



Original article

More than green: Tree structure and biodiversity patterns differ across canopy change regimes in Baltimore's urban forest

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ABSTRACT

Urban tree canopy is a vital component of modern cities, and many cities aspire to increase canopy cover to bolster ecosystem services. However, canopy-based ecosystem services are disproportionately provided by old, large-statured trees. Legacies of where trees have been maintained are evident in urban landscapes and loss of these older trees is not easily negated by current planting efforts. Furthermore, tree species composition varies considerably across the landscape and may determine present canopy longevity, structure, and contribution to ecosystem amenities or disamenities. In this study, we leverage long-term data from 188 i-Tree Eco plots in Baltimore, Maryland, USA, to examine how canopy changes across 10 years are related to tree structure and biodiversity patterns. We delineated five distinct clusters based on starting canopy conditions (high or low) and the trajectory of canopy change (gain or loss) and compared how tree structure, community composition, and turnover varied among clusters and across time. We found distinct patterns of tree structure and biodiversity across clusters that can help clarify links between land use, community composition, and ecosystem function in the urban forest. Residential land use typically had fewer and larger trees than forests or parks and residential clusters supported high biodiversity. Regardless of land use, areas with increasing canopy were dominated by native species while areas experiencing canopy declines were dominated by non-native species. In all, these findings help untangle the complexities of urban forest change and clarify different canopy vulnerabilities and assets across an urban landscape.

1. Introduction

Across the contiguous United States, trees in cities provide trillions of dollars of benefits (Nowak et al., 2002). These include, but are not limited to, shade and evapotranspirative cooling (Bowler et al., 2010; Gill et al., 2007), rainfall interception and runoff reduction (Zabret and Šraj, 2019), and improvements in residents' health and wellbeing (Willis and Petrokofsky, 2017). Broadly speaking, individual cities have recognized that urban forests are important resources for mitigating climate challenges (Gill et al., 2007), building resilience (Huff et al., 2020) and enhancing ecosystem services (Akbari et al., 2001; Escobedo et al., 2011). Regardless of the known benefits, the term "urban forest" is not clearly defined or applied (Livesley et al., 2016). Generally, it refers to different permutations of "all trees in the city" (Konijnendijk et al., 2006), but in practice, assessments may be broken down further, for example, to trees owned and maintained by the city (Clark et al., 1997),

or trees that remain in forested contexts where canopy extends over natural soils (Livesley et al., 2016). Regardless of confusing nomenclature, many temperate cities are striving to achieve 40% canopy cover in their urban forest, as was previously recommended by American Forests (<https://www.americanforests.org/blog/no-longer-recommend-40-percent-urban-tree-canopy-goal/>), but see Ziter et al., 2019), although strategies and success rates are variable. Indeed, changing the tree canopy across an urban landscape is more complex than simply planting trees.

Biophysical and social features coalesce to determine where urban plants—including trees—are located (Aronson et al., 2016; Avolio et al., 2015; Roman et al., 2018a,b). Some trees in cities are legacies of previous habitats or land uses and have existed on the landscape since before it was heavily urbanized. Others were recruited more recently via natural germination processes or invasions, and still many others were planted purposefully by people, who have different priorities and

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purviews for planting (Conway and Vander Vecht, 2015; Nowak, 2012). Furthermore, these mechanisms for tree establishment function across a matrix of public and privately owned land parcels, which are known to differ in tree species, ages, and sizes (Steenberg, 2018). Since trees come to exist on the heterogeneous landscape through both social and ecological mechanisms, it is important that urban trees and the canopy they comprise are evaluated in a way that is both ecologically and socially relevant (Conway and Bourne, 2013). Indeed, much previous work has attempted to pare down extreme urban heterogeneity by evaluating differences in tree diversity within or across different land uses (Bourne and Conway, 2014; Swan et al., 2017). While this approach has helped justify the examination of e.g., residential neighborhoods, private land ownership poses logistical challenges for access and long-term study (Jim and Zhang, 2015). Furthermore, attempting to fit the ecology of urban trees into a socially-defined land use classification system may not always accommodate the complexity and heterogeneity of urban ecological systems (Cadenasso et al., 2007).

Disparate establishment, management, and priorities result in an urban canopy that is the sum of highly variable and dynamic parts. Indeed, trees of different sizes, structures (the physical form of a tree including height, number of stems etc. e.g. shade trees vs. small ornamentals), and species are distributed across a landscape that is heterogeneous in both biophysical and social conditions (McPherson et al., 1997). Individual tree species have different lifespans and intrinsic growth rates, are differentially suited to specific habitats, and may contribute to different ecosystem services or disservices (Vaz Monteiro et al., 2017). For example, large broadleaf trees provide more shade and sequester more carbon (McPherson and Muchnick, 2005; Nowak and Crane, 2002), but are also more likely to cause damage in a storm (Hauer et al., 1993; Roman et al., 2018a,b; Schmidlin, 2009). Single-stemmed trees or strips of monoculture plantings may be liabilities for canopy losses in extreme weather or pest outbreaks (Laćan and McBride, 2008), but can also be perceived positively by local residents as they appear tidy and cared-for (Nielsen and Jensen, 2007).

Tree communities are known to differ significantly across neighborhoods (Avolio et al., 2018), and the precise locations of individual trees influences the costs and benefits provided to local residents (Dobbs et al., 2011; Palliwoda et al., 2020). Fine-scale differences in the numbers and species of trees per canopy area can have meaningful consequences for ecosystem services, ecological function, and sustainability, as the species and size of trees in a given site are known to influence for example, carbon storage and pollution mitigation (Potgieter et al., 2019). However, data that adequately define these differences and variances across an urban landscape through time are often limited. City-wide tree inventories are often restricted to municipally-owned trees, which excludes a substantial component of the canopy which is privately owned and managed (Daniel et al., 2016; Nowak et al., 1996), and data on tree plantings often do not account for trees already present on the landscape (Vogt et al., 2015; Zipperer et al., 1997). Analyses that rely on remotely sensed or aerial photography for changes through time or in space (Parmehr et al., 2016; Walton et al., 2008) do not offer the necessary resolution to identify biodiversity, which is important to fully comprehend ecological function and community dynamics that creates the urban forest. Attempting to include and understand all types of trees in all types of places is a critical step towards managing the urban forest for equity and ecosystem function.

