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REPORT

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Current street tree communities reflect race-based housing policy and modern attempts to remedy environmental injustice

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Abstract

Humans promote and inhibit other species on the urban landscape, shaping biodiversity patterns. Institutional racism may underlie the distribution of urban species by creating disproportionate resources in space and time. Here, we examine whether present-day street tree occupancy, diversity, and composition in Baltimore, MD, USA, neighborhoods reflect their 1937 classification into grades of loan risk—from most desirable (A = green) to least desirable (D = "redlined")—using racially discriminatory criteria. We find that neighborhoods that were redlined have consistently lower street tree α-diversity and are nine times less likely to have large (old) trees occupying a viable planting site. Simultaneously, redlined neighborhoods were locations of recent tree planting activities, with a high occupancy rate of small (young) trees. However, the community composition of these young trees exhibited lower species turnover and reordering across neighborhoods compared to those in higher grades, due to heavy reliance on a single tree species. Overall, while the negative effects of redlining remain detectable in present-day street tree communities, there are clear signs of recent investment. A strategy of planting diverse tree cohorts paired with investments in site rehabilitation and maintenance may be necessary if cities wish to overcome ecological feedbacks associated with legacies of environmental injustice.

KEYWORDS
community composition, ecosystem services, environmental justice, housing policy, street trees, tree canopy, urban biodiversity

INTRODUCTION

A major goal of community ecology is to understand the mechanisms that explain species abundance and distribution. In built environments, many of the classic determinants of community assembly such as the regional species pool, dispersal, and abiotic and biotic filters are strongly influenced by human behaviors and decision-making (Aronson et al., 2016; Avolio et al., 2021; Swan et al., 2017). For instance, patterns of urban biodiversity are driven by social factors, although mechanisms that explain relationships such as those between biodiversity

and income (Kinzig et al., 2005; Chamberlain et al., 2020) and residents' education (Avolio et al., 2015) are complex. One relatively unexplored social factor that may drive patterns of urban species diversity are consequences of institutionalized racism and segregation (Schell et al., 2020; Watkins & Gerrish, 2018), which produce disproportionate distributions of resources in space and time.

Understanding the ecology of cities requires looking through the spatial and temporal lenses of institutionalized racism and its roles in policy, planning, and management (Pickett & Grove, 2020; Schell et al., 2020; Grove et al., 2018). These social–ecological patterns reflect long-term systems and landscapes of racial segregation, exclusion, and inclusion, prejudice and privilege (Goetz et al., 2020). In the USA, from the 1900s to mid-1930s, Blacks, Jews, and Catholics were often excluded, through a variety of mechanisms, from lower density, more amenity-rich neighborhoods with parks and tree-lined streets (Rothstein, 2017). In the mid-1930s, many of these local activities were codified and made spatially explicit at a national level although the Homeowners' Loan Corporation (HOLC), which was established to support federal reinvestment in housing after the Great Depression (Rothstein, 2017). The HOLC mapped and classified 239 cities by loan risk—from most desirable to least desirable for investment—with corresponding colors from green to red. Neighborhoods that were "redlined" had characteristics of high-density, poor housing stock, proximity to undesirable land uses, and large percentages of Black, Catholic, Jewish, or immigrant populations.

While institutionalized racism predated the HOLC's redlining maps and has persisted in other ways, the spatially explicit grading of urban neighborhoods added to and reinforced residential segregation across the USA. Redlining has become associated with lower homeownership rates (Aaronson et al., 2021); shorter life expectancies and poorer overall health (Gee, 2002; Nardone et al., 2020); and the allocation of other disamenities and amenities in cities throughout the USA. Urban ecologists have also documented correlations among HOLC classification and key biotic and abiotic attributes. For example, formerly A-graded neighborhoods are significantly cooler than D-graded neighborhoods (Hoffman et al., 2020; Wilson, 2020) with nearly twice the total vegetation (Namin et al., 2020) and tree canopy (Locke et al., 2021), and higher net ecosystem services (Nowak et al., 2022).

