservative, in case the unknown individuals belonged to a species already accounted for, we present results for total canopy and ground richness without them. Similarly, without an identity, no unknown individual could be included in richness analyses by growth habit or native status. Finally, one ground survey lacked a documented herbaceous species richness and was excluded from total and herbaceous growth habit ground richness analyses.

Community composition

We assessed community composition based on species frequency (presence or absence of a species in a survey) rather than abundance (total number of individuals of a species found per survey), standardizing by m^2 for ground data. We did not base community composition analyses on species abundance in part due to difficulties determining which vines were connected. Additionally, time limitations during field surveys prevented documentation of the age/size of each individual plant. Consequently, a species with a large number of seedlings present in a survey could skew evenness and diversity index comparisons, appearing to be more influential in a community than a species with a single, mature tree present in the survey, even though the latter would cover more area and contribute substantially more to ecosystem services such as carbon sequestration and temperature regulation. Thus, we felt that abundance would not accurately represent the green spaces we surveyed. We used frequency tables to reflect evenness based on frequency data, showing the proportion of surveys or m^2 in which common species appeared within an income bracket or occupation status.

To discern differences in canopy and ground community composition—total, by growth habit, and by native status—between income categories and by occupation status, non-metric multidimensional scaling and permutational analyses of variance were done using the vegan package (v2.6.4, Oksanen et al., 2025). To assemble the NMDS, we used the function metaMDS with two dimensions and trymax=100. To create the distance matrix with the function vegdist, we used Bray-Curtis dissimilarity. All PERMANOVAs were calculated with the adonis2 function and 999 permutations. For each PERMANOVA, we checked for differences in group dispersion, but none were significant and thus we do not report them alongside differences in group centroids. Individuals identified only to the genus level that were included in total richness analyses were similarly retained here; observations of plants unknown even at the genus level were omitted.

In analyses of canopy community composition separated by native status, some blocks could not be included due to lack of data. In the native canopy analysis by income category, FS4S (low-income) had no native species; in the non-native canopy analysis by income category, HM2N (medium-income) had no non-native species. Similarly, in the native canopy analysis by occupation status, US1S abandoned properties had no native species in addition to the aforementioned lack of native species in occupied FS4S canopies.

Results

Canopy

Income

Canopy cover on occupied properties ranged from 0.0-96.6%, with mean cover increasing significantly with increasing income (coef=1.94, $F_{1,80.9}$ = 5.09, p=0.03) (Figure 2A). We identified 70 canopy species across all occupied properties. Canopy richness per survey ranged from 0–6 species, with mean richness increasing significantly with increasing income (coef=0.02, z=6.87, p<0.01) (Figure 2B). When separated by growth habit, the positive relationship between total canopy richness and income seems to be driven by increasing mean tree richness along an income gradient (coef=0.04, z=1.98, p=0.05) (Figure 2C), as only one herbaceous and one shrub species grew tall enough to be captured in canopy photos and vine richness did not change significantly with income (z=-1.16, p=0.25) (Figure 2D). Separated by native status, native canopy richness per survey increased significantly with increasing income (coef=0.11, z=2.86, p<0.01), while non-native canopy richness declined (coef=-0.10, z=-2.49, p=0.01) (Figures 2E, F).

Canopy community composition on low-income blocks was distinct from that of high-income blocks, while medium-income blocks shared similarities with both, based on ordination analysis (Figure 3A). Income explained 22% of variation in the overall canopy, with marginally significant differences between income categories ($F_{2,15}$ = 2.08, p=0.06). This overall variation seems to be the result of significant differences in tree ($F_{2,15}$ = 2.26, p=0.03), vine ($F_{2,15}$ = 2.38, p=0.04), and non-native ($F_{2,14}$ = 2.89, p=0.02) communities, for which income explained 23%, 24%, and 29% of variation, respectively (Figures 3B–D). Though native canopy communities also appeared to be distinct between low- and high-income blocks based on ordination (Figure 3E), there were no statistically significant differences between income categories ($F_{2,14}$ = 1.69, p=0.12).

Half of the most frequently observed canopy species (those appearing in 10% or more of surveys) in the low-income category were vines (Table 2). By contrast, vines were not among the most frequently observed canopy species in the medium-income category, and there was only one vine among the most frequently observed canopy species in the high-income category. Additionally,

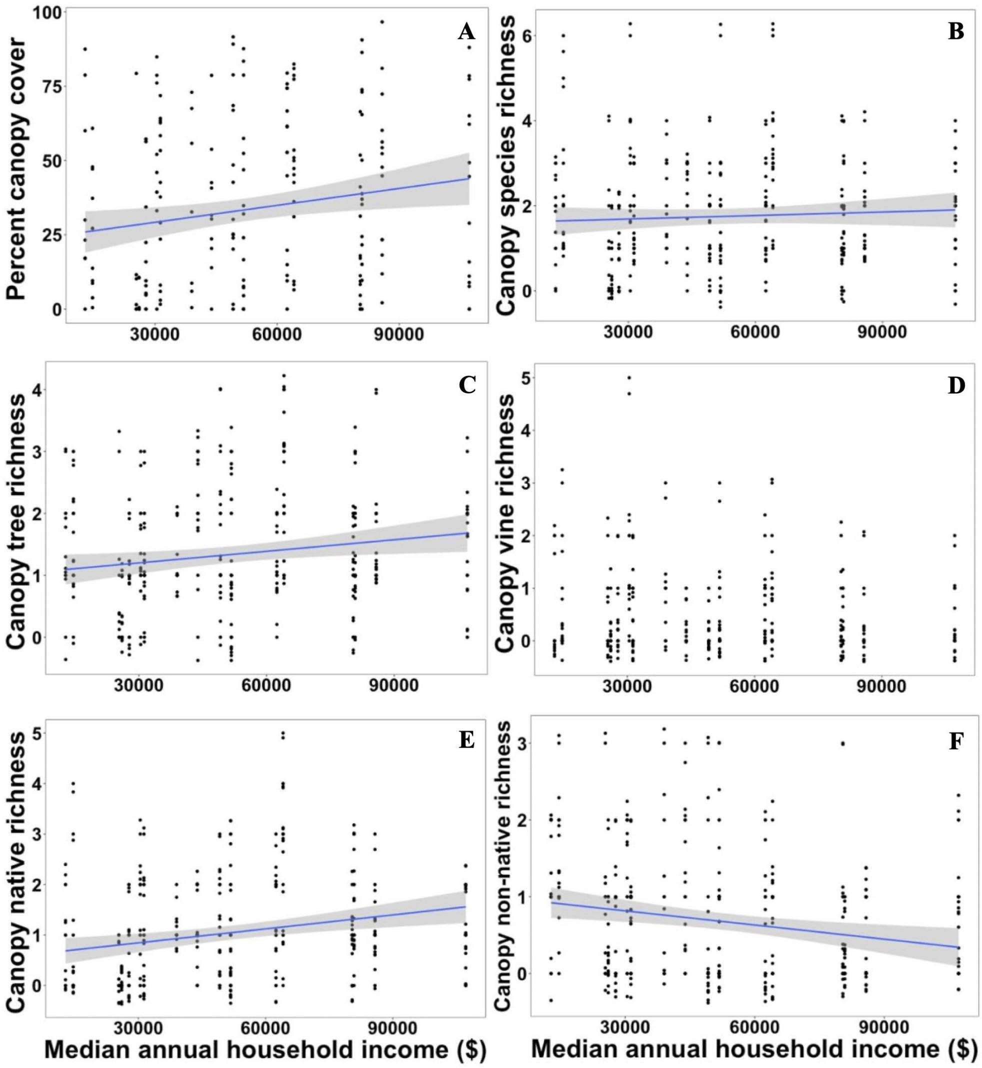

FIGURE 2
Average canopy survey **(A)** cover and **(B)** total richness on occupied properties increased significantly with increasing median annual household income. Separated by growth habit, average richness of **(C)** trees in canopy surveys increased significantly along an income gradient, with no significant difference in **(D)** vine species richness. **(E)** Average native species richness also increased along an income gradient, while **(F)** average non-native species richness decreased. The shaded areas represent 95% confidence intervals. A jitter was applied to the data to avoid multiple points overlapping; all values are positive integers.

the most frequently observed species in the low-income category were mostly non-native (4/6), while the reverse was true of high-income blocks (4/6 native); of the two species to appear in 10% or more of canopy surveys on medium-income blocks, one was native and one non-native. *Morus alba*, a non-native tree, was the only species to appear in 10% or more of canopy surveys in each income category.