Here, we leverage a long-term tree dataset from 202 plots distributed randomly across Baltimore, Maryland (USA) to examine how urban tree canopy, structure, and diversity changed over a ten-year period. We quantify canopy and tree diversity gains, turnovers, and losses as a critical and novel step in defining the urban forest and identifying metrics and conditions that support sustained increases in canopy cover. We use a novel clustering approach geared at assessing regimes of canopy change across the City to ask the following questions:

- 1) How is canopy change associated with specific land uses across Baltimore City?
- 2) How are particular regimes of canopy change associated with the structure, size, or density of trees?
- 3) How do species richness and turnover differ between groups of sites with different canopy change regimes?

2. Methods

2.1. Study area

Baltimore, Maryland, (USA) is a temperate city located on a harbor on the Atlantic seaboard. Summers are typically hot and humid (average 26.0 C with 10 cm rainfall monthly), while winters are mild (average 0.8 C). The city falls in the range of the historic northern forest and includes several large patches of second-growth forest within the city limits, in addition to other typical urban forest elements such as street trees and maintained parks. Baltimore also has a complex social landscape. The city is said to be a “City of Neighborhoods”, and long-term demographic patterns are largely influenced by a long history of segregation and racism. Since 1950, Baltimore’s population has shrunk by more than 35% (planning.baltimorecity.gov), resulting in high levels of vacancy that has resulted in open land parcels that require unique solutions.

2.2. Data

Baltimore is also home to the Baltimore Ecosystem Study (BES), a National Science Foundation funded Long-Term Ecological Research (LTER) site that has been supporting the collection of myriad data on the system since 1998. Tree data presented here were collected in accordance with i-Tree Eco protocols developed by Nowak (2020) for comparative urban forestry between cities and for ecosystem services forecasting. Full datasets can be found at <https://portal.edirepository.org/nis/mapbrowse?packageid=knb-lter-bes.1000.170>.

In 1999, researchers established 202 permanent 11.3 m radius plots, which were distributed across public and private land in the city of Baltimore for sampling woody plants. Plot locations were established with a stratified random sampling approach using land use categories obtained from a 1996 municipal land use map in order to adequately represent the heterogeneity of urban land use types. The woody plant inventories contain abundance data for individual species as well as measurements of tree structure (DBH, tree height, number of stems, canopy width). In 2004, 2009, and 2014, researchers resampled woody plants and also assessed land use, ground cover, and tree canopy cover as estimates of percent cover across each plot (USDA Forest Service, 2020).

In order to make direct links between canopy, biodiversity, and ecological function, we use existing datasets from 2004, 2009, and 2014, when plot-level canopy cover and biodiversity data were collected. We excluded sites with missing canopy cover data at one or more of these time periods, and we eliminated one site that had outlier measurements for canopy cover in 2009 and was likely sampled or recorded incorrectly. In total, we included 188 sites distributed across the city of Baltimore in our investigation of tree canopy change. To analyze tree structure and biodiversity dynamics, we assessed all plots that had trees present at all three sample points (i.e. trees were present to evaluate and there were no missing tree records, $n = 100$, Fig. 1).

2.3. Canopy change regime clustering

Because canopy change through time is non-linear, we sought to group sites based on similar starting conditions and short- and long-term trajectories. We used a hierarchical, agglomerative clustering protocol similar to Anderson and Minor (2020) to classify plot-level canopy change in sites sampled across the three time points (see Belbin and McDonald, 1993; Steenberg, 2018). To do this, we compiled data for each plot location to create variables for canopy cover per plot in 2004