Here, we use the HOLC map for the City of Baltimore to examine the relationship between institutional racism and current urban street tree biodiversity and community composition. Street trees are an ideal model system for exploring these questions because their persistence can reflect current and historic human decisions and require substantial investment in planting and sustained maintenance. Further, as sessile, long-lived organisms, the diversity and composition of different size classes of urban trees reflect past (larger trees) and current (small trees) societal investment. Specifically, we ask whether neighborhoods that were redlined contain different present-day occupancy rates of tree size classes, patterns of α-diversity, or patterns of community composition such as species turnover or reordering. We predict that street trees in these neighborhoods (i.e., D-graded) will exhibit: (1) lower occupancy of large (i.e., older) trees, (2) lower α-diversity, (3) different tree species community composition compared with better-rated HOLC neighborhoods, and (4) lower among-neighborhood species turnover (i.e., homogenization). However, these predictions may not hold in smaller (i.e., younger) cohorts of street trees that have been established more recently. Here, we predict that older assemblages, indicated by trees with larger diameters, should exhibit significant divergence in assemblage structure by HOLC grade, but such differences should be less distinct with newer assemblages (smaller trees) due to similar contemporary planting practices implemented citywide.

METHODS

Site

Baltimore, MD (39.2904° N, 76.6122° W) is a city of 585,708 people built on 209.6 km^2 in a temperate forest ecosystem typical of the United States mid-Atlantic region (US Census Bureau, 2022).

Data

We obtained polygons representing the spatial extent of the HOLC neighborhoods coded by their grade classifications (A, B, C, or D) from the Mapping Inequality Project (Nelson et al., 2019). To align with the city tree dataset, we included only the portion of HOLC-defined neighborhoods entirely within present-day Baltimore City. Approximately 42% of Baltimore's area today was classified by HOLC in the 1930s, with ∼4.38% as grade A, ∼19.9% as grade B, ∼12.7% as grade C, and ∼9.24% as grade D. We obtained the street tree dataset from Baltimore City's Department of Recreation & Parks (Randolph, personal communication); it is also publicly available to download (Baltimore City Department of Recreation & Parks, 2021). Data were collected by an independent contractor for the City of Baltimore during 2017–2018, and include all live and dead street trees, and opportunities for new street trees, such as empty pits. We consider all the places where a street tree could be planted (i.e., whether one is or not) to be the total

viable street tree sites. For each location we extracted: latitude/longitude, tree species, tree diameter at 1.37 m [diameter in inches at breast height (dbh)], and tree condition (living, dead, stump, vacant site). dbh is the best available proxy for tree age, but we recognize the limitations of this metric including species or site-based differences in growth rate and maximum size (Magarik et al., 2020; Troxel et al., 2013).

To ensure that we compared occupancy and diversity accurately across the different numbers and sizes of neighborhoods in each HOLC grade, we tailored the analysis for each question. Briefly, when comparing the proportion of viable planting sites occupied by living trees within neighborhoods nested within a HOLC grade, we used an equal-area subsampling approach. We divided each neighborhood into equal-sized 3.3 hectare hexagonal polygons ("hexes") (Figure 1a), where each hex then served as a tree community subsample of equal area within the completely censused street tree population of each neighborhood. We selected this size and shape based on boundaries; it was the largest area polygon that enabled replication within all neighborhoods. When comparing overall street tree species α-diversity across HOLC grade levels, we used individual-based accumulation curves rarefied to a common number of trees or sample coverage. Last, to determine whether overall species composition and β-diversity metrics differed across HOLC grades, we used "neighborhood" as the sampling unit aggregating all trees located within the neighborhood bounds. To account for differing street tree abundances across neighborhoods due to street density (Appendix S1: Figure S3), we compared an abundance-based (Bray–Curtis) and relative abundance-based (Morisita–Horn) dissimilarity metric; this made no qualitative difference and only Bray–Curtis results are reported.

Trees were spatially joined to hexes and hexes were spatially joined to their containing neighborhood polygon. Each street tree has a unique hex code and HOLC-defined neighborhood grade. Unidentified living trees (19) were included in the occupancy analysis and excluded from the diversity/composition analysis. Within Baltimore City limits, 36 neighborhoods were given HOLC grades in the 1930s and today contain street trees (n = A: 4, B: 12, C: 13, D: 7). Equal-area hexes drawn within the boundaries of these neighborhoods contain 90,051 viable street tree locations. Of these, 49,458 contain living trees identifiable as 228 species/hybrids or, in seven cases, genera when species designation is very difficult (e.g., *Prunus* spp.).

Statistical analysis

All analyses were done in R (R Development Core Team, 2020). All code is at https://doi.org/10.5281/zenodo.7117811.