Occupation status

While canopy cover ranged from 0% to approximately 88% on both abandoned and neighboring occupied properties (Figure 4A), mean cover on abandoned properties (42.9 ± 27.3%) was significantly higher (t_{85} = 3.18, p<0.01) than mean cover on neighboring occupied properties (23.1 ± 27.0%). Of the 33 known canopy species observed on the six Baltimore blocks with high abandonment, 22 were documented on abandoned properties (28 surveys) and 26 species plus an unknown individual were documented on neighboring occupied properties (59 surveys). Like cover, the range of canopy richness per survey was similar on properties of different occupation status (abandoned: 0–5 species, neighboring occupied properties: 0–6 species) but mean richness was significantly greater (t_{85} = 2.89, p<0.01) on abandoned properties (2.5 ± 1.4) compared to neighboring occupied properties (1.5 ± 1.5) (Figure 4B). Separated by growth habit and native status, increased total richness on abandoned properties seems to be the result of increased mean tree (t_{85} = 2.62, p=0.01), vine (t_{85} = 1.77, p=0.08), and non-native (t_{85} = 3.32, p<0.01) species richness. Abandoned properties had, on average, 1.5 ± 0.8 tree species, 0.9 ± 0.9 vine species, and 1.7 ± 1.2 non-native species, while neighboring occupied properties only had 1.0 ± 0.9, 0.5 ± 1.0, and

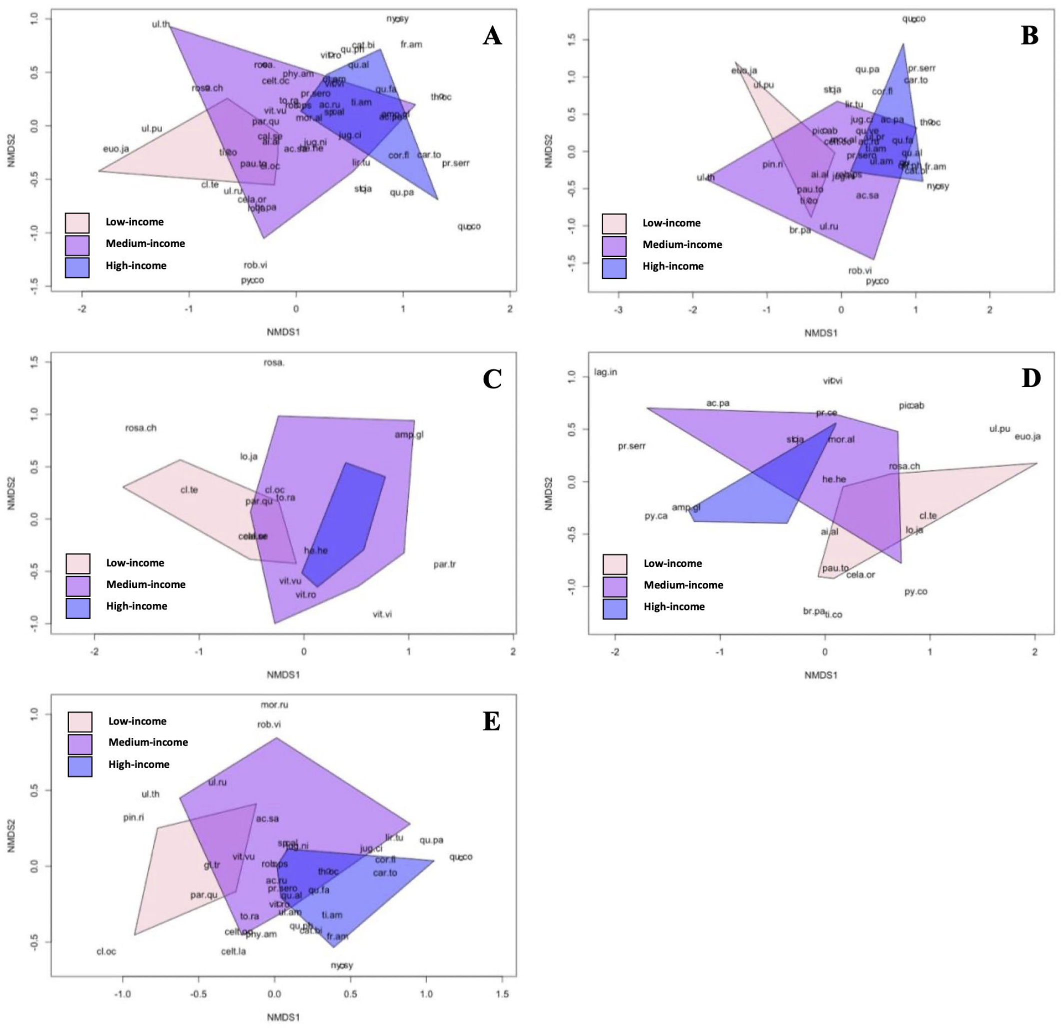

FIGURE 3
The canopy taxa documented on occupied parcels are abbreviated in black (abbreviations explained in Supplementary Table S-1). Each polygon represents a relative income category, with blocks creating the points. The model arranges blocks based on similarity; the closer together two blocks are, the more similar their canopies are, and the farther apart they are, the more different their canopy community. **(A)** Overall, canopies were significantly different between income categories, with composition on low-income blocks distinct from those on high-income blocks, and overlap in the middle. Separated by growth habit and native status, **(B)** trees, **(C)** vines, and **(D)** non-native canopy communities were also significantly different between income categories. **(E)** While native canopy communities appeared visually to be distinct between low- and high-income blocks, there were no significant differences.

0.9 ± 0.9 tree, vine, and non-native species, respectively (Figures 4C–E). Mean native species richness was similar (t_{85} = 0.72, p=0.47) between abandoned (0.8 ± 1.2) and neighboring occupied (0.6 ± 1.0) properties (Figure 4F).

Based on ordination analysis, canopy communities were not distinct between abandoned and neighboring occupied properties, with no statistically significant difference by occupation status ($F_{1,10}$ = 0.02, p=0.94). No differences emerged when separating by growth habit or native status, with similarities in tree ($F_{1,10}$ = -0.01, p=0.99), vine ($F_{1,10}$ = 1.54, p=0.25), native ($F_{1,10}$ = 1.38, p=0.26), and non-native ($F_{1,10}$ = 0.30, p=0.85) communities. In fact, six of the 10 most frequently observed canopy species on the six blocks in the low-income category were common to both abandoned and neighboring occupied properties (Table 2). Most species were non-native, and half were vines.

Ground

Income

Vegetated ground cover on occupied properties ranged from 0–100% (Figure 5A), with mean percent cover increasing significantly with increasing income (coef=6.55, $F_{1,80.9}$ = 36.22, p<0.01). We observed 113 woody species (shrub, tree, or vine) across all ground surveys on occupied properties and two unique woody genera with no individuals identified to a species level. Total ground species richness per m^2 (woody and non-woody combined) ranged from 0–3.6 species, with no significant difference in richness along the income gradient ($F_{1,80.9}$ = 1.76, p=0.19), though the minimum richness observed per m^2 was greater at higher incomes (Figure 5B). This seems to be largely due to higher minimum herbaceous species richness (Figure 5C). Both mean herbaceous

TABLE 2 Most plant species that appeared in 10% or more of canopy surveys on occupied properties of low-income blocks were non-native (bolded), while the most frequently occurring canopy species in the high-income category were largely native. More species of vines (underlined) appeared in canopies in the lowest two income categories compared to canopies on high-income blocks, which mostly contained trees as the most frequent growth habit. All six species that appeared in 10% or more of canopy surveys on occupied properties of low-income blocks were also among the most frequently-documented canopy species on abandoned properties on those same blocks. Four additional species appeared in 10% or more of surveys on abandoned properties, half of which were non-native and most of which were vines. Besides vines, trees were the other frequently-documented canopy growth habit.

Relative income category or occupation status	Species (survey frequency)									
Abandoned	***Morus alba*** (43%)	***Celastrus orbiculatus*** (32%)	***Paulownia tomentosa*** (32%)	***Ailanthus altissima*** (14%)	*Parthenocissus quinquefolia* (14%)	***Ulmus pumila*** (14%)	*Vitis vulpina* (14%)	*Celtis laevigata* (11%)	**Clematis terniflora** (11%)	***Hedera helix*** (11%)
Low	***Ailanthus altissima*** (22%)	***Paulownia tomentosa*** (15%)	***Morus alba*** (14%)	*Vitis vulpina* (12%)	***Celastrus orbiculatus*** (10%)	*Parthenocissus quinquefolia* (10%)	–	–	–	–
Medium	***Morus alba*** (24%)	*Ulmus americana* (12%)	–	–	–	–	–	–	–	–
High	*Ulmus americana* (25%)	***Morus alba*** (17%)	*Catalpa bignonioides* (14%)	*Acer rubrum* (11%)	***Hedera helix*** (11%)	*Juglans nigra* (10%)	–	–	–	–

Species in bold are non-native; underlined species are vines.

(coef=0.03, $F_{1,80.6}$ = 6.34, p=0.01) and shrub (coef=0.004, $F_{1,83.4}$ = 8.49, p<0.01) (Figure 5D) species richness increased with increasing income. There were no differences in mean tree ($F_{1,86.4}$ = 0.06, p=0.81), vine ($F_{1,83.3}$ = 2.64, p=0.11), or native species richness ($F_{1,88.0}$ = 0.0001, p=0.99) along in income gradient (Figures 5E–G), and a marginally significant decrease in non-native species richness (coef=-0.01, $F_{1,82.8}$ = 3.20, p=0.08) (Figure 5H).

Woody ground community composition on low-income blocks was distinct from that of high-income blocks, while medium-income blocks shared similarities with both, based on ordination analysis (Figure 6A). Income explained 22% of variation, with significant differences between income categories ($F_{2,15}$ = 2.08, p=0.01). This overall variation seems to be the result of significant differences in tree ($F_{2,15}$ = 2.19, p=0.01), vine ($F_{2,15}$ = 2.36, p=0.02), native ($F_{2,15}$ = 1.79, p=0.02), and non-native ($F_{2,15}$ = 2.47, p<0.01) communities, for which income explained 22%, 24%, 19%, and 25% of variation, respectively (Figures 6B–E). There were not enough data to analyze shrub communities; however, their absence was not uniform across income categories, with shrubs documented in only two low-income blocks compared to five medium-income blocks and all six high-income blocks.