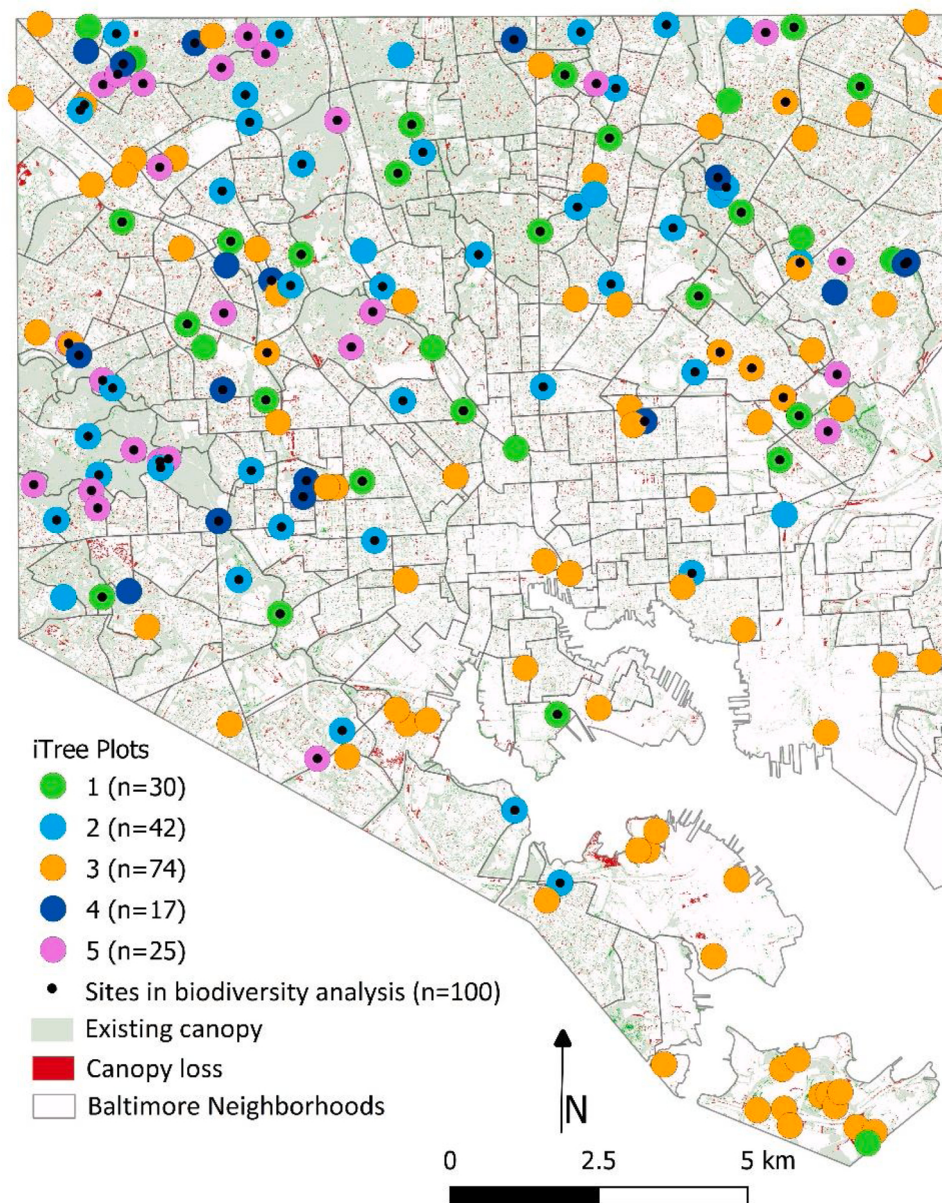


Fig. 1. Map of 188 i-Tree Eco plots in Baltimore, MD, USA, clustered by canopy change regime (colored circles). Sites with trees present in all 3 years (marked with a black dot) were used in the biodiversity analysis. Canopy cover and loss layers are generated from remotely-sensed data, collected as part of BES (Cary Institute of Ecosystem Studies and O'Neil-Dunne, 2017).

(starting condition), canopy change from 2004 to 2009, canopy change from 2009 to 2014, and net canopy change (from 2004 to 2014). We standardized all of our variables to have a mean of zero and a variance of 1, and used the maximum average silhouette width (the distance between clusters, (Lengyel and Botta-Dukát, 2019)) to determine that 5 clusters was an optimal structure for this dataset (Supplemental Fig. 1). From here, we calculated a distance matrix and used a Ward's D clustering method, which minimizes the within-cluster variance when building the dendrogram. We then assessed whether canopy changes differed between our five determined clusters using a one-way analysis of variance (ANOVA) for each variable. We followed significant ANOVA with post-hoc pairwise analysis using Tukey's Honest Significant Difference test. To account for repeated analyses, we assessed ANOVA significance at $\alpha = 0.01$, and Tukey's HSD significance at $\alpha = 0.05$.

2.4. Analyzing changes in tree structure

We assessed differences in four dimensions of plot-level tree structure across the three sample periods, including mean number of stems per tree, mean tree height and canopy extent per tree, and mean primary (largest) stem DBH with linear mixed effects models using the "lme4" package in R 3.10 (Bates et al., 2020). Each of the structural variables was considered at the plot-level separately in a model with fixed effects for cluster ID (factor leveled 1–5) and time (continuous variable, 2004 = year 1, 2009 = year 6, 2014 = year 11). Year was also treated as a random effect to account for shared responses to weather or other unknown year-specific drivers across plots. We assessed effect sizes and upper/lower 95% confidence intervals. We compared clusters 1–4 to the patterns of cluster 5 because cluster 5 had high starting canopy (median 80.2%) and a majority of these sites were in continuous forest patches (see Results, Table 1), which are known to be distinct from other elements of urban forests (Steenberg, 2018).

Table 1

Land use classifications for each i-Tree plot location (N = 188). Bolded numbers indicate the most frequent land use association for each cluster.

Cluster	Commercial/ Industrial	Forested Area	Institutional	Park	Residential	Transport	Vacant	Total
1 (low canopy/loss)	5	1	0	1	19	2	2	30
2 (low canopy/gain)	3	5	6	5	21	1	1	42
3 (no canopy)	30	1	5	5	22	8	3	74
4 (high canopy/loss)	0	3	1	1	11	1	0	17
5 (high canopy/gain)	0	14	0	3	7	0	1	25
Total	38	24	12	15	80	12	7	188

2.5. Analyzing diversity patterns

We assessed biodiversity patterns for all sites with trees present in 2004, 2009, and 2014 (N = 100). We removed sites that had no trees present as well as sites where biodiversity data were missing for any of the three time points. While sites without complete biodiversity data recorded were distributed across clusters, sites without trees present were largely grouped in cluster 3; only 6 sites (of 74) in cluster 3 had trees present and biodiversity recorded at all three time points. For this reason, and for the negligible amount of canopy contributed by cluster 3 across the city, we eliminated sites in cluster 3 from subsequent biodiversity analyses.

To analyze differences in species composition and abundance we used permutational multivariate analysis of variance (PERMANOVA, Anderson, 2017) with 9999 permutations, using the “adonis” function (vegan package R 4.0.3, Oksanen et al., 2020) where the full community (including abundance of each species) was the response variable. We

included first-order terms for cluster and year as well as for plot ID to account for repeated measures in our design. We also included interaction term cluster*year to assess cluster-specific variation in community composition over time.

To assess the species turnover across clusters, we analyzed the Sørensen's Index for 999 resampled permutations of randomly selected pairs of plots in clusters 1, 2, 4, and 5 through time (“betapart” package in R 4.0.3, Baselga et al., 2021; Baselga and Orme, 2012). Sørensen's Index is a measure of similarity between two sites, where values range from 0 (completely distinct) to 1 (identical). We used an analysis of variance (ANOVA) followed by Tukey's Honest Significant Difference post-hoc test and assessed significance at $\alpha = 0.05$. We also inspected the species accumulation curves for each cluster to get an idea of how adding additional sites increases the number of species reported per cluster. A species accumulation curve with a continuous slope indicates additions of new species with each additional sampled site while a reduction in slope with added sites indicates the community has been