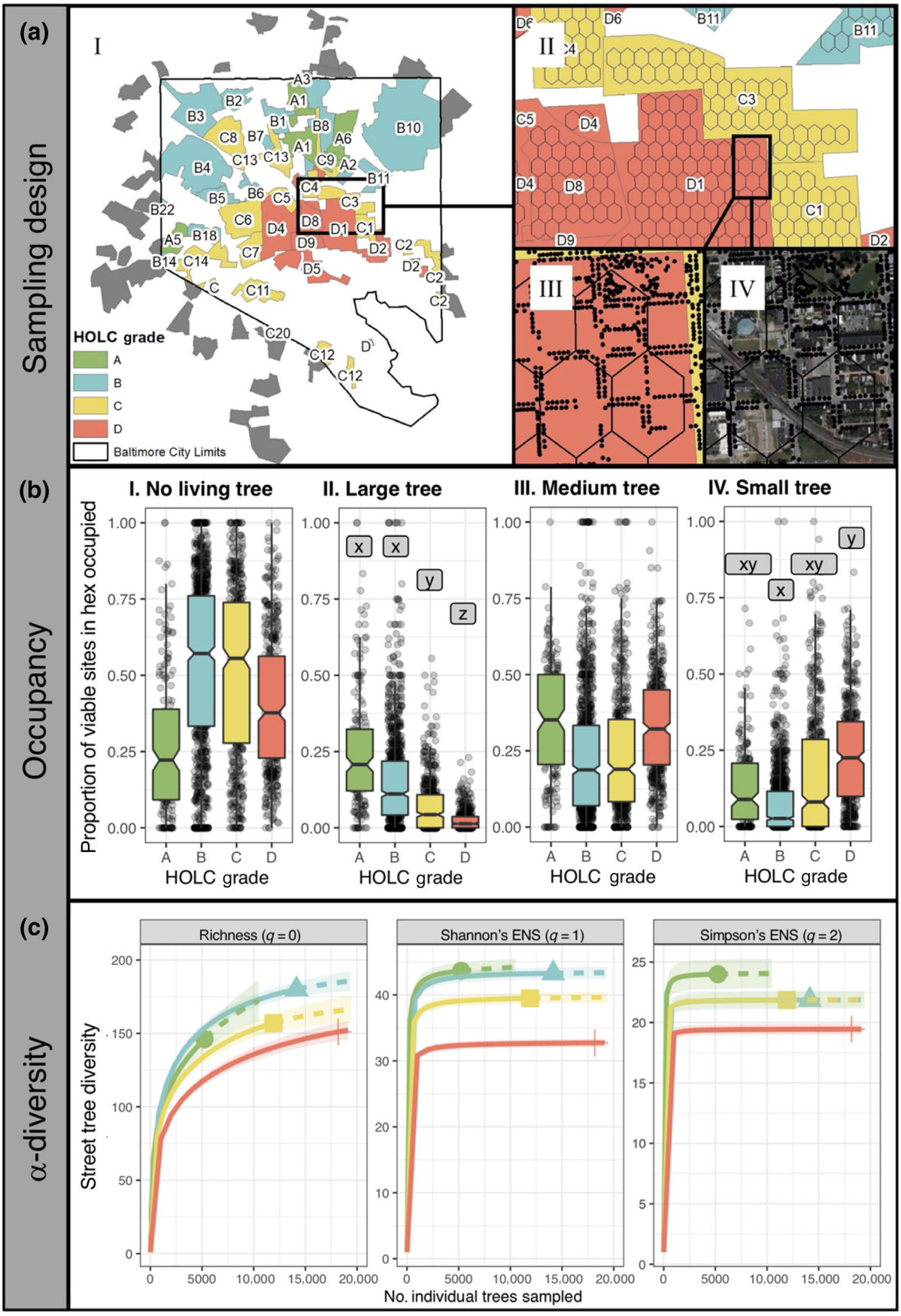
For each diversity analysis we first analyzed all street trees in the dataset. Next, to study how planting practices and communities may have changed over time, we divided the trees into size classes assuming that smaller trees of dbh ≤ 5 inches (12.7 cm) represent recent plantings, while larger (presumably older) trees of dbh ≥ 20 inches (50.8 cm) are likely to reflect planting and management practices from many decades ago (see Figure 2a for tree size distributions; Appendix S1 for methodological details).