Non-native vine *Hedera helix* and native vine *Parthenocissus quinquefolia* were the most frequently observed woody ground species in all income categories (Table 3). Other woody ground species were less similar between income categories, with only non-native vine *Ampelopsis glandulosa* appearing in 2% or more of one square meter segments of ground surveys in more than one category (both medium- and high-income blocks). All frequently observed woody ground species were either vines (8/11) or non-native (7/11), if not both (4/11).

Occupation status

Vegetated ground cover ranged from 0.5% or 0% to 100% on abandoned and neighboring occupied properties, respectively (Figure 7A), with no significant difference in mean cover by occupation status (t_{85} = 1.18, p=0.24). We observed 52 woody species across ground surveys on the six Baltimore blocks with the most abandonment, as well as two unique genera with no individuals identified to a species level. Of these, 36 species and a unique genus were documented on abandoned properties (28 surveys) and 38 species and both unique genera were documented on neighboring occupied properties (59 surveys). Total richness per m^2 (woody and non-woody combined) ranged from 0-3.6 species (Figure 7B), with no significant difference by occupation status (t_{85} = -0.60, p=0.55). The lack of significance for all species combined is likely due to opposing relationships for herbaceous and tree species richness and non-significance for other groups. Abandoned properties had fewer herbaceous species (t_{85} = -1.82, p=0.07) on average (0.5 ± 0.4 species per m^2 compared to 0.6 ± 0.4 species per m^2 on neighboring occupied properties) but more tree species (t_{85} = 2.20, p=0.03) on average (0.2 ± 0.1 species per m^2 compared to 0.1 ± 0.1 species per m^2 on neighboring occupied properties) (Figure 7C, D). There was no significant difference in mean vine species richness per m^2 by occupation status (t_{85} = 0.61, p=0.54), and we did not test for statistical differences in shrubs as only four individuals were documented—two on abandoned properties and two on neighboring occupied properties (Figures 7E, F). Nor were there any significant differences in richness separated by native status, with similar mean native (t_{85} = 1.52, p=0.13) and non-native (t_{85} = 1.12, p=0.26) species richness per m^2 on both abandoned and neighboring occupied properties (Figures 7G, H).

Based on ordination analysis, woody ground communities were not distinct between abandoned properties and neighboring occupied properties, and there were no statistically significant differences ($F_{1,10}$ = 0.59, p=0.86). No further differences emerged when separating by growth habit or native status, with similarities in tree ($F_{1,10}$ = 0.33, p=0.98), vine ($F_{1,10}$ = 1.02, p=0.51), native

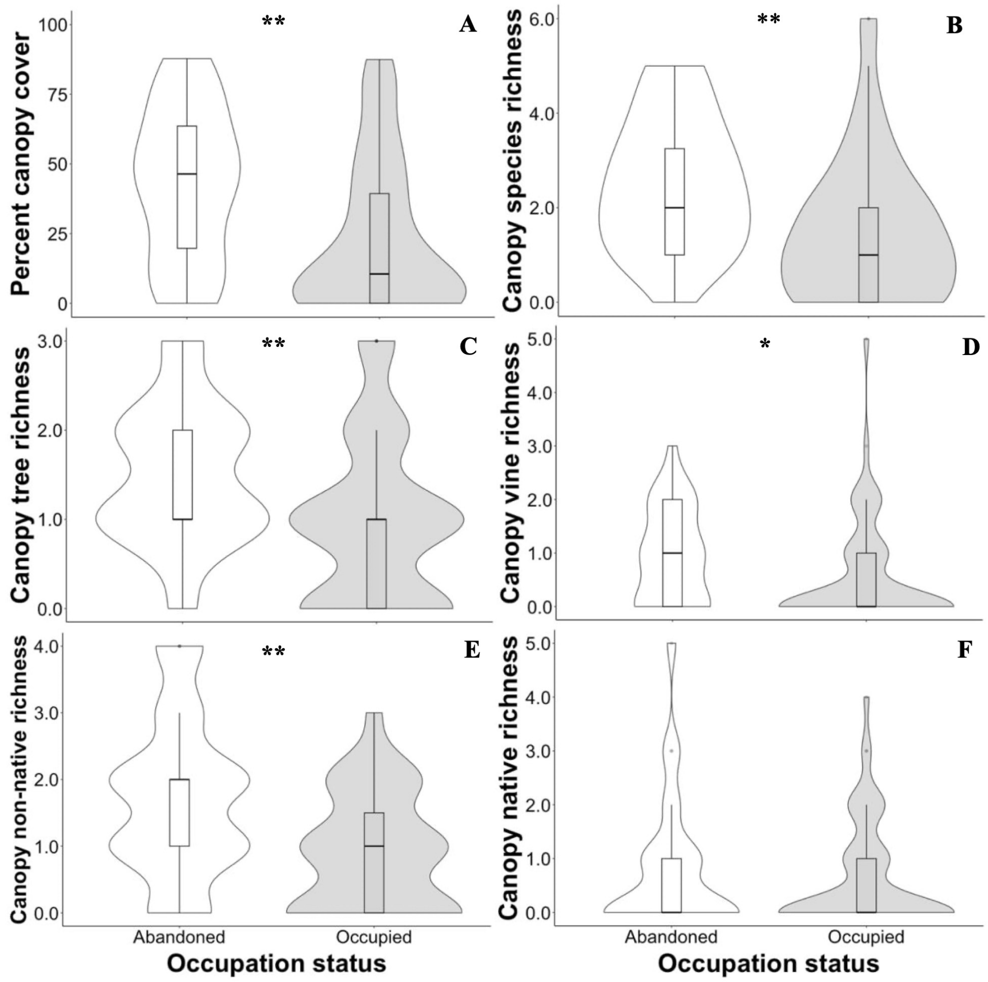

FIGURE 4
Mean **(A)** percent canopy cover, **(B)** total canopy species richness, **(C)** tree richness, **(D)** vine richness, and **(E)** non-native species richness were all greater on abandoned properties compared to neighboring occupied properties on blocks with substantial abandonment rates. **(F)** There were no differences in native species richness by occupation status. Statistical significance (p < 0.10) is indicated by an asterisk; a double asterisk indicates significance of p ≤ 0.05.

($F_{1,10}$ = 0.10, p=1.00), and non-native ($F_{1,10}$ = 1.00, p=0.46) communities between abandoned and neighboring occupied properties.

All six of the most frequently observed species on abandoned properties were also among the most frequently observed on neighboring occupied properties (Table 3). In particular, the vines *C. orbiculatus*, *H. helix*, and *P. quinquefolia* were the three most frequently documented woody species in ground surveys on the six low-income blocks for both occupation statuses. In fact, five of the six most frequently documented species on abandoned properties were vines, and four were non-native. Only two additional species were observed frequently on low-income occupied properties—a non-native tree and a native vine.

Discussion

We found that both canopy and ground vegetation on lower-income residential properties in Baltimore, MD, and the Washington, D.C. metropolitan region covered less area than their higher-income counterparts. We recorded an 18% increase in average canopy cover and an almost 62% increase in average ground cover between the lowest- and highest-income blocks (a $95,000 gap). Furthermore, low ground cover on lower-income properties was often the result of paved surfaces, while low ground cover on higher-income properties was more often the result of bare soil, offering the possibility of even more vegetative ground cover in higher-income neighborhoods, and greater disparity, over time. These results are consistent with other urban canopy (e.g., Clarke et al., 2013; Avolio et al., 2015; Wheeler et al., 2022) and ground (Clarke et al., 2013; Lewis et al., 2017) surveys from major cities including Los Angeles, CA, Phoenix, AZ, and New Orleans, LA, that also found a positive relationship between vegetated cover and income.

We also found that total canopy richness increased with increasing income, as in the seminal research by Hope et al. (2003) in Phoenix, AZ; however, in our study, this likely resulted

