RESEARCH ARTICLE

Divergence in diversity-area relationships between lawn and non-lawn plants in residential landscapes

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Abstract

Context In residential landscapes, people acknowledge and manage larger, conventionally attractive plants differently than smaller, less conspicuous ones, possibly leading to a divergence in basic ecological patterns such as diversity-area relationships.

Objectives We tested for divergences between lawn and non-lawn plants, and compared perceived and actual species richness in residential yards. We hypothesized that: (1) homeowner perception of plant species richness is more related to the actual species richness of non-lawn plants than lawn plants, and (2) alpha diversity will increase more rapidly, and beta diversity will decrease less rapidly for non-lawn plants than lawn plants with increasing spatial scale.

Methods We recorded all plant species in four 5×5 m plots in the front and back yards of 30 residences within four neighborhoods of Gainesville, Florida, United States (*N* = 238 plots). We also

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surveyed residents regarding perceived plant species richness in their yards.

Results We detected 386 non-lawn and 140 lawn species. Perceived plant species richness was positively related to non-lawn species richness and negatively related to lawn species richness, suggesting perception of species richness largely reflected that of non-lawn plants. Alpha diversity increased more rapidly, while beta diversity decreased less rapidly, for non-lawn plants than lawn plants with increasing spatial scale, although lawns had more species within individual yards.

Conclusions Findings revealed a divergence in diversity-area relationships between different plant groups in residential landscapes. We should consider the ecological implications of unrecognized plant diversity, particularly in residential lawns, where management often strives to limit plant species richness.

Introduction

Urbanization is increasing worldwide (Radwan et al. 2021), contributing to pervasive anthropogenic disruptions to ecological processes (Seto et al. 2012; Scheffers et al. 2016). Therefore, greater importance is now placed on urban green spaces to be repositories of biodiversity (Galluzzi et al. 2010). While urban areas may be perceived as "concrete jungles" (Lepczyk et al. 2017) or environmentally homogenous at larger spatial scales (Groffman et al. 2017), they include a diversity of ecosystem types, often with high plant species richness (Müller and Werner 2010; Lepczyk et al. 2017). These urban ecosystems include gardens, parks, street plantings, remnant natural areas, etc. (Lepczyk et al. 2017), and provide a suite of ecosystem services—one of many motivations for conserving urban biodiversity (Dearborn and Kark 2010). Nevertheless, not all urban ecosystems possess equal ecological value (Lepczyk et al. 2017).

Residential landscapes, in particular, encompass relatively large areas with many micro-habitats providing potential refuge to plant and animal species (Müller and Werner 2010). Within these landscapes, plants are the taxonomic group most directly manipulated by humans (Faeth et al. 2011). Plants, as an important component of urban biodiversity, also support higher trophic levels (Barthlott et al. 1996), thus, the human impact on plants in residential landscapes may affect other taxa and broader ecological processes. The importance of residential plant communities is exemplified by the suite of benefits plants provide to residents, including cooling (Park et al. 2021), food resources (Diekmann et al. 2020), and contributions to human mental health and general well-being (Bratman et al. 2019).

While drivers like climate, geology, and vegetation history are known to affect the biodiversity of natural ecosystems (Barthlott et al. 1996), these variables may have less of an effect in urban ecosystems (Loram et al. 2008). Urban plant diversity may be influenced by socioecological dynamics such as value-based judgments, household demographics, local parcel environmental characteristics (Chowdhury et al. 2011), policies, municipal codes, and homeowner's association (HOA) regulations (Burr et al. 2018), and perception of landscape aesthetics, e.g., *cues to care* (Nassauer 1995). Aesthetic preferences directly link humans to ecological processes, and thus the desire to see and live in places perceived as aesthetically pleasing may drive land-use change (Gobster et al. 2007). In fact, aesthetics is residents preferred trait for outdoor plants, ranking higher than traits such as ease of maintenance, presence of fruit, and tolerance to climate (Kaya et al. 2018). While we know that aesthetics is important for landscape design (Nassauer 1995) and plant selection (Gobster et al. 2007), the degree to which these plant preferences affect patterns of plant diversity and composition are less understood (e.g., Kendal et al. 2012; Minor et al. 2016; Belaire et al. 2016; Hostetler 2021).

The plants that people acknowledge and value in landscapes tend to be larger, conventionally aesthetic,

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ornamental species (Schroeder and Cannon 1983; Hardy et al. 2000; Gobster et al. 2007). People are attracted to plants that exhibit high resource availability with traits such as large flowers and large leaves (Kendal et al. 2012) more so than gardens with smaller flowers which are often perceived as "weeds" (Nassauer 1995). These aesthetic preferences and design choices often occur without consideration for ecological function or services (Gobster et al. 2007), therefore a large proportion of urban plant diversity with important biophysical functional roles (Milesi et al. 2005; Ignatieva et al. 2020) may be overlooked. The broader concept of plant blindness, as defined by Wandersee and Schussler (2001) is the inability to see or notice plants, or to overlook their importance or aesthetic features, especially compared to animals. Based on that definition, we refer to the biased attention and value attributed to some plants based on traits such as size, color, perceived aesthetic, and utility as *selective plant blindness* (Nassauer 1995; Gobster et al. 2007). If selective plant blindness contributes to peoples' disproportionate attention to ornamental plant species due to particular aesthetics (Nassauer 1995; Gobster et al. 2007), one might predict human perception of plant species richness to be more strongly related to the actual species richness of larger, more noticeable, or ornamental, plants than to less conspicuous plants in the landscape.