Tree occupancy

We used an occupancy-based analysis to standardize for the uneven distribution of viable street tree locations across HOLC grades (Appendix S1: Figure S3). We used each hex as an equal-area subsample of a neighborhood and built four separate models to determine whether the HOLC grade designation predicted whether each viable location for a tree within that hex was differentially likely to contain (1) a living small tree, (2) a living medium tree, (3) a living large tree, or be (4) currently empty or occupied by a dead tree. We used the glmer() function in lme4 to build logistic regression models with a binomial distribution with HOLC grade as a fixed effect and each neighborhood nested within a HOLC grade as a random intercept to account for the spatial nonindependence of hexes within a neighborhood (Bates et al., 2012). The "weights" command weighted each hex based on the total number of viable tree sites it contained. Drop1() performed a chi-squared likelihood ratio test (LRT) comparing an intercept only model to one containing HOLC grade as a fixed effect. If HOLC grade was a significant predictor, we used emmeans() to perform posthoc tests comparing the HOLC grades using Tukey adjustments for multiple comparisons (Russell, 2021).

Tree α-diversity

Using only live, identified species, we created individual-based species accumulation curves with the iNext package (Chao et al., 2014) for each HOLC grade for richness ($q = 0$), Shannon's effective number of species (ENS; $q = 1$), and Simpson's ENS ($q = 2$). We chose these metrics to represent a gradient across the two aspects of α-diversity, species number, and relative abundance. To calculate effect sizes for diversity across HOLC grades, we used interpolation with estimateD() for equivalent numbers of trees or sample coverage.



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Tree composition and β-diversity

We performed nonmetric multidimensional scaling (NMDS) using the metaMDS() function in the vegan R package (Oksanen et al., 2007) to visualize differences in the species composition of the 33 neighborhoods containing a sufficient number of living trees using Bray–Curtis distance (n = A: 4, B: 9, C: 13, D: 7). Next, we ran a PERMANOVA using adonis() in the vegan package to assess whether HOLC grades differed in species composition by testing for differences in the centroid means of the neighborhoods in each grade. Finally, as a measure of β-diversity, we examined how tree community composition varies among neighborhoods within a HOLC grade. To do this, we calculated pairwise metrics based on the rank abundance curves of each neighborhood to examine species reordering and species differences or turnover (Avolio et al., 2019) using RAC_difference() in the codyn R package (Hallett et al., 2016). To test for differences among HOLC grades, we ran a one-way ANOVA with HOLC grade as a fixed effect. We corrected for multiple hypothesis testing using a Benjamini–Hochberg correction with p.adjust(). Last, to visualize which species might be driving compositional differences, we made rank abundance curves for the small and large tree cohorts.

RESULTS

Street tree site occupancy

HOLC grades differed in the proportion of viable sites currently occupied by large trees (Figure 1b.II, LRT = 38.2, $p < 0.001$); with a decreasing probability from A- and B-, to C-, to D-graded neighborhoods. Specifically, a viable tree location in an A neighborhood is almost nine times more likely to have a living large tree present than a viable site in a D neighborhood (log odds ratio A:D = 8.76 ± 3.176 95% CI, $p = <0.0001$). HOLC grade also predicts the chance that a small tree is currently occupying that site (Figure 1b.IV; LRT = 8.57, $p = 0.036$). We see the opposite pattern to large trees with the greatest chance of occupancy by small trees in D-graded neighborhoods. However, the only pairwise difference detected is a 60% decreased chance of occupancy of a small tree in B compared with D neighborhoods (odds ratio B:D = 0.40 ± 0.13 95% CI, $z = -2.87$, $p = <0.02$). Sites with no living tree (LRT = 1.35, $p = 0.72$), or medium-sized trees (LRT = 4.36, $p = 0.23$) did not differ by HOLC grade.