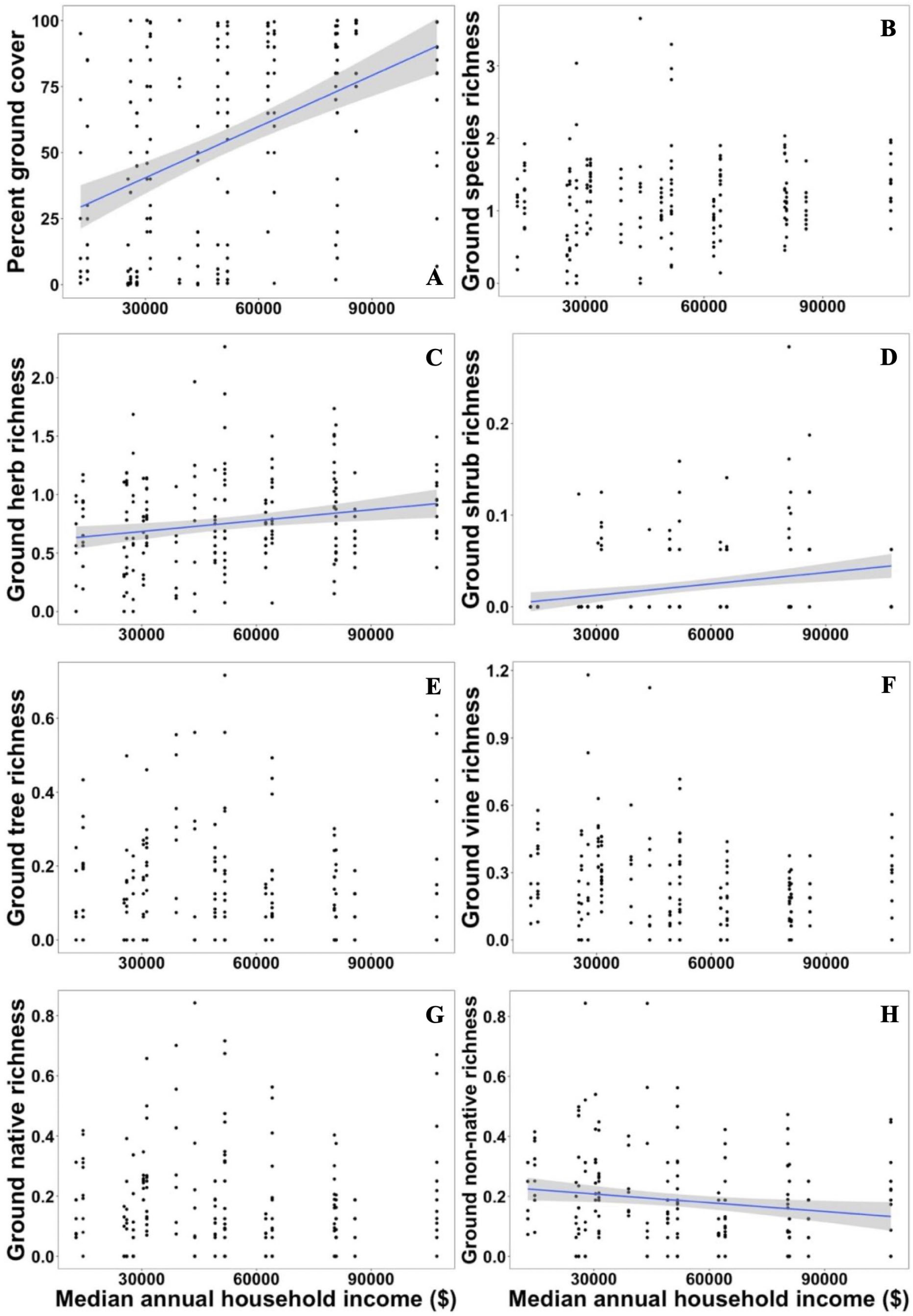

FIGURE 5
(A) Percent vegetated ground cover increased significantly with increasing median annual household income. **(B)** There was no difference in total ground species richness per m^2 along an income gradient. Separated by growth habit, both **(C)** herbaceous and **(D)** shrub species richness per m^2 increased with increasing income, with no change in **(E)** tree or **(F)** vine species richness per m^2. Separated by native status, there was no difference in **(G)** native species richness along an income gradient while **(H)** non-native species richness decreased significantly with increasing income. The shaded areas represent 95% confidence intervals. A jitter was applied to the data to avoid multiple points overlapping; all values are positive.

from fewer low-richness plots on higher-income properties, as opposed to consistent high-richness surveys. Interestingly, we found no significant association between total ground species richness and income, counter to past findings of a negative ground diversity-income relationship in cities like Chicago, IL, Boston, MA, and Miami, FL (Lowenstein and Minor, 2016; Wheeler et al., 2017; Blanchette et al., 2021). Those researchers have suggested that a negative relationship between ground species

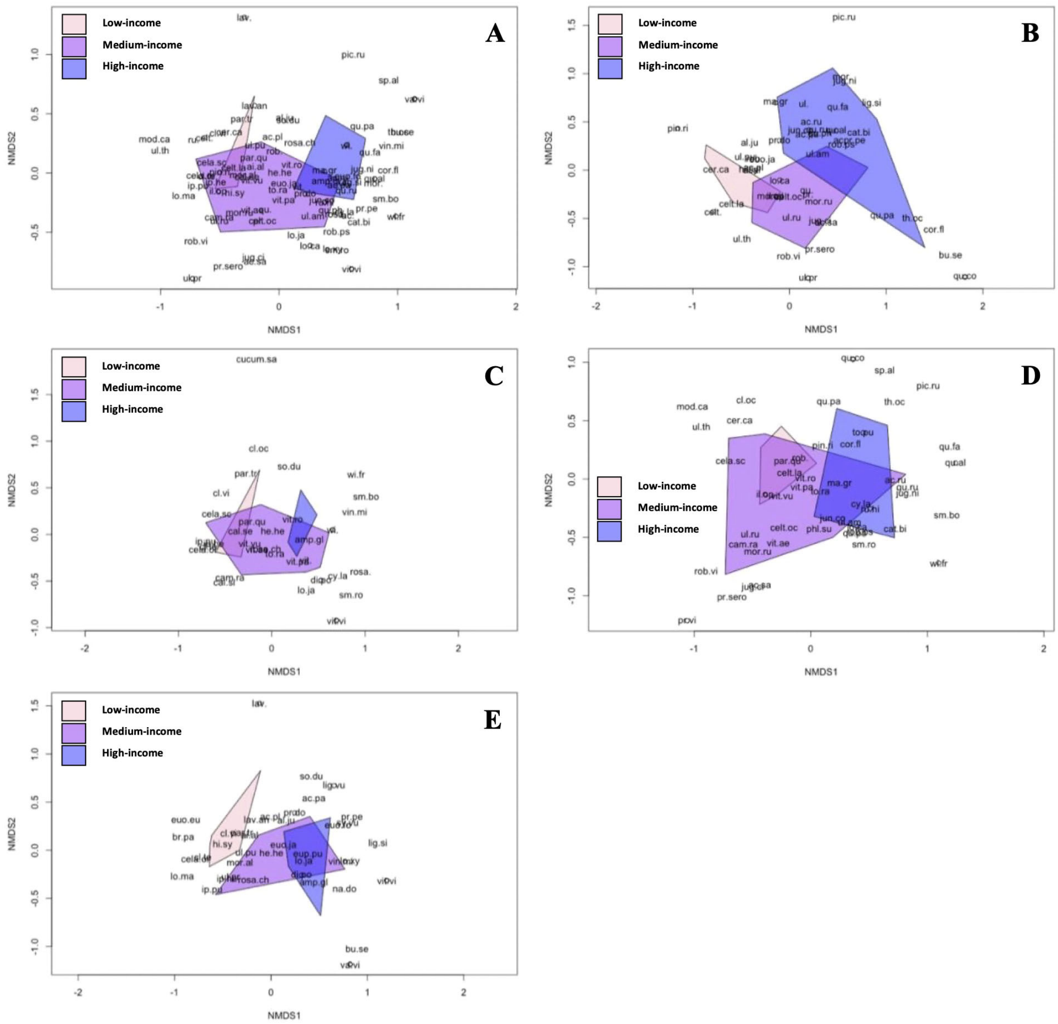

FIGURE 6
Woody ground taxa documented on occupied parcels are abbreviated in black (abbreviations explained in Supplementary Table S-1). Each polygon represents a relative income category, with blocks creating the points. The model arranges blocks based on similarity; the closer together two blocks are, the more similar their woody ground communities are, and the farther apart they are, the more different. **(A)** Overall, woody ground communities were significantly different between income categories, with composition on low-income blocks distinct from those on high-income blocks, and overlap in the middle. Differences in woody ground community composition were evident and significant for both **(B)** trees and **(C)** vines between low- and high-income blocks. Woody ground communities were also significantly different between income categories by native status, with **(D)** native communities being distinct between low- and high-income blocks and **(E)** non-native communities being distinct on low-income blocks compared to both medium- and high-income blocks.

richness and income may result from culturally-influenced expectations for monoculture lawns in wealthier neighborhoods and an accompanying increase in resources to devote to removing herbaceous weeds. Yet, we observed the opposite pattern—an average 10 m^2 plot on the highest-income block had 2.9 more herbaceous species and 0.4 more shrub species than an average 10 m^2 plot on the lowest-income block. Similarly, canopies on the highest-income block had, on average, an additional 0.4 tree species and one extra native species. Others have also found more tree species in wealthier urban neighborhoods of Baltimore County, MD, and Salt Lake City, UT (Avolio et al., 2018, 2020). Personal observations suggest that these positive relationships reflect a more actively managed landscape with gardens and ornamentals, compared to one in which a few opportunistic species take over. Only non-native species richness decreased with increasing income, with the highest-income block likely to have one fewer non-native species per canopy survey and 10 m^2 ground plot compared to canopy and ground communities on the lowest-income block. We are the first, to our knowledge, to report more native species and fewer non-native species in residential urban green spaces with increasing income, which is in line with results from some whole-yard plant composition studies (Lewis et al., 2017; Cavender-Bares et al., 2020), but not others (Avolio et al., 2020; Cavender-Bares et al., 2020).

Nor is there much in the literature to compare to our analyses of community composition, which was significantly different between income categories for trees, vines, and non-native species both in the canopy and on the ground; ground communities also contained

TABLE 3 The same two vines (underlined), one native and one non-native (bolded), were the most frequently-documented woody ground species across occupied blocks of all income categories. Other woody (shrub, tree, vine) species appearing in 2% or more of one square meter sections of ground surveys were less similar between income categories, but were mostly vines, with just two tree species in the lowest income category and a shrub in the highest. Half or more of the frequently-documented species within each income category were non-native. Most of the common woody ground species on low-income occupied properties also appeared frequently on abandoned properties. Vines were the most common growth habit on abandoned properties, with just one tree species, and most species were non-native.