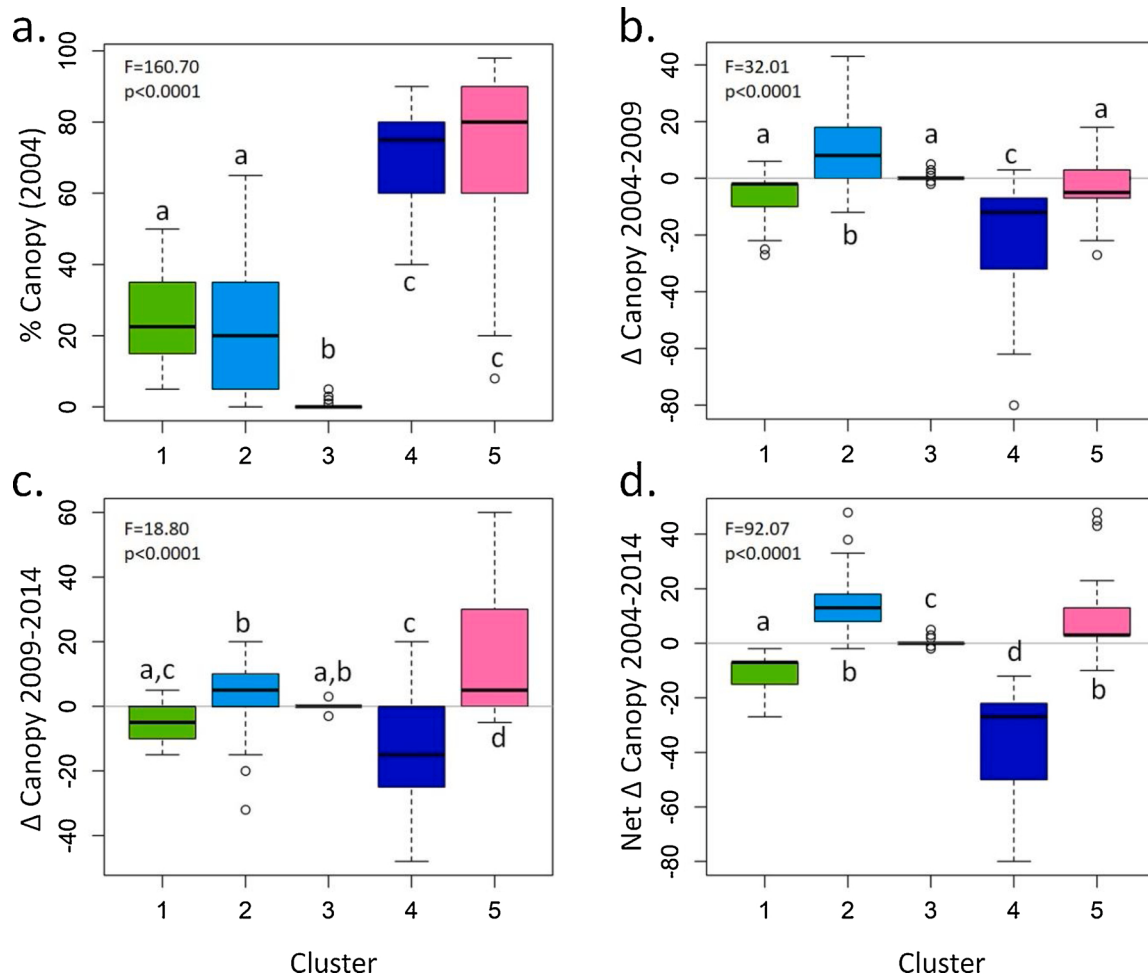


Fig. 2. Differences in a) starting canopy (2004) b) canopy change (2004-2009) c) canopy change (2009-2014) and d) net canopy change (2004-2014) across clusters. Letters denote significant Tukey's Honest Significant Difference tests, assessed at $\alpha = 0.05$.

fairly comprehensively sampled.

3. Results

3.1. Canopy change regime clustering

We differentiated 5 clusters, organized around both starting canopy cover and the primary trajectory of canopy change during the study period (Fig. 2). Clusters included between 17 and 74 i-Tree plots, and the final dendrogram was a strong fit for the data (agglomerative coefficient 0.98). The biggest group (cluster 3) consisted of sites that had little or no canopy in 2004 and did not change much from these starting conditions. Clusters 1 and 2 had low canopy in 2004 (median ~20% canopy cover) while clusters 4 and 5 had high canopy in 2004 (median ~80% canopy cover). There were two primary trajectories of canopy change among clusters that began with either high or low canopy. Clusters 1 (low canopy) and 4 (high canopy) exhibited canopy losses over time, while clusters 2 (low canopy) and 5 (high canopy) primarily gained canopy over time. In short, we identified five regimes of canopy change that map accurately across our clusters: c1) low canopy/loss, c2) low canopy/gain, c3) no canopy, c4) high canopy/loss, and c5) high/canopy gain (Fig. 2).

Canopy metrics differed significantly among the cluster types: starting canopy (ANOVA $df = 4$, $F = 160.70$, $p < 0.001$), change 2004–2009 (ANOVA $df = 4$, $F = 32.01$, $p < 0.0001$), change 2009–2014 (ANOVA $df = 5$, $F = 18.80$, $p < 0.0001$), and net change 2004–2014 (ANOVA $df = 4$, $F = 92.07$, $p < 0.0001$). While some clusters exhibited no difference for an individual variable, our post-hoc analyses demonstrated that redundancies between clusters were typically only identified in one variable (Fig. 2).

Clusters were distributed across the landscape (Fig. 1), but distinct canopy change regimes were associated with particular land use classifications (Fisher's Exact test $p < 0.001$; Table 1). Relevant land use classifications were collapsed from field assignments by Jiang et al. based on methods from Pouyat et al. (2007). Clusters 1, 2, and 4 were primarily associated with residential areas. Cluster 5 had high canopy to begin with and experienced canopy gains through time, and a majority of these sites were in forested areas, including designated natural forest habitat in city parks and recreational spaces (e.g. for hiking). Cluster 3, which had little or no tree cover in any time point, was primarily associated with commercial and industrial sites, but also included a large number of residential sites.

3.2. Analyzing changes in structure and diversity

3.2.1. Structure

Most trees across all clusters were single stem trees (approximately 81% of all trees in each year) and the remaining trees ranged from 2 to 10 stems in 2004 and 2–6 stems in 2009 and 2014. There was some variation in the prevalence of multi-stem trees across clusters; plots in clusters 1, 2, 3, and 4 all had a higher mean number of stems per tree than the more forested sites in cluster 5 (Table 2). The highest mean number of stems per tree was found in cluster 4 (high canopy/loss; Supplemental Table 1), which could be related to the high abundance of non-native species such as *Ailanthus altissima* and *Ulmus pumilia* in that cluster (see below).