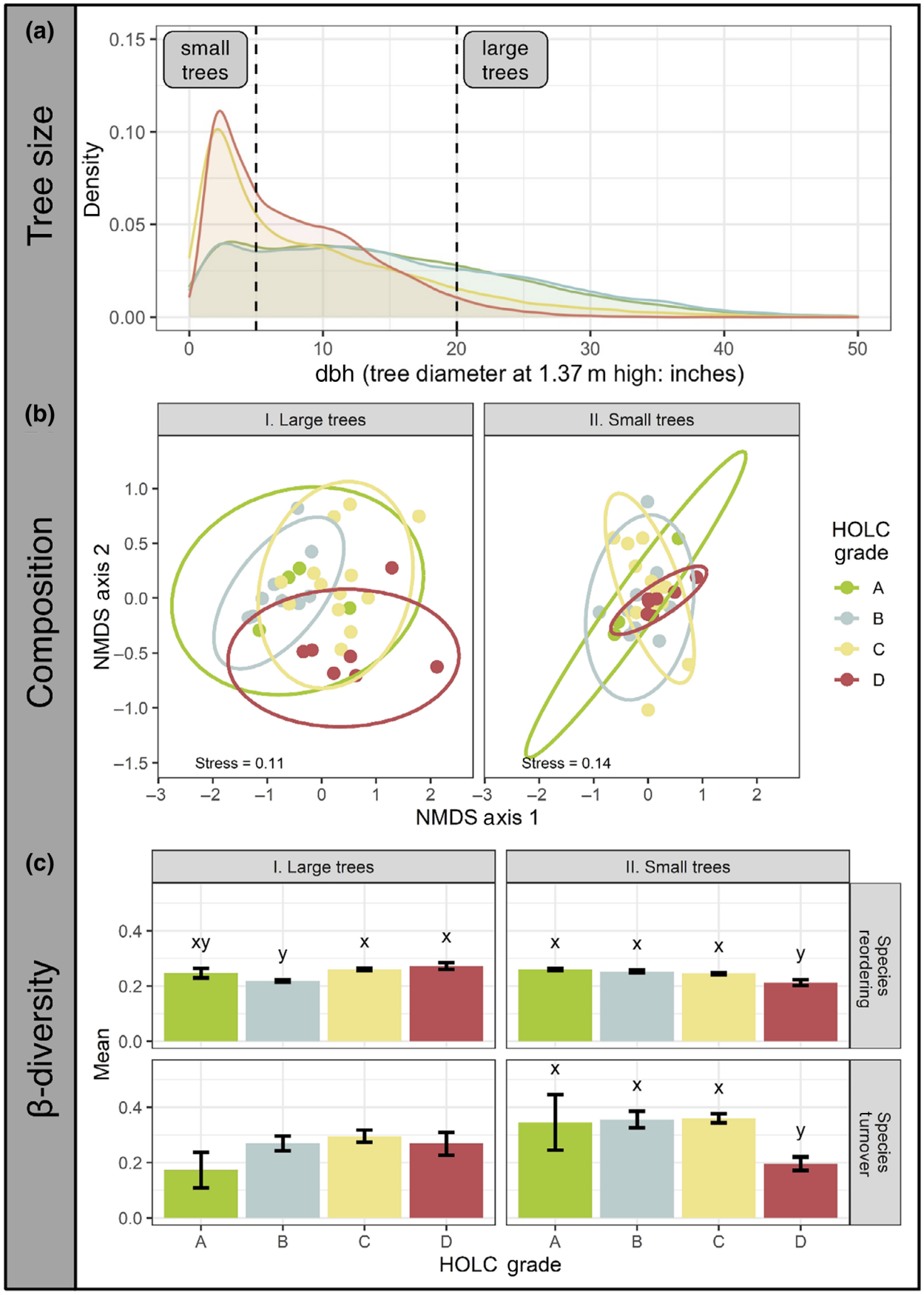
Tree α-diversity

Individual-based accumulation curves revealed that, for the same number of living street trees, diversity was higher in areas categorized by HOLC as grades A and B, and lowest in grade D (Figure 1c). This was consistent for richness (0), Shannon's effective number of species (1), and Simpson's effective number of species (2). A-graded areas had 23% higher species richness, 35% higher Shannon's effective number of species, and 24% higher Simpson's effective number of species than D-graded areas standardized at the minimum observed HOLC grade sample size of $n = 5238$ trees. Similar results were obtained when comparing at the minimum observed 99.3% sampling coverage; A-rated areas had 33%, 36%, and 24% higher diversity than D, respectively, across diversity measures (Appendix S1: Figure S2). The metrics that incorporate relative abundance (1) and (2) showed that D neighborhoods have a less diverse pool of equally abundant species (Figure 1c).