Relative income category or occupation status	Species (frequency per m^2)							
Abandoned	***Celastrus orbiculatus*** (6.0%)	***Hedera helix*** (5.8%)	*Parthenocissus quinquefolia* (5.0%)	*Toxicodendron radicans* (3.7%)	***Clematis terniflora*** (2.9%)	***Morus alba*** (2.4%)	–	–
Low	*Parthenocissus quinquefolia* (5.3%)	***Hedera helix*** (5.0%)	***Celastrus orbiculatus*** (4.0%)	***Clematis terniflora*** (3.5%)	***Ailanthus altissima*** (3.0%)	*Toxicodendron radicans* (2.7%)	***Morus alba*** (2.1%)	*Vitis vulpina* (2.0%)
Medium	***Hedera helix*** (3.4%)	*Parthenocissus quinquefolia* (2.8%)	*Cynanchum laeve* (2.1%)	***Ampelopsis glandulosa*** (2.0%)	–	–	–	–
High	***Hedera helix*** (4.8%)	*Parthenocissus quinquefolia* (2.4%)	***Ampelopsis glandulosa*** (2.1%)	***Euonymous fortunei*** (2.0%)	–	–	–	–

Species in bold are non-native; underlined species are vines.

distinct native species composition between income categories. Though Avolio et al. (2020) reported similar tree communities in yards of different income brackets in Baltimore County, our studies used different extremes in the income gradient included, making direct comparisons difficult. The canopy species we documented most frequently on low-income blocks were more likely to be vines and/or non-native than on high-income blocks. The two most common ground species we identified across all income categories were both vines: non-native vine *Hedera helix* and native vine *Parthenocissus quinquefolia*. Collectively, our cover, richness, and composition results suggest that green spaces on occupied residential properties in higher-income urban neighborhoods should be a more effective moderator of local surface temperatures (e.g., Armson et al., 2012), increase mental health benefits (Wood et al., 2018), better support urban wildlife diversity (Berthon et al., 2021), and appear better cared for (Berland et al., 2020).

Canopies on abandoned properties differed from those on neighboring occupied properties in some important ways, but ground vegetation was largely similar. Canopies on abandoned properties had nearly twice the cover of neighboring occupied properties, supporting conclusions of prior studies in Baltimore and Detroit, MI, that abandoned properties are important contributors to block-wide canopy cover despite, or perhaps because of, a lack of management (Little et al., 2017; Endsley et al., 2018). We also found that abandoned properties contained an additional species per canopy survey primarily due to increased tree, vine, and non-native species richness. While total richness did not differ for ground vegetation by occupation status, an average 10 m^2 plot on an abandoned property contained one fewer herbaceous and one additional tree species compared to an average 10 m^2 plot on a neighboring occupied property. As others have noted, denser cover and higher richness resulting from spontaneous growth may not be desirable if vegetation is overgrown and/or non-native (e.g., Riley et al., 2018; Berland et al., 2020; Jiang et al., 2022), which the majority of the most frequently documented species on the lowest-income blocks were. Community composition did not differ significantly between abandoned and neighboring occupied properties in either the canopy or on the ground, suggesting that vegetation spreads across a block regardless of occupation status. Contextualizing these results is challenging due to the emphasis in the literature on vacant lots, as opposed to abandoned properties, in studies of unoccupied urban parcels. Without disturbance and removal from demolition, it makes sense that the quick-growing and easily spread nature of crawling vines, seedlings, and herbaceous plants could lead to a high degree of similarity in vegetation between abandoned and occupied properties on the same block.

In summation, we found that both canopy and ground vegetation on low-income residential properties were less extensive and more likely to contain non-native species than their higher-income counterparts, with distinctly different tree, vine, and non-native communities present. Abandoned properties had more canopy cover and higher tree richness compared to occupied neighbors, but were otherwise very similar. It is interesting to note that vegetation on low-income blocks—including both occupied and abandoned properties—had a high degree of similarity between ground and canopy communities, with half or more of the most frequently documented species being the same. Most of these were vines, which appeared to readily climb up tree trunks or human-made structures to create canopy in addition to ground cover. By contrast, almost none of the most frequently documented ground species on medium- and high-income blocks were among the most common canopy species and vice versa, with *H. helix* on high-income blocks as the exception. Other than *H. helix*, none of the vines common to medium- and high-income ground communities were frequently observed in the canopies, which were dominated by trees. It is likely that active resident interventions prevent the degree of recruitment that occurs on low-income blocks.

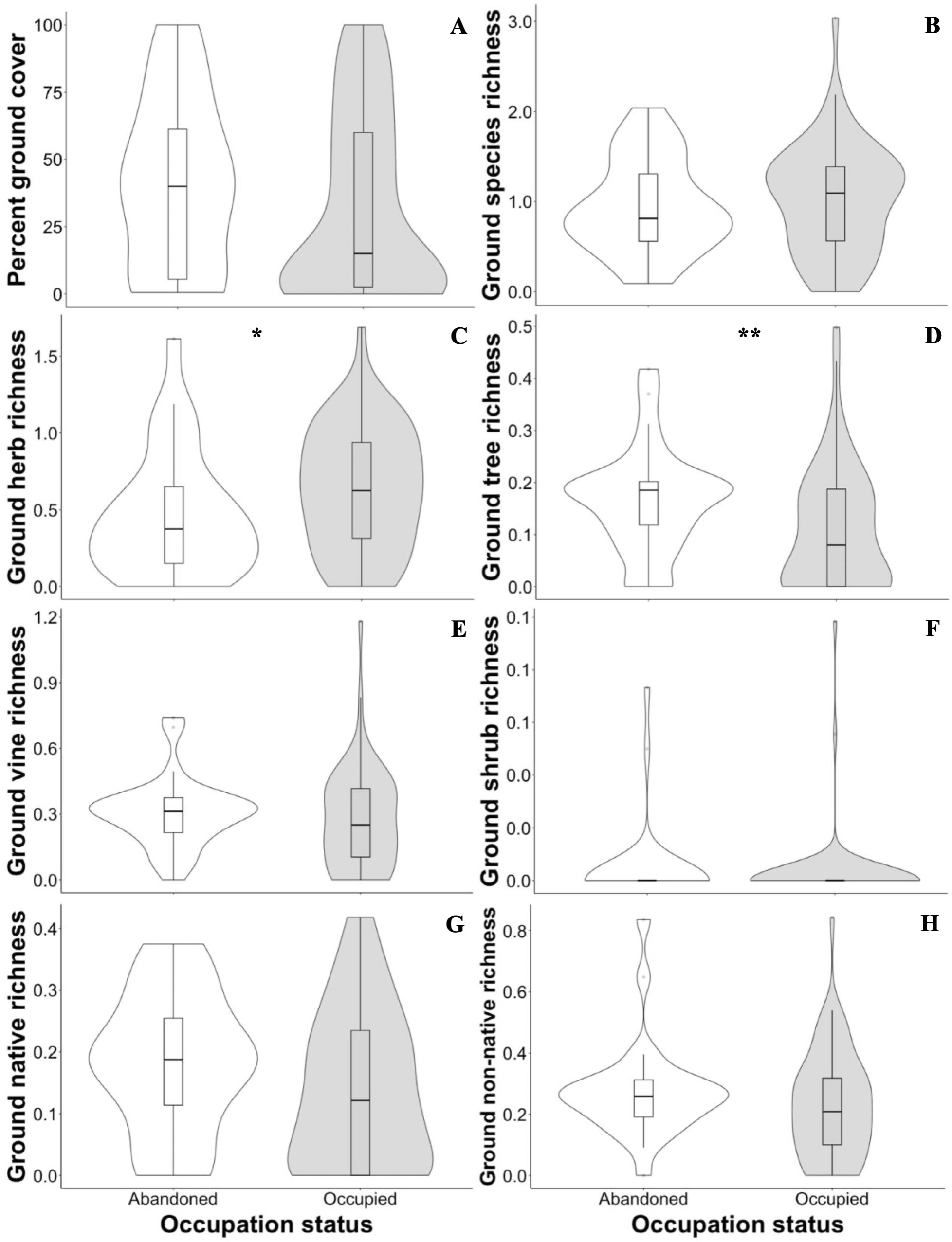

FIGURE 7
There was no significant difference in mean **(A)** vegetated ground cover or **(B)** total species richness per m^2 between abandoned and neighboring occupied properties. Separated by growth habit, **(C)** there was a marginally significant difference in mean herbaceous richness, with fewer species per m^2 on abandoned properties. **(D)** Conversely, there were significantly more tree species per m^2 on average on abandoned properties. There were no significant differences in mean **(E)** vine species per m^2 on properties of different occupation status. **(F)** Shrub communities were not analyzed separately due to their low frequency; only four ground surveys (two on abandoned properties, two on occupied properties, each on a different block) included a shrub. There were no significant differences by native status, with similar mean **(G)** native and **(H)** non-native species richness per m^2 on abandoned and neighboring occupied properties. Statistical significance ($p < 0.10$) is indicated by an asterisk; a double asterisk indicates significance of $p \leq 0.05$.

One potential explanation for the income-vegetation relationships we observed is the luxury hypothesis, which posits that increased wealth leads to more disposable income that residents can spend on planting/removing preferred/non-preferred vegetation, or that they can afford properties with already-desirable landscaping (Hope et al., 2003; Troy et al., 2007). Additionally, residents in higher income brackets are more likely to own their home while those in lower income brackets are more likely to rent; renters may be especially unwilling to invest time and money in maintaining land they do not own (Nielson and Smith, 2005; Jenerette et al., 2013; Berland et al., 2023). In many U.S. cities, including Baltimore and Washington, D.C., patterns of wealth and homeownership were shaped by discriminatory loan policies; the effects of this practice, commonly known as redlining, from the mid-1900s are still observable in present-day urban green space patterns (e.g., Grove et al., 2018; SChinasi et al., 2022; Yang et al., 2023). Abandoned properties typically have no one actively managing the vegetation, creating opportunities for successional trajectories favoring fast-growing vines and colonizing non-native species. This means lower-income households, besides having fewer resources for yard care, may also contend with higher source pressure from nearby non-native plant communities.