Preference for larger landscape plant species relative to less-conspicuous species, and differences in management, may affect patterns of plant biodiversity in residential landscapes. Greater species richness with increasing spatial scale or sample area is a long-understood concept in ecology (Wilson and Shmida 1984); however, these relationships may vary due to human influence on plants in urban areas. For instance, there is a known divergence in species-area relationships for different plant growth forms in residential landscapes (Su et al. 2021). Given the unpredictable effect that humans can have on both alpha diversity (average species richness for a given site) and beta diversity (species compositional dissimilarity among sites) (Socolar et al. 2016), the effects of human manipulation of plant composition on plant diversity-area relationships for both diversity metrics should be tested.

Yards, defined as all vegetated parts of a residential property or parcel (sensu Blanchette et al. 2021), are a good model ecosystem for detecting potential divergences in plant diversity-area relationships in residential landscapes. Homeowners may apply different maintenance practices to certain plant groups in the landscape. Cultivated, ornamental plant species often receive supplemental resources like water and fertilization, while spontaneous, self-recruiting plant species are often manipulated with practices like mowing and weeding (Pearse et al. 2018). In addition, homeowners often prefer variety in ornamental plant traits (Blanchette et al. 2021), facilitating unique plant combinations among yards. Simultaneously, homeowners often desire monoculture turfgrass lawns (Blanchette et al. 2021), and self-recruiting lawn species are found to be homogenous across large spatial scales (Wheeler et al. 2017). Lawns make up 2% of total U.S. land cover (Milesi et al. 2005) and their contribution to urban plant diversity needs further consideration. Potential differences in how *lawn plants* (turfgrasses and other small, often self-recruiting and mowed species) and *non-lawn plants* (larger, often ornamental landscaping plants, trees, shrubs, and herbaceous species) are perceived and managed make these plant groups ideal for observing a potential divergence in diversity-area relationships.

The objective of our study was to determine if lawn and non-lawn plants contribute to divergences in fundamental diversity-area relationships among residential yards. Because homeowners are more aware of and connected to larger, conventionally aesthetic plants than smaller, less-conspicuous plants, we hypothesized that: (H1) perceived plant species richness is more strongly related to actual species richness of non-lawn plants than lawn plants (Fig. 1A); Additionally, due to variation in ornamental plant choice and homogenization between lawns, we hypothesized that: (H2) alpha diversity will increase more rapidly, and beta diversity will decrease less rapidly, for non-lawn versus lawn plants with increasing spatial scale (Fig. 1B, C; respectively). In addition to testing H1 and H2, we noted the native, non-native, or invasive status of the plants detected, and curated a species list due to increased interest in residential landscaping plant species origin (Salisbury et al. 2017), the influence of plant origin on the ecological value of urban landscapes (Chong et al. 2014), and contributions of the ornamental plant trade to biological invasions (Bradley et al. 2012).
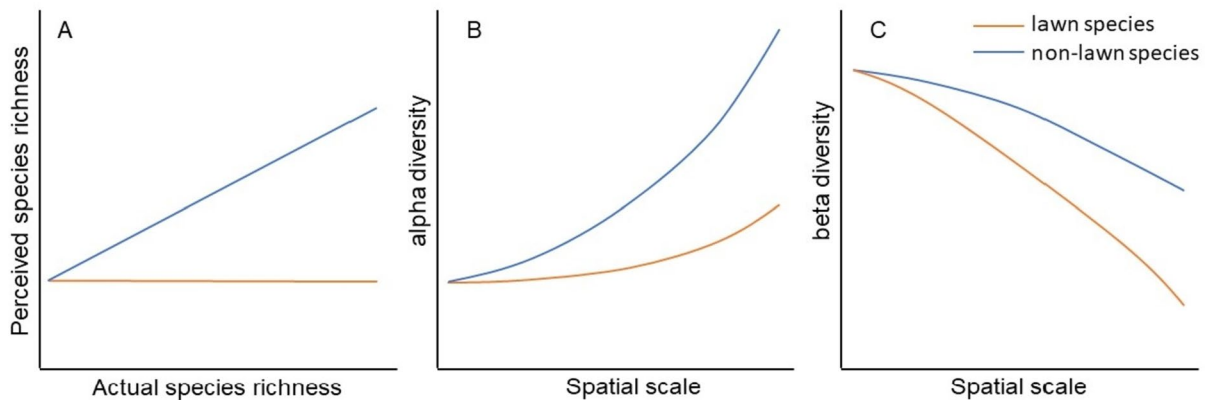
Fig. 1 Conceptualization of hypotheses. **A** H1 states that perceived species richness would be more strongly and positively related to non-lawn than lawn plant species, **B** H2 states that alpha diversity will increase more rapidly (we did not expect alpha diversity to saturate at the spatial scales of our study; areas larger than neighborhoods would likely need sampling to detect such saturation) and **C** beta diversity will decrease less rapidly for non-lawn than lawn plants with increasing spatial scale

Materials and methods

Study area

Gainesville is a mid-sized city in Alachua County, North-Central Florida, United States (U.S.), with a population of 141,085 people in 2020 (57.5% White, 20.6% Black or African American, 7.8% Asian, 0.