Tree composition and β-diversity

Small and large tree community composition differed depending on HOLC grade (Figure 2b; small trees: $F_{3,29} = 1.792$, adj $p = 0.013$; large trees: $F_{3,29} = 2.53$, adj $p < 0.001$). This was visually confirmed through rank abundance curves (Figure 3), in which the dominant species differed across HOLC grades and between tree sizes. Further, the extent of their dominance in the community differed. We formally assessed species reordering and

FIGURE 1 (a) Historic HOLC grade designations for 36 neighborhoods across Baltimore City, MD (I). We subsampled these neighborhoods (II) using equal-area 3.3 ha hexagonal polygons (II) that contained both living street trees and viable locations where trees could be planted that were currently dead or empty (black dots in III and IV satellite imagery). (b) Occupancy of a given street tree location varied among HOLC grades and by tree size class. Each gray datapoint is the proportion of total viable street trees within a 3.3 ha hexagon that currently contains large (II), medium (III), or small trees (IV) or are currently empty without a living tree present (I). (c) Neighborhoods that were redlined (grade D) consistently contain a less diverse community of living street tree species as shown by individual-based accumulation curves using three diversity metrics that range in weighting the importance of species number versus relative abundance (V). *Note*: The analysis only includes street trees; any park trees (i.e. the top right of (a)III and (a)IV) were removed.



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turnover among neighborhoods to understand which components of β-diversity differed by HOLC grade. For large trees, we found that there was more reordering across neighborhoods within C- and D-grades compared with the B-grade (Figure 2c; $F_{3,137} = 10.63$, adj $p = <0.001$). We found no differences in species turnover among HOLC grades (Figure 2c; $F_{3,137} = 0.93$, adj $p = 0.428$). For small trees, there was less reordering across neighborhoods within D-grades compared with all other grades, meaning that tree communities have more similar ordering of dominant and rare species from one D neighborhood to another (Figure 2c; $F_{3,137} = 6.943$, adj $p < 0.001$). We also found greater species turnover in small trees across neighborhoods with B- or C-grades compared with D-grades (Figure 2c; $F_{3,137} = 6.87$, adj $p < 0.001$). However, overall tree community composition at the neighborhood level did not differ by grade ($F_{3,29} = 0.164$; adj $p = 0.110$; Appendix S1: Figure S4).

DISCUSSION

The street tree population of a city can comprise a significant and unique fraction of urban tree biodiversity and provide valuable ecosystem services (Avolio et al., 2015, 2018) sometimes at rates higher than urban forest trees (Ponte et al., 2021). Street tree diversity is generally higher in neighborhoods with older housing stock (Avolio et al., 2018), but additional factors remain under active investigation. Here, we demonstrate that neighborhoods that were redlined have lower present-day street tree diversity and different compositional patterns than higher graded ones. Further, the patterns found in communities of large trees compared with small trees may provide a window into past and current societal investment in urban neighborhoods.

Collectively, D-graded neighborhoods consistently had lower levels of street tree α-diversity than A-graded neighborhoods. This was true for species richness as well as metrics that more strongly weight relative abundance (Shannon's and Simpson's effective number of species) indicating that the patterns are not driven by a few rare species. Instead, these communities are likely to be less even and dominated by fewer tree species. As a more diverse urban forest supports a greater biodiversity of other taxa and may be more resilient to pest and disease outbreaks (i.e., the dilution effect) (Raupp et al., 2010), these differences may jeopardize the continued provision of critical biophysical and social–cultural ecosystem services to city residents into the future (Salmond et al., 2016). Furthermore, there may be consequences of low street tree diversity for nonhuman animals living in cities. Street tree diversity is likely to increase the diversity of insect herbivores (Schmitt & Burghardt, 2021) and, in turn, arthropod and avian predators (Lerman et al., 2021).

The surviving large street trees in a city have experienced decades of changing human and societal investments and values. As a result, we predicted lower occupancy of large trees and lower among-neighborhood species turnover and reordering of large trees in D neighborhoods compared with higher grades. Our analysis provides strong support for the first prediction, but little for the second. Specifically, we found that a viable street tree site in Baltimore is nine times more likely to contain a large tree if that location is in an A neighborhood rather than in a D neighborhood. Our findings of higher occupancy of large diameter trees in higher graded neighborhoods are consistent with other studies finding greater tree canopy in A-graded neighborhoods of cities throughout the USA (Locke et al., 2021). Because larger trees provide greater ecosystem service benefits than smaller trees (Nowak, 2020), neighborhood investment patterns of the past continue to contribute to present-day environmental inequalities. Contrary to our predictions, we found higher species reordering across neighborhoods with C- or D-grades compared with B-, and no difference in species turnover. While impossible to attribute to a direct cause, this pattern is consistent with equally diverse initial communities across grades, with higher random tree mortality in lower grades leading to differentially dominant species.