Many factors besides income—though often related to or mediated by wealth—may explain differences in residential green space, such as a household's culture, influence, power, or willingness to invest time and effort. For example, wealthier homeowners may have different expectations for their local landscape, better connections to local government officials, and/or more motivation to exert sway over decisions regarding unoccupied or public property in their neighborhood (Jordan et al., 2019; Biehler et al., 2025). Moreover, these individualistic variables exist within broader ecological, social, and political systems that provide context and limitations to urban ecosystems. These individual and broader factors have been incorporated into frameworks such as the human ecosystem framework (Force and Machlis, 1997; Machlis et al., 1997), social stratification hypothesis (Grove et al., 2006a), ecology of prestige (Grove et al., 2014), and POSE framework (Poulton Kamakura et al., 2024). Neighborhood age is also a noteworthy factor, as older neighborhoods have had more time for local vegetation to grow (e.g., Grove et al., 2006b; Boone et al., 2010; Clarke et al., 2013). Such legacy effects may be especially important for explaining the distribution of slow-to-mature species and older plants, such as the trees that create canopy cover, and less relevant for fast-growing species and young plants or seedlings often found in ground communities (Al-Kofahi et al., 2012).

One limitation of our study is that we were unable to collect data on these covariates to income, such as property tenure or yard maintenance activities. Future studies could integrate resident behavior and household characteristics to better disentangle social and ecological drivers. Another limitation is that we did not quantify the total area of cover per species. We suspect that doing so would have better emphasized the importance of vines in distinguishing between socioeconomically diverse plant communities. While income was not a significant predictor of vine richness in either canopy or ground surveys, the cover contribution of the vine species present was more extensive on lower-income properties (personal observation). Nor did we document the condition of the vegetation present. In canvassing Baltimore street trees, Shcheglovitova (2020) observed that trees were more often dead or dying in low-income neighborhoods than their high-income counterparts, and more often considered "stressed" by Baltimore's Department of Planning. It is possible that, in addition to unequal cover, richness, or composition, inequalities exist in plant health along an income gradient on private land, too.

Still, our study provides valuable, fine-scale information about how plant communities differ along income gradients and between properties of different occupation status. While socioeconomically diverse blocks may appear equally 'green' by satellite (Little et al., 2017), this does not reflect equivalent plant communities, in either structure, richness, or composition. Furthermore, our study highlights important differences in plant growth habit and native status between neighborhoods with different wealth, which has important implications for ecosystem services and resident experiences with urban green space (Biehler et al., 2025). This fine-scale knowledge may aid assessments of the services or disservices residents receive from the vegetation around their homes, and could better inform the maintenance of urban green spaces for the benefit of local residents. Strategies that prioritize native vegetation and support management in under-resourced neighborhoods may enhance ecosystem services and biodiversity equity.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

SR: Investigation, Visualization, Conceptualization, Formal analysis, Funding acquisition, Data curation, Writing – review & editing, Project administration, Methodology, Writing – original draft. SL: Formal analysis, Funding acquisition, Methodology, Writing – review & editing, Conceptualization. PL: Writing – review & editing, Funding acquisition, Conceptualization, Supervision, Resources, Methodology.

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Conflict of interest

The author(s) declared that this work was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Correction note

This article has been corrected with minor changes. These changes do not impact the scientific content of the article.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2026.1623853/full#supplementary-material

References

Akee, R., Jones, M. R., and Porter, S. R. (2019). Race matters: income shares, income inequality, and income mobility for all U.S. Races. *Demography* 56, 999–1021. doi: 10.1007/s13524-019-00773-7

Al-Kofahi, S., Steele, C., VanLeeuwen, D., and St Hilaire, R. (2012). Mapping land cover in urban residential landscapes using very high spatial resolution aerial photographs. *Urban Forestry Urban Greening* 11, 291–301. doi: 10.1016/j.ufug.2012.05.001

Anderson, E. C., Avolio, M. L., Sonti, N. F., and LaDeau, S. L. (2021). More than green: Tree structure and biodiversity patterns differ across canopy change regimes in Baltimore's urban forest. *Urban Forestry Urban Greening* 65, 127365. doi: 10.1016/j.ufug.2021.127365

Armson, D., Stringer, P., and Ennos, A. R. (2012). The effect of tree shade and grass on surface and globe temperatures in an urban area. *Urban Forestry Urban Greening* 11, 245–255. doi: 10.1016/j.ufug.2012.05.002

Avolio, M., Blanchette, A., Sonti, N. F., and Locke, D. H. (2020). Time is not money: Income is more important than lifestage for explaining patterns of residential yard plant community structure and diversity in Baltimore. *Front. Ecol. Evol.* 8. doi: 10.3389/fevo.2020.00085

Avolio, M. L., Pataki, D. E., Gillespie, T. W., Jenerette, G. D., McCarthy, H. R., Pincetl, S., et al. (2015). Tree diversity in southern California's urban forest: The interacting roles of social and environmental variables. *Front. Ecol. Evol.* 3. doi: 10.3389/fevo.2015.00073

Avolio, M. L., Pataki, D. E., Trammell, T. L. E., and Endter-Wada, J. (2018). Biodiverse cities: The nursery industry, homeowners, and neighborhood differences drive urban tree composition. *Ecol. Monogr.* 88, 259–276. doi: 10.1002/ecm.1290

Bates, D., Maechler, M., Bolker, B., and Walker, S. (2015). Fitting linear mixed-effects models using lme4. *J. Stat. Software* 67, 1–48. doi: 10.18637/jss.v067.i01

BDHCD (2024). *Housing code requirements* (Baltimore, MD: Baltimore City Department of Housing & Community Development). Available online at: https://dhcd.baltimorecity.gov/resources-renters/housing-code-requirements.

Berland, A., Locke, D. H., Herrmann, D. L., and Schwarz, K. (2020). Beauty or blight? Abundant vegetation in the presence of disinvestment across residential parcels and neighborhoods in toledo, OH. *Front. Ecol. Evol.* 8. doi: 10.3389/fevo.2020.566759

Berland, A., Locke, D. H., Herrmann, D. L., and Schwarz, K. (2023). Residential land owner type mediates the connections among vacancy, overgrown vegetation, and equity. *Urban Forestry Urban Greening* 80, 127826. doi: 10.1016/j.ufug.2022.127826

Berland, A., Shiflett, S. A., Shuster, W. D., Garmestani, A. S., Goddard, H. C., Herrmann, D. L., et al. (2017). The role of trees in urban stormwater management. *Landscape Urban Plann.* 162, 167–177. doi: 10.1016/j.landurbplan.2017.02.017

Berthon, K., Thomas, F., and Bekessy, S. (2021). The role of 'nativeness' in urban greening to support animal biodiversity. *Landscape Urban Plann.* 205, 103959. doi: 10.1016/j.landurbplan.2020.103959

Biehler, D., LaDeau, S., Baker, J., Bode-George, Y., Pitas, J. H., Jordan, R. C., et al. (2025). Segregation histories, wealth, and community engagement shape inequitable burdens of urban greening. *Ann. Am. Assoc. Geographers* 115, 513–534. doi: 10.1080/24694452.2024.2433011

Bigsby, K. M., McHale, M. R., and Hess, G. R. (2014). Urban morphology drives the homogenization of tree cover in Baltimore, MD, and Raleigh, NC. *Ecosystems* 17, 212–227. doi: 10.1007/s10021-013-9718-4

Blanchette, A., Trammell, T. L. E., Pataki, D. E., Endter-Wada, J., and Avolio, M. L. (2021). Plant biodiversity in residential yards is influenced by people's preferences for variety but limited by their income. *Landscape Urban Plann.* 214, 104149. doi: 10.1016/j.landurbplan.2021.104149

Blaug, M. (1972). The correlation between education and earnings: What does it signify? *Higher Educ.* 1, 53–76. doi: 10.1007/BF01956881

Boone, C., Buckley, G., Grove, M., and Sister, C. (2009). Parks and people: an environmental justice inquiry in baltimore, maryland. *Ann. Assoc. Am. Geographers - Ann. Assn Amer Geogr.* 99, 767–787. doi: 10.1080/00045600903102949

Boone, C. G., Cadenasso, M. L., Grove, J. M., Schwarz, K., and Buckley, G. L. (2010). Landscape, vegetation characteristics, and group identity in an urban and suburban watershed: why the 60s matter. *Urban Ecosyst.* 13, 255–271. doi: 10.1007/s11252-009-0118-7

Branas, C. C., Cheney, R. A., MacDonald, J. M., Tam, V. W., Jackson, T. D., and Ten Have, T. R. (2011). A difference-in-differences analysis of health, safety, and greening vacant urban space. *Am. J. Epidemiol.* 174, 1296–1306. doi: 10.1093/aje/kwr273

Cavender-Bares, J., Cubino, J. P., Pearse, W. D., Hobbie, S. E., Lange, A. J., Knapp, S., et al. (2020). Horticultural availability and homeowner preferences drive plant diversity and composition in urban yards. *Ecol. Appl.* 30, e02082. doi: 10.1002/eap.2082

Chawla, L. (2015). Benefits of nature contact for children. *J. Plann. Literature* 30, 433–452. doi: 10.1177/0885412215595441

CHPSA (n.d.). *Property standards authority* (Capitol Heights, MD: Capitol Heights Property Standards Authority). Available online at: https://www.capitolheightsmd.gov/2170/Property-Standards-Authority.