Trees in plots assigned to cluster 5 (high canopy/gain) were, on average, 2 m shorter than trees in clusters 2 and 3, but did not differ from trees in cluster 4 (Supplemental Table 1). Cluster 1 had the tallest trees in 2004 and 2009 (~11 m), but experienced a 3 m decline in mean tree height between 2009 and 2014, which was driven primarily by the addition of small trees in the 2014 census (Fig. 3).

Overall, there was net reduction in plot-level canopy extent from 2004 to 2014, though the confidence interval for the fixed year effect included zero (Table 2). Plots in cluster 5, which generally had more individual trees, had smaller average crown widths than those in clusters

Table 2

Results of linear mixed effects models for tree structural components within plots: mean tree height, mean crown width, mean number of stems per tree, and mean DBH of primary stems. All models were assessed at the plot level ($N = 100$). Estimates of cluster effects and CI are in relation to a baseline of cluster 5, which was primarily forested sites.

Model	Parameter	Estimate (Variance + SD)	Lower 95% CI	Upper 95% CI
Number of Stems Per Tree ~ Cluster + Year + 1 Year	Intercept	1.19	1.08	1.30
	Cluster 1	0.24	0.12	0.37
	Cluster 2	0.26	0.15	0.36
	Cluster 3	0.17	0.08	0.27
	Cluster 4	0.49	0.35	0.63
Tree Height ~ Cluster + Year + 1 Year	Year	0.01	-0.04	0.05
	1 Year	(<0.01 + <0.01)		
	Intercept	8.53	7.52	9.55
	Cluster 1	1.94	1.05	2.83
	Cluster 2	1.95	1.21	2.69
	Cluster 3	2.47	1.80	3.14
	Cluster 4	0.91	-0.08	1.89
	Year	-0.03	-0.17	0.09
	1 Year	(0.06 + 0.25)		
	Intercept	4.31	3.11	5.50
Crown Width ~ Cluster + Year + 1 Year	Cluster 1	0.86	0.37	1.35
	Cluster 2	1.48	1.08	1.89
	Cluster 3	1.30	0.93	1.67
	Cluster 4	0.25	-0.30	0.79
	Year	-0.03	-0.19	0.13
Primary Stem DBH ~ Cluster + Year + 1 Year	1 Year	(1.56 + 0.39)		
	Intercept	12.40	10.66	14.31
	Cluster 1	8.11	5.67	10.54
	Cluster 2	5.52	3.49	7.54
	Cluster 3	6.49	4.66	8.32
	Cluster 4	3.25	0.54	5.96
	Year	0.10	-0.94	0.23
	1 Year	(<0.01 + <0.01)		

1 (low canopy/loss), 2 (low canopy/gain), or the no canopy cluster 3 (Supplemental Table 1), but did not differ significantly from trees in cluster 4. However, plots in cluster 5 had nearly three times as many trees as plots in cluster 4 (median $\pm$ SE, cluster 5: 17 ± 2.4 vs. cluster 4: 6 ± 1.1 trees per plot). Clusters 1, 2, and 4 had lower abundances, with medians of three trees per plot in clusters 1 and 2 and one tree per plot in cluster 3.

Plots in clusters 1–4 had significantly larger DBH trees on average than plots in cluster 5, due to the higher density of small DBH trees in cluster 5 (Fig. 3). Trees in cluster 1 were the largest on average, and the decline in mean DBH between 2009 and 2014 was associated with additions of small trees rather than the loss of large trees (Fig. 3). There was also a sizable addition of small trees in cluster 4 sites between 2009 and 2014, but this did not result in an appreciable change to mean DBH because cluster 4 had a much higher density of large trees in 2004 that continued to grow through time (Fig. 3). However, plots in cluster 4 did lose some mid-size trees (Fig. 3), which helps to explain the consistent mean DBH and decreasing canopy cover through time, despite the increase in the number of trees across all cluster 4 plots (from 104 in 2009 to 144 in 2014).

3.3. Diversity

Across the city, *Fagus grandifolia* (American beech) was the most abundant species, comprising between 12.9% and 14.8% of stems recorded in any given year. The three next most abundant species (*Prunus serotina*, *Ailanthus altissima*, and *Quercus rubra*) comprised approximately 5–7% of stems, and indicated some turnover through time. In 2004, *P. serotina* (black cherry), *A. altissima* (tree of heaven), and *Q. rubra* (red oak) each had between 75 and 55 stems, respectively, while these numbers diminished to 47 stems for each species by 2014. Additionally, by 2014, *Ulmus pumilia* (Siberian elm) was among this second tier of dominant species, representing 4.7% of stems across