Baltimore City currently has sustainability goals of increasing tree canopy cover to 40% by 2040 and increasing biodiversity (City of Baltimore, 2019); our results regarding small (young) trees in the city may reflect the first goal. We find a higher present-day occupancy of small trees in D neighborhoods compared with B neighborhoods. We propose that this difference reflects the City's TreeBaltimore initiative and the work of partner organizations to address tree inequities. However, small tree communities in D-graded neighborhoods also have a significantly different composition and lower species turnover among neighborhoods than higher graded

FIGURE 2 Tree size distribution, community composition, reordering, and turnover of tree communities by neighborhood HOLC grade. (a) Smoothed kernel density estimate of tree sizes (dbh) for each HOLC grade. The dotted lines indicate the dbh cutoffs used to classify "small" and "large" tree designations used for the "old" versus "young" community comparisons. (b) Nonmetric multidimensional scaling (NMDS) of large and small tree communities. Ellipse denotes the 95% confidence interval for that distribution of points. (c) Reordering and species turnover of large and small trees among HOLC grades. Significant differences ($p < 0.05$) are denoted by letters.

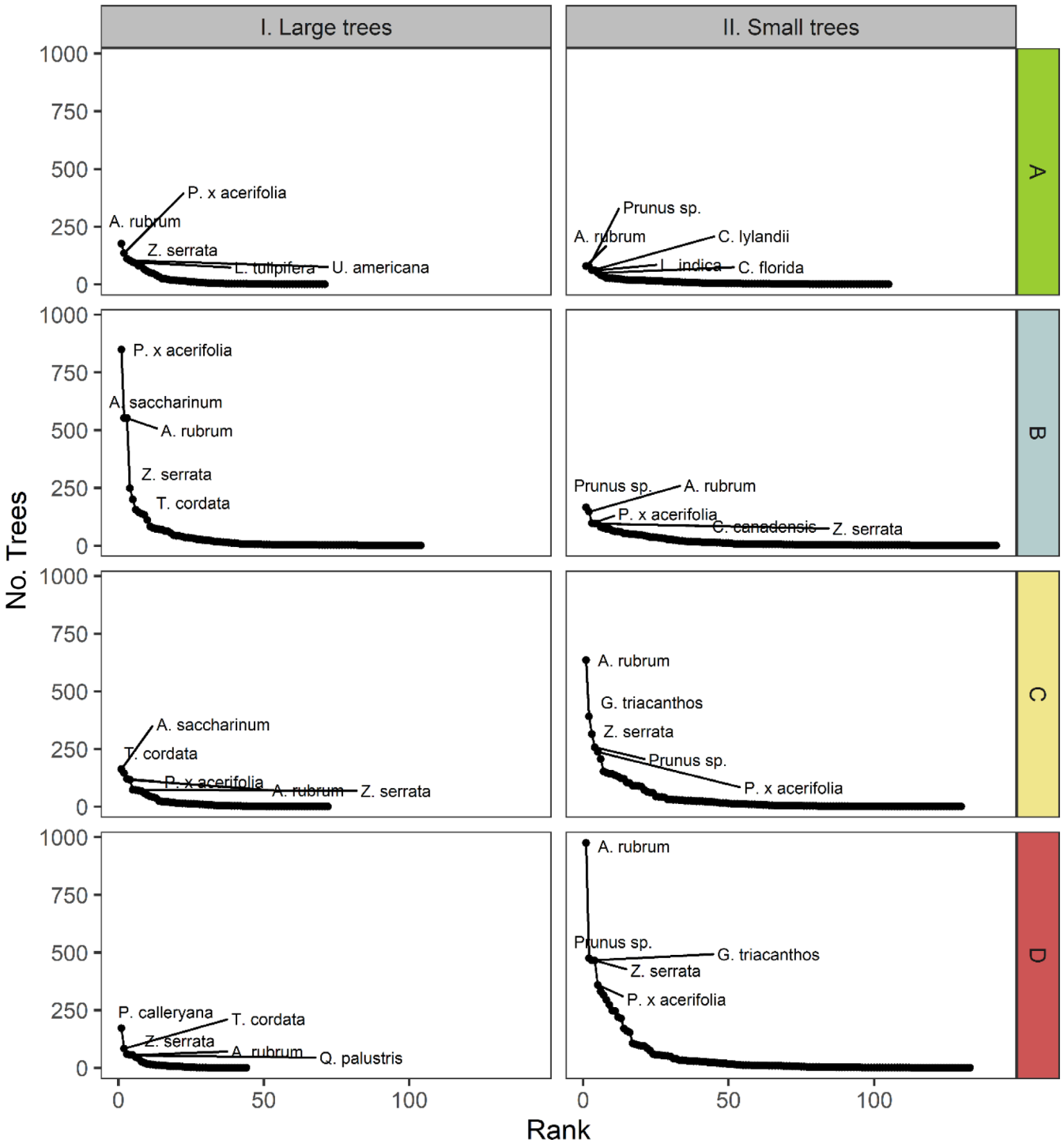

FIGURE 3 Rank abundance curves of large and small trees for each HOLC grade. The five most abundant species are labeled. Neighborhoods that were redlined have a less even (steeper curve) community structure of young (small) trees dominated by *Acer rubrum*.

neighborhoods because of the heavy reliance on *A. rubrum*. This species is commonly used because of its ability to grow in the difficult growing conditions of D neighborhoods (C. Murphy, personal communication). These conditions are likely to be reflective of environmental legacies of the city's history of institutionalized racism and segregation: narrow streets and sidewalks, compacted soils, high temperatures, poor air quality, and low canopy cover (Grove et al., 2018; Namin et al., 2020; Nardone et al., 2020). Unfortunately, *Acer rubrum* is also highly susceptible to the Asian long-horned beetle, an invasive species currently present on the East Coast of the USA, but not yet in Baltimore, MD (Raupp, 2006). This highlights a likely legacy feedback of redlining, past disinvestment may make it more challenging to successfully repopulate these areas with a diverse tree

community, limiting a city's ability to address biodiversity goals. To overcome feedbacks, cities could target higher tree survival through direct investment in rehabilitation of tree pits, appropriate species selection, protection of root zones, and consistent post-planting maintenance. However, to be successful, such tactics may require concomitant social investments to engage affected communities and mitigate fears of gentrification and displacement (Pickett et al., 2017).