Clarke, L. W., Jenerette, G. D., and Davila, A. (2013). The luxury of vegetation and the legacy of tree biodiversity in Los Angeles, CA. *Landscape Urban Plann.* 116, 48–59. doi: 10.1016/j.landurbplan.2013.04.006

Cubino, J. P., Cavender-Bares, J., Hobbie, S. E., Pataki, D. E., Avolio, M. L., Darling, L. E., et al. (2019). Drivers of plant species richness and phylogenetic composition in urban yards at the continental scale. *Landscape Ecol.* 34, 63–77. doi: 10.1007/s10980-018-0744-7

DCDOB (2023). *Report Excessive Vegetation and/or Excessive Trash on Properties* (District of Columbia: DC Department of Buildings). Available online at: https://dob.dc.gov/node/1620791.

Donovan, G. H., and Prestemon, J. P. (2012). The effect of trees on crime in portland, oregon. *Environ. Behav.* 44, 3–30. doi: 10.1177/0013916510383238

EdScape (2019). *Total Population Trends* (District of Columbia: Office of the Deputy Mayor for Education). Available online at: https://edscape.dc.gov/page/pop-and-students-total-population-trends.

Endsley, K. A., Brown, D. G., and Bruch, E. (2018). Housing market activity is associated with disparities in urban and metropolitan vegetation. *Ecosystems* 21, 1593–1607. doi: 10.1007/s10021-018-0242-4

EPA (2024). *Ecoregions of North America* (Environmental Protection Agency). Available online at: https://www.epa.gov/eco-research/ecoregions-north-america.

Force, J. E., and Machlis, G. E. (1997). The human ecosystem Part II: Social indicators in ecosystem management. *Soc. Natural Resour.* 10, 369–382. doi: 10.1080/08941929709381035

Grove, J. M., Cadenasso, M. L., Burch, W. R.Jr., Pickett, S. T. A., Schwarz, K., O'Neil-Dunne, J., et al. (2006a). Data and methods comparing social structure and vegetation structure of urban neighborhoods in baltimore, maryland. *Soc. Natural Resour.* 19, 117–136. doi: 10.1080/08941920500394501

Grove, J. M., Locke, D. H., and O'Neil-Dunne, J. P. M. (2014). An ecology of prestige in New York City: Examining the relationships among population density, socio-economic status, group identity, and residential canopy cover. *Environ. Manage.* 54, 402–419. doi: 10.1007/s00267-014-0310-2

Grove, M., Ogden, L., Pickett, S., Boone, C., Buckley, G., Locke, D. H., et al. (2018). The legacy effect: understanding how segregation and environmental injustice unfold over time in baltimore. *Ann. Am. Assoc. Geographers* 108, 524–537. doi: 10.1080/24694452.2017.1365585

Grove, J. M., Troy, A. R., O'Neil-Dunne, J. P. M., Burch, W. R., Cadenasso, M. L., and Pickett, S. T. A. (2006b). Characterization of households and its implications for the vegetation of urban ecosystems. *Ecosystems* 9, 578–597. doi: 10.1007/s10021-006-0116-z

Gulachenski, A., Ghersi, B. M., Lesen, A. E., and Blum, M. J. (2016). Abandonment, ecological assembly and public health risks in counter-urbanizing cities. *Sustainability* 8, 491. doi: 10.3390/su8050491

Gunawardena, K. R., Wells, M. J., and Kershaw, T. (2017). Utilising green and bluespace to mitigate urban heat island intensity. *Sci. Total Environ.* 584-585, 1040–1055. doi: 10.1016/j.scitotenv.2017.01.158

Heo, S., and Bell, M. L. (2023). Investigation on urban greenspace in relation to sociodemographic factors and health inequity based on different greenspace metrics in 3 US urban communities. *J. Exposure Sci. Environ. Epidemiol.* 33, 218–228. doi: 10.1038/s41370-022-00468-z

Hope, D., Gries, C., Zhu, W., Fagan, W. F., Redman, C. L., Grimm, N. B., et al. (2003). Socioeconomics drive urban plant diversity. *Proc. Natl. Acad. Sciences U.S.A.* 100, 8788–8792. doi: 10.1073/pnas.1537557100

Hummel, D. (2020). The effects of population and housing density in urban areas on income in the United States. *Local Economy* 35, 27–47. doi: 10.1177/0269094220903265

Jenerette, G., Miller, G., Buyantuev, A., Pataki, D., Gillespie, T., and Pincetl, S. (2013). Urban vegetation and income segregation in drylands: A synthesis of seven metropolitan regions in the southwestern United States. *Environ. Res. Lett.* 8, 044001. doi: 10.1088/1748-9326/8/4/044001

Jiang, S., Sonti, N. F., and Avolio, M. L. (2022). Tree communities in Baltimore differ by land use type, but change little over time. *Ecosphere* 13, e4054. doi: 10.1002/ecs2.4054

Jordan, R. C., Sorensen, A. E., Biehler, D., Wilson, S., and LaDeau, S. (2019). Citizen science and civic ecology: merging paths to stewardship. *J. Environ. Stud. Sci.* 9, 133–143. doi: 10.1007/s13412-018-0521-6

Kendal, D., Williams, K. J. H., and Williams, N. S. G. (2012). Plant traits link people's plant preferences to the composition of their gardens. *Landscape Urban Plann.* 105, 34–42. doi: 10.1016/j.landurbplan.2011.11.023

Ko, Y. (2018). Trees and vegetation for residential energy conservation: A critical review for evidence-based urban greening in North America. *Urban Forestry Urban Greening* 34, 318–335. doi: 10.1016/j.ufug.2018.07.021

Kremer, P., Hamstead, Z. A., and McPhearson, T. (2013). A social-ecological assessment of vacant lots in New York City. *Landscape Urban Plann.* 120, 218–233. doi: 10.1016/j.landurbplan.2013.05.003

Kuznetsova, A., Brockhoff, P., and Christensen, R. (2017). lmerTest package: tests in linear mixed effects models. *J. Stat. Software* 82, 1–26. doi: 10.18637/jss.v082.i13

LaDeau, S. L., Leisnham, P. T., Biehler, D., and Bodner, D. (2013). Higher mosquito production in low-income neighborhoods of Baltimore and Washington, DC: Understanding ecological drivers and mosquito-borne disease risk in temperate cities. *Int. J. Environ. Res. Public Health* 10, 1505–1526. doi: 10.3390/ijerph10041505

Leisnham, P. T., LaDeau, S. L., Saunders, M. E. M., and Villena, O. C. (2021). Condition-specific competitive effects of the invasive mosquito *Aedes albopictus* on the resident *Culex pipiens* among different urban container habitats may explain their coexistence in the field. *Insects* 12, 993. doi: 10.3390/insects12110993

Leong, M., Dunn, R. R., and Trautwein, M. D. (2018). Biodiversity and socioeconomics in the city: a review of the luxury effect. *Biol. Lett.* 14, 20180082. doi: 10.1098/rsbl.2018.0082

Lewis, J. A., Zipperer, W. C., Ernstson, H., Bernik, B., Hazen, R., Elmqvist, T., et al. (2017). Socioecological disparities in New Orleans following Hurricane Katrina. *Ecosphere* 8, e01922. doi: 10.1002/ecs2.1922

Lin, J., Wang, Q., and Huang, B. (2021). Street trees and crime: What characteristics of trees and streetscapes matter. *Urban Forestry Urban Greening* 65, 127366. doi: 10.1016/j.ufug.2021.127366

Little, E., Biehler, D., Leisnham, P. T., Jordan, R., Wilson, S., and LaDeau, S. L. (2017). Socio-ecological mechanisms supporting high densities of *Aedes albopictus* (Diptera: Culicidae) in Baltimore, MD. *J. Med. Entomology* 54, 1183–1192. doi: 10.1093/jme/tjx103

Lowenstein, D. M., and Minor, E. S. (2016). Diversity in flowering plants and their characteristics: integrating humans as a driver of urban floral resources. *Urban Ecosyst.* 19, 1735–1748. doi: 10.1007/s11252-016-0563-z

Machlis, G. E., Force, J. E., and Burch, W. R.Jr (1997). The human ecosystem Part I: The human ecosystem as an organizing concept in ecosystem management. *Soc. Natural Resour.* 10, 347–367. doi: 10.1080/08941929709381034

Menendian, S., Gambhir, S., and Hsu, C.-W. (2022). *Single-family zoning in Greater Los Angeles* (Berkeley, CA: Other & Belonging Institute at UC Berkeley). Available online at: https://belonging.berkeley.edu/single-family-zoning-greater-los-angeles.