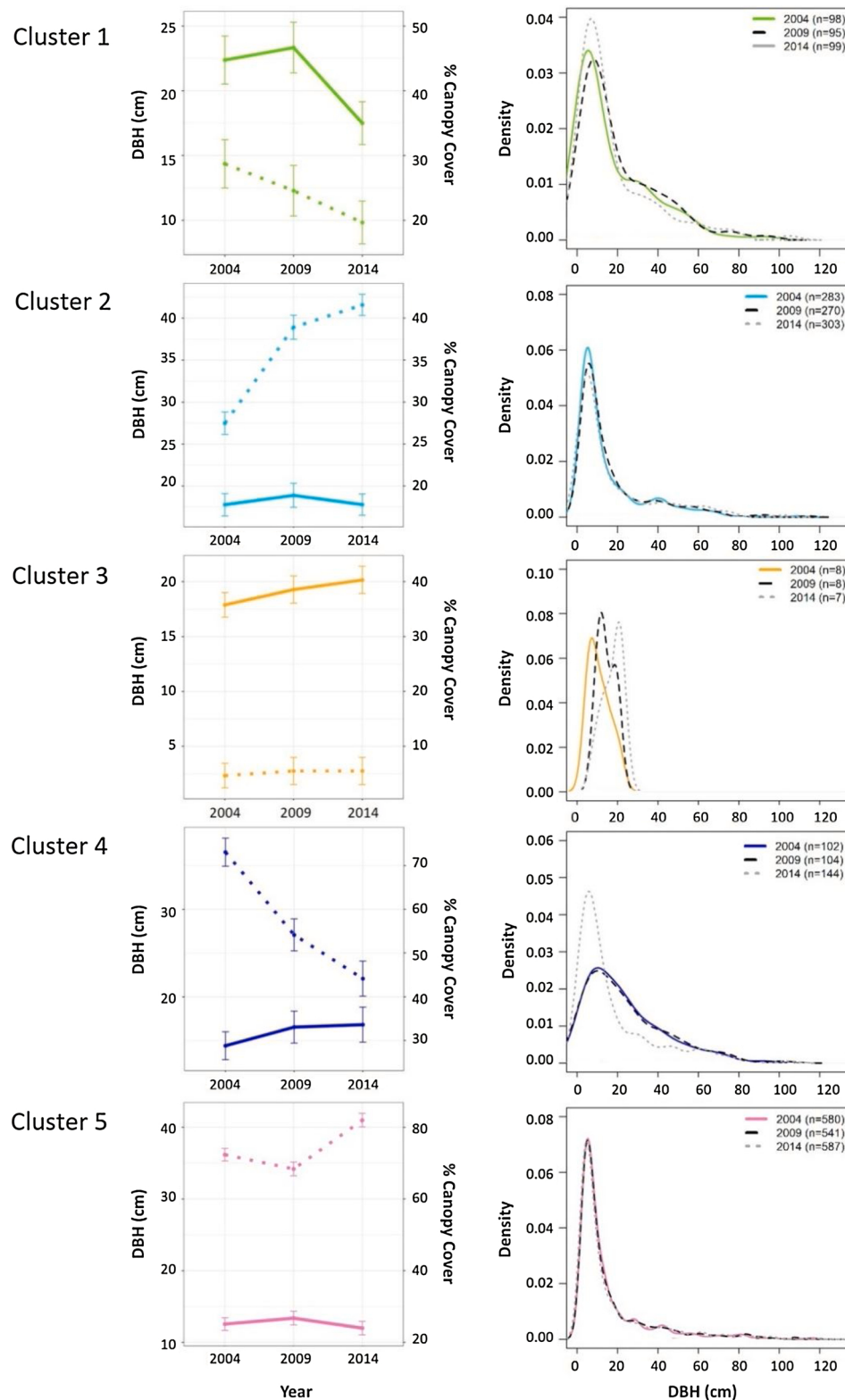


Fig. 3. Change in mean DBH of primary stems (solid line), canopy cover (secondary y axis, dashed lines), and density of DBH distribution over time for each cluster.

Baltimore.

While not a factor used in cluster identification, species composition differed significantly among the five cluster types (PERMANOVA results Table 3, community results Supplemental Table 2). We did not find a significant directional change in community diversity across the study period (year effect, Table 3, in any cluster type (Cluster* Year effect, Table 3). Cluster 2 (low canopy/gain) had the highest species richness,

with 56–60 species documented in each sample year (264–276 total trees per year, see Supplemental Table 2 for cluster-specific tree diversity data through time). Cluster 1 (low canopy/loss) and cluster 4 (high canopy/loss), which were also located primarily in residential sites, had much lower species richness, with 33–35 species (81–100 total trees) and 26–29 species (102–137 total trees), respectively. Species accumulation curves for all clusters had steeply increasing slopes

Table 3

Results of PERMANOVA analysis for the full community at each site, grouped by cluster and analyzed over time.

Variable	DF	F	R ²	P
Cluster	1	6.84	0.022	<0.001 ***
Year	1	0.24	<0.001	1
Plot ID	1	3.75	0.012	<0.001 ***
Cluster * Year	1	0.10	<0.001	1
Residual	295		0.956	
Total	299		1.00	

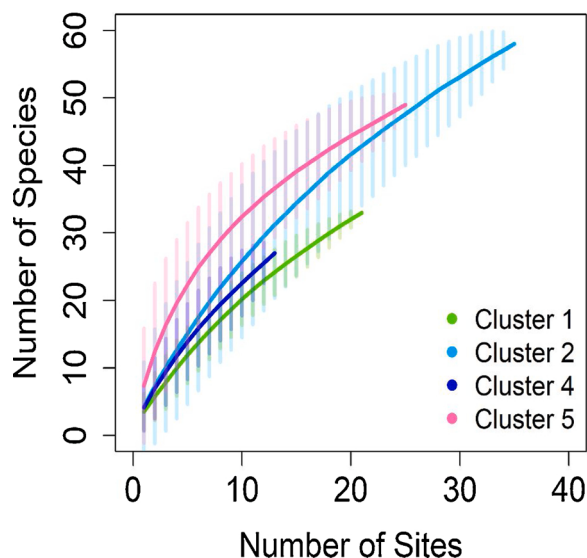


Fig. 4. Species accumulation curves by cluster for biodiversity data at sites in clusters 1, 2, 4 and 5 (N = 94). Sites from cluster 3 were omitted due to small sample size and few trees. Data is shown for 2009, but other years displayed the same pattern.

(Fig. 4), indicating that sampling of sites does not fully capture the taxonomic richness of tree communities across our study system. The cluster that captured sites with the most intact canopy, cluster 5 (high canopy/gain), also had the highest species richness. This cluster also showed the most pronounced reduction in slope through time, consistent with high dominance of a single species (*F. grandifolia*, Supplemental Table 2).

Plots in the two clusters with canopy gains over our study period (cluster 2 and cluster 5) were both dominated by native species (Supplemental Table 2). *Fagus grandifolia* (American beech) was the most abundant, followed by *Prunus serotina* (black cherry). *Fraxinus* species were prominent in these plots as well, although *Fraxinus americana* was lost from cluster 2 after 2009.

Tree diversity in cluster 4, comprised of predominantly residential plots with high starting canopy that has declined, was particularly dynamic during the study period. Non-native *Ulmus pumila* (Siberian elm) became dominant across plots in cluster 4 between 2009 and 2014. There were no records of *U. pumila* in cluster 4 in the first two sample years, but by 2014, it comprised nearly a quarter of the trees recorded across cluster 4 plots, due to an increase from zero to 34 stems of *U. pumila* at a single plot. Other non-native species present in cluster 4 remained consistent through time, including *Ailanthus altissima*, and *Morus alba* (white mulberry). *Pinus strobus* (eastern white pine), a native species of conifer found in Eastern hardwood forests, was also abundant.