Redlining in cities has a long history in the USA. We emphasize that our observational study relates a major social driver of urban heterogeneity to an ecological pattern, and in no way infers any causal relationship between redlining and the ecological patterns we have documented. While mounting evidence demonstrates statistical associations between redlining and the amount of vegetation (Hoffman et al., 2020; Locke et al., 2021; Schell et al., 2020), the role of other social–ecological factors must also be considered to fully understand drivers of ecological community structure in cities. Here we focused on street trees, whose establishment is entirely anthropogenically determined. It is possible that the patterns we observed in Baltimore may resemble patterns in other United States cities with a history of redlining, but this remains to be tested. These patterns may also be reflected in other components of the urban tree community, such as remnant forest patches and vacant lots more likely to be shaped by natural processes of recruitment and competition; this should be tested rigorously in future work.

In urban settings, classical community assembly processes are modified by complex human-driven legacies, lags, and feedbacks among decision-making, environmental constraints, and ecological interactions. Here, we document significantly lower street tree diversity and compositional differences in neighborhoods that were redlined. These compositional differences are likely to make the tree canopy in these neighborhoods less resistant to pests and resilient to disturbance and alter the distribution of other ecosystem services in cities provided by urban trees. We also found evidence that current planting practices aim to correct for past injustices by increasing the number of new trees installed in these neighborhoods. However, we did not document an increase in diversity of newly planted trees. If cities wish to avoid continued signatures of redlining within the street tree composition of urban neighborhoods, then replanting campaigns could focus on planting diverse tree cohorts, while also investing in site rehabilitation before tree planting and tree maintenance after planting to increase tree survival.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

The 2017–2018 tree inventory of Baltimore, MD is publicly available to download at https://baltimore.maps.arcgis.com/apps/webappviewer/index.html?id=d2cfbbe9a24b4d988de127852e6c26c8 (Baltimore City Department of Recreation & Parks, 2021) and was processed as described in our "*Methods*" section with code provided at https://doi.org/10.5281/zenodo.7117811. GIS shapefiles for HOLC grade designations are publicly available under a creative commons license download from the Mapping Inequality American Panorama Project (Nelson et al., 2019) at https://dsl.richmond.edu/panorama/redlining/#loc=10/39.676/-77.019&text=downloads by searching for Baltimore, MD.

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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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