Nassauer, J. I., Webster, N. J., Sampson, N., and Li, J. (2021). Care and safety in neighborhood preferences for vacant lot greenspace in legacy cities. *Landscape Urban Plann.* 214, 104156. doi: 10.1016/j.landurbplan.2021.104156

Nielson, L., and Smith, C. L. (2005). Influences on residential yard care and water quality: Tualatin Watershed, Oregon. *J. Am. Water Resour. Assoc.* 41, 93–106. doi: 10.1111/j.1752-1688.2005.tb03720.x

Nowak, D. J., Crane, D. E., and Stevens, J. C. (2006). Air pollution removal by urban trees and shrubs in the United States. *Urban forestry urban greening* 4, 115–123. doi: 10.1016/j.ufug.2006.01.007

Nowak, D. J., Hirabayashi, S., Doyle, M., McGovern, M., and Pasher, J. (2018). Air pollution removal by urban forests in Canada and its effect on air quality and human health. *Urban Forestry Urban Greening* 29, 40–48. doi: 10.1016/j.ufug.2017.10.019

NYCPlanning (2023). *Residence districts* (New York, NY: New York City Department of City Planning). Available online at: https://www.nyc.gov/site/planning/zoning/districts-tools/residence-districts-r1-r10.page:~:text=Residence%20districts%20are%20the%20most,the%20city%27s%20zoned%20land%20area.

Oksanen, J., Simpson, G., Blanchet, F., Kindt, R., Legendre, P., Minchin, P., et al. (2025). vegan: Community Ecology Package. *R package version 2.8-0*. https://vegandevs.github.io/vegan/

Pearsall, H., and Christman, Z. (2012). Tree-lined lanes or vacant lots? Evaluating non-stationarity between urban greenness and socio-economic conditions in Philadelphia, Pennsylvania, USA at multiple scales. *Appl. Geogr.* 35, 257–264. doi: 10.1016/j.apgeog.2012.07.006

Ponte, S., Sonti, N. F., Phillips, T. H., and Pavao-Zuckerman, M. A. (2021). Transpiration rates of red maple (*Acer rubrum* L.) differ between management contexts in urban forests of Maryland, USA. *Sci. Rep.* 11, 22538. doi: 10.1038/s41598-021-01804-3

Poulton Kamakura, R., Bai, J., Sheel, V., and Katti, M. (2024). Biodiversity is not a luxury: Unpacking wealth and power to accommodate the complexity of urban biodiversity. *Ecosphere* 15, e70049. doi: 10.1002/ecs2.70049

R Core Team (2021). *R: A language and environment for statistical computing* (Vienna, Austria: R Foundation for Statistical Computing).

Riley, C. B., and Gardiner, M. M. (2020). Examining the distributional equity of urban tree canopy cover and ecosystem services across United States cities. *PloS One* 15, e0228499. doi: 10.1371/journal.pone.0228499

Riley, C. B., Perry, K. I., Ard, K., and Gardiner, M. M. (2018). Asset or liability? Ecological and sociological tradeoffs of urban spontaneous vegetation on vacant land in shrinking cities. *Sustainability* 10, 2139. doi: 10.3390/su10072139

Rothman, S. E. (2024). Mosquitoes and Vegetation Across Socioeconomic Gradients. College Park, MD: University of Maryland, College Park. Ph.D. dissertation.

Rothman, S. E., LaDeau, S. L., and Leisnham, P. T. (2026). Urban socioeconomic, vegetation, and mosquito interactions in the USA: Current research and future directions. *Ecosphere*.

Saunders, K. (2020). Improving the Surveillance and Control of Vector Mosquitoes in Heterogeneous Landscapes. College Park, MD: University of Maryland. M.S. thesis.

SChinasi, L. H., Kanungo, C., Christman, Z., Barber, S., Tabb, L., and Headen, I. (2022). Associations between historical redlining and present-day heat vulnerability housing and land cover characteristics in Philadelphia, PA. *J. Urban Health-Bulletin New York Acad. Med.* 99, 134–145. doi: 10.1007/s11524-021-00602-6

Selbig, W. R., Loheide, S. P., Shuster, W., Scharenbroch, B. C., Coville, R. C., Kruegler, J., et al. (2022). Quantifying the stormwater runoff volume reduction benefits of urban street tree canopy. *Sci. Total Environ.* 806, 151296. doi: 10.1016/j.scitotenv.2021.151296

Shcheglovitova, M. (2020). Valuing plants in devalued spaces: Caring for Baltimore's Street trees. *Environ. Plann. E: Nat. Space* 3, 228–245. doi: 10.1177/2514848619854375

Soga, M., Evans, M. J., Tsuchiya, K., and Fukano, Y. (2021). A room with a green view: the importance of nearby nature for mental health during the COVID-19 pandemic. *Ecol. Appl.* 31, e2248. doi: 10.1002/eap.2248

Song, S., Tu, R., Lu, Y., Yin, S., Lin, H., and Xiao, Y. (2022). Restorative effects from green exposure: A systematic review and meta-analysis of randomized control trials. *Int. J. Environ. Res. Public Health* 19, 14506. doi: 10.3390/ijerph192114506

South, E. C., Hohl, B. C., Kondo, M. C., MacDonald, J. M., and Branas, C. C. (2018). Effect of greening vacant land on mental health of community-dwelling adults: A cluster randomized trial. *JAMA Network Open* 1, e180298–e180298. doi: 10.1001/jamanetworkopen.2018.0298

South, E. C., Kondo, M. C., Cheney, R. A., and Branas, C. C. (2015). Neighborhood blight, stress, and health: A walking trial of urban greening and ambulatory heart rate. *Am. J. Public Health* 105, 909–913. doi: 10.2105/ajph.2014.302526

Spotswood, E. N., Benjamin, M., Stoneburner, L., Wheeler, M. M., Beller, E. E., Balk, D., et al. (2021). Nature inequity and higher COVID-19 case rates in less-green neighbourhoods in the United States. *Nat. Sustainability* 4, 1092–1098. doi: 10.1038/s41893-021-00781-9

Tan, Z., Lau, K. K.-L., and Ng, E. (2016). Urban tree design approaches for mitigating daytime urban heat island effects in a high-density urban environment. *Energy Buildings* 114, 265–274. doi: 10.1016/j.enbuild.2015.06.031

Troy, A. R., Grove, J. M., O'Neil-Dunne, J. P. M., Pickett, S. T. A., and Cadenasso, M. L. (2007). Predicting opportunities for greening and patterns of vegetation on private urban lands. *Environ. Manage.* 40, 394–412. doi: 10.1007/s00267-006-0112-2

Troy, A., Grove, J. M., and O'Neil-Dunne, J. (2012). The relationship between tree canopy and crime rates across an urban–rural gradient in the greater Baltimore region. *Landscape urban Plann.* 106, 262–270. doi: 10.1016/j.landurbplan.2012.03.010

Twohig-Bennett, C., and Jones, A. (2018). The health benefits of the great outdoors: A systematic review and meta-analysis of greenspace exposure and health outcomes. *Environ. Res.* 166, 628–637. doi: 10.1016/j.envres.2018.06.030

U.S. Census Bureau (2020a). *Median household income in the past 12 months (in 2018 inflation-adjusted dollars)*. Available online at: https://data.census.gov/cedsci/table?q=b19013&tid=ACSDT1Y2018.B19013&hidePreview=true (Accessed January 7, 2022).

U.S. Census Bureau (2020b). *Median household income: 2015–2019*. Available online at: https://www.census.gov/library/visualizations/interactive/acs-median-household-income-2015-2019.html (Accessed January 7, 2022).

USDA, NRCS. (2022). "PLANTS Help," in *NRCS* (National Plant Data Team, Greensboro, NC USA). Available online at: https://plants.sc.egov.usda.gov/DocumentLibrary/Pdf/PLANTS_Help_Document_2022.pdf.

Wang, Y., and Akbari, H. (2016). The effects of street tree planting on urban heat island mitigation in Montreal. *Sustain. Cities Soc.* 27, 122–128. doi: 10.1016/j.scs.2016.04.013

Wheeler, M. M., Larson, K. L., Bergman, D., and Hall, S. J. (2022). Environmental attitudes predict native plant abundance in residential yards. *Landscape Urban Plann.* 224, 104443. doi: 10.1016/j.landurbplan.2022.104443

Wheeler, M. M., Neill, C., Groffman, P. M., Avolio, M., Bettez, N., Cavender-Bares, J., et al. (2017). Continental-scale homogenization of residential lawn plant communities. *Landscape Urban Plann.* 165, 54–63. doi: 10.1016/j.landurbplan.2017.05.004

Wood, E., Harsant, A., Dallimer, M., and Hassall, C. (2018). Not all green space is created equal: Biodiversity predicts psychological restorative benefits from urban green space. *Front. Psychol.* 9, 391823. doi: 10.3389/fpsyg.2018.02320

Woods, A. J., Omernik, J. M., and Brown, D. D. (1999). *Level III and IV Ecoregions of Delaware, Maryland, Pennsylvania, Virginia, and West Virginia* (Corvallis, OR: Environmental Protection Agency).

Yang, Y., Cho, A., Nguyen, Q., and Nsoesie, E. O. (2023). Association of neighborhood racial and ethnic composition and historical redlining with built environment indicators derived from street view images in the US. *JAMA Network Open* 6, e2251201–e2251201. doi: 10.1001/jamanetworkopen.2022.51201

Yang, J., McBride, J., Zhou, J., and Sun, Z. (2005). The urban forest in Beijing and its role in air pollution reduction. *Urban Forestry Urban Greening* 3, 65–78. doi: 10.1016/j.ufug.2004.09.001

Zhou, W., Troy, A., Morgan Grove, J., and Jenkins, J. C. (2009). Can money buy green? Demographic and socioeconomic predictors of lawn-care expenditures and lawn greenness in urban residential areas. *Soc. Natural Resour.* 22, 744–760. doi: 10.1080/08941920802074330