Sørensen's index, a measure of similarity between two sites, had the highest difference between the clusters with low starting canopy (Fig. 5). Although both clusters are primarily residential, Sørensen's index was greatest in cluster 2 (low canopy/gain), (Fig. 5, ANOVA df = 3, F = 275.60, p < 0.0001) and lowest in cluster 1 (low canopy/loss).

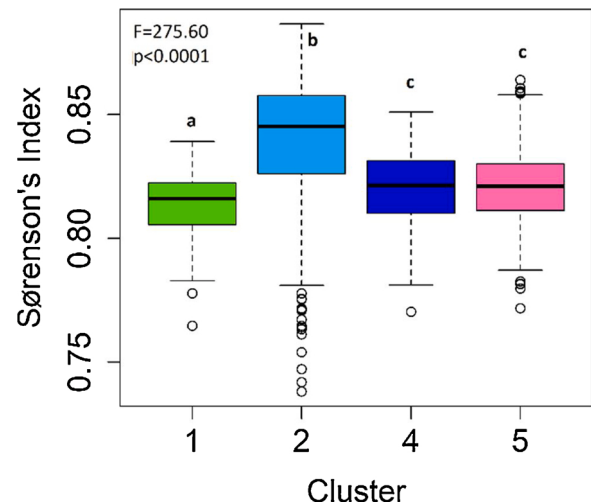


Fig. 5. Boxplot of Sørensen's indices for presence/absence communities at resampled pairs of sites in clusters 1, 2, 4, and 5 (N = 94, permutations per site = 999). Black lines indicate medians and boxes include quartiles above and below. Sites with a Sørensen's index of 1 are identical, and lower values indicate higher dissimilarity. Site 3 is excluded from this analysis. Graphed data is from 2009, but other years displayed the same pattern.

4. Discussion

Worldwide, cities are making conscious efforts to increase urban tree canopy to bolster ecosystem services. However, increasing canopy across an urban landscape is a complex and non-linear process that must be based on an understanding of how individual trees come to exist in their current location and structural form across a social-ecological system (Roman et al., 2018a,b). Here, we leverage a longitudinal urban tree dataset from Baltimore, Maryland, USA, to examine how community composition and tree structure vary in space and time, through the lens of fine-scale canopy change. While percent canopy cover across the plots did not change in a meaningful way between 2004 and 2014 (mean $\pm$ SD 26.05% $\pm$ 30.43 in 2004, 25.32% $\pm$ 31.33 in 2014), our results highlight the considerable variation in the fine-scale changes that are often glossed over when assessments are made at too broad a scale. Importantly, these fine-scale changes can represent larger losses and compositional changes for specific land use types and neighborhoods. Our methods and findings support a clear, process-based identification of canopy change patterns in both space and time at scales most relevant to urban residents and planners. We found that sample plots clustered into five different canopy change regimes based on starting conditions and trajectories of change over the 10-year period of data collection. We show that these five canopy change regimes (no canopy, all combinations of high/low starting canopy and gain/loss through time) are associated with different land use types and further describe differences in plot-level tree structure and community composition, which reflect different vulnerabilities and assets in Baltimore's urban forest.

Across Baltimore, 85% of the sites that began with higher canopy cover (clusters 4 and 5) dramatically exceed the 40% city-wide canopy cover goal, but sites with the highest canopy cover at the initial sample period (in 2004) were subject to different patterns of change through time. Canopy cover in mostly forested plots (cluster 5) did not decline from 2004 to 2014, while cover in more residential plots (cluster 4) declined by an average of 21% during the same period (Fig. 2d). Differences in tree structure between these cluster types provide some indication for understanding how changes in canopy occurred (e.g. growth vs. planting, loss of many small vs. single large trees). A greater number of smaller trees were identified across cluster 5 plots in later sample dates, which resulted in a lower mean DBH but not a reduction in

the number of large trees or canopy extent (Fig. 3). We believe this is indicative of high sapling recruitment in these more intact forested sites (Buchholz and Pickering, 1978), and is likely evidence of trees sizing into the i-Tree Eco sample protocol (minimum 1 inch (2.54 cm) DBH). While cluster 4 plots also experienced a shift towards more abundant and smaller stem trees, they seemed to have lost a number of mid-size stems by 2014, contributing to overall canopy declines (Fig. 3). It is possible that increases in the prevalence of smaller trees in cluster 4 were offset by faster growth of the larger trees, as might be expected in residential areas where trees experience less competition for resources than in forested areas (Pace et al., 2018).

Linkages between tree structure, community composition, and canopy cover were especially apparent in the sites with low starting canopy (clusters 1 and 2), which were predominantly residential land use. Between clusters 1 and 2, over 40% of plots had only one tree, and these trees tended to be larger in DBH, height, and individual crown extent. These results are unsurprising as light competition is much lower among sparse neighborhood trees (Pace et al., 2018), but suggest that many residential neighborhoods may be vulnerable to significant canopy loss resulting from the loss of a single large tree. While they had different starting conditions, residential sites in clusters 1 (low starting canopy) and 4 (high starting canopy) both experienced canopy losses, which were likely driven by the simultaneous loss of mid-size trees and replacement with smaller trees from 2009 to 2014 (Fig. 3). Unlike clusters 2 and 5 which were both dominated by native species, changes in clusters 1 and 4 were associated with high densities of small-statured non-native species in these clusters, including *Ailanthus altissima*, *Morus alba*, and *Ulmus pumila*, which generally have both high fecundity and high mortality rates (Nowak et al., 2004). *U. pumila* was a particularly notable invader in one cluster 4 site, with 34 stems established between 2009 and 2014. While the spontaneous germination of non-native trees in the urban forest is unavoidable and sometimes beneficial (Kowarik, 2005; Riley et al., 2018), these potential benefits must be weighed against negative perceptions of scrubby, unkempt vegetation and the risks of losing associated biodiversity and ecosystem services supported by large-statured native trees (Burghardt and Tallamy, 2015). In all, the similarities and differences we documented in Baltimore's residential urban canopy emphasize the need for and challenges of protecting, managing, and increasing residential tree canopy. This is particularly important as the social dynamics of residents within a city are known to be profound drivers of urban forest composition and cover (Roman et al., 2021).

Across plot types that had low canopy cover in the initial sample period (clusters 1 and 2), a key difference in trajectory in subsequent years seems to be associated with land use and ownership. Cluster 2 (low canopy/gains) included more institutional and public sites (e.g. campuses, manicured city parks), while cluster 1 (low canopy/losses) was predominantly residential. Local ordinances and city investments in urban forest management have been linked to increased canopy extent (Hilbert et al., 2019), and institutions are powerful forces in shaping culture and collective environmental actions (Ostrom, 2004). This difference in trajectory between clusters 1 and 2 highlights challenges with defining and managing an urban forest that has diverse stakeholders at multiple scales. Additionally, our findings may exemplify how management and resource commitment by the City or other institutional stakeholders (e.g. college campuses) may be needed to achieve broader canopy sustainability goals. More than 75% of the 202 plots in this study were classified as having low or no canopy (interquartile range: 0–43% canopy cover) and a majority of these low-canopy plots were located along the perimeter of Baltimore's Inner Harbor, the major downtown business and tourist district located at the southeast end of the city (Fig. 1). While tree survival in these sites is likely to be challenging due to high impervious surface cover and shading from built infrastructure, downtown areas represent an opportunity for municipal and institutional engagement in targeted canopy establishment.

Differences in taxonomic richness and species turnover across plots

with different canopy change regimes may indicate variation in local ecosystem services, corroborating the understanding that tree diversity is a critical component of ecosystem function (Morgenroth et al., 2016). Most notably, our data suggest that a high dominance of non-native species is associated with canopy losses through time. This is an important component of understanding the future of ecosystem services from trees in cities. While individual non-native trees contribute comparable regulating, supporting, and cultural services as similarly-sized native species (Schlaepfer et al., 2020), our results suggest that plot-scale and city-scale ecosystem services resulting from the tree canopy may be jeopardized in areas with a higher component of non-native species. Beyond the native/non-native dichotomy, there are other tree functional classifications that provide insight into the distribution of ecosystem services across a city. For instance, while conifers were rare across Baltimore, *Pinus strobus* accounted for between 8.8% and 12.7% of trees consistently across years in cluster 4 plots (high canopy/loss). This species, and evergreens more generally, provides different ecosystem services and habitats compared to deciduous trees (Manes et al., 2012; Marini et al., 2017), and also support different biogeochemical and hydrological profiles in urban soils and runoff (Michopoulos 2011). Biodiversity patterns were most similar across cluster 2 (low canopy/gains) and 5 (high canopy/gains) plots. Both cluster types were dominated by native species *Fagus grandifolia* and *Prunus serotina* and both gained canopy during the study period. *F. grandifolia* is a relatively slow-growing species, but it currently comprises almost 10% of the basal area of the urban forest sampled across Baltimore. Its dominance at sites where canopy is increasing suggests that it may be an important facilitating species in this urban forest and warrants further examination. Likewise, the predominance of *F. grandifolia* (American beech) in these positive canopy trajectory clusters highlights a potential vulnerability of this urban canopy to disease, including beech scale, a fast-spreading and aggressive cause of tree mortality inflicted by beech bark beetle infestations (*Cryptococcus fagisuga* Lind.) (Hemiptera; Coccidae), particularly as the pathogen has been identified in Maryland (<https://extension.umd.edu/hgic/topics/beech-bark-disease>). Loss of beech trees in residential and forested parts of the city that are contributing to canopy gains through time would certainly drive our reported dynamics in a different direction and could be catastrophic for Baltimore's urban forest.

Canopy change and biodiversity trajectories are important to consider when approaching urban forest management, particularly as it relates to the intersections between urban trees and local residents. Many residential trees were either *Acer* species, which have highly allergenic pollen, or weedy non-native species that may not correspond with “cues to care”—landscape characteristics that elicit positive associations and behaviors for local nature (Nassauer, 1995). These ecosystem disservices dramatically reduce the well-being benefits of urban trees at the local scales where people live. Furthermore, sites dominated by a single species (or stem) are most vulnerable to losses. Indeed, forested sites (cluster 5) experienced losses in *Fraxinus* species from 2004–2014. This predates known *Fraxinus* street and park tree removal, which began pre-emptively to circumvent Emerald Ash Borer (EAB) infestations in 2015 (personal communication, City of Baltimore), but may reflect early EAB mortality in forested sites before EAB detection and subsequent management of City-owned trees. Furthermore, these specific losses and apparent inconsistencies with pest-driven tree mortality suggest the importance of understanding tree biodiversity at the landscape scale. Certain species and genera—notably *Acer*, *Ulmus*, *Fraxinus*, and *Fagus* which are all dominant genera in Baltimore—are more susceptible than others to current or predicted pests (Laćan and McBride, 2008), and pathogen risks are likely to increase under climate change (Tubby and Webber, 2010). Moving forward, it will be necessary to complement findings from retrospective approaches such as our linking canopy change to tree diversity with contemporary and future risks.

5. Conclusions

Our study illustrates how broad city-wide assessments of canopy cover can miss important, fine-scale differences in species composition and tree structure, and highlights important links between land use, canopy cover, and tree structure and diversity. Urban forests span public and private lands and have different functions and benefits at different spatial scales, and the human-derived land use classifications often used in assessments of the urban forest are in themselves insufficient to explain the complexity of urban forest dynamics. We found that residential areas dominated by non-native species exhibited canopy losses over time, regardless of the starting canopy conditions. Alternatively, residential and forested areas dominated by native species experienced canopy gains. Furthermore, we found that trajectories of canopy increases—regardless of starting conditions—were more likely to be on institutional or public land, underscoring the importance of collective urban forest engagement and management. Furthermore, a species-level analysis of plots across the city highlights the assets and vulnerabilities of Baltimore's urban forest, particularly in light of the risks of pest damage to the abundant beech trees. As cities warm in the face of climate change, understanding the patterns of canopy change and the links to tree structure and biodiversity is critical to help managers, urban planners, and other stakeholders care for the urban forest and the ecosystem services it provides.

Author statement

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Declaration of Competing Interest

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

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:<https://doi.org/10.1016/j.ufug.2021.127365>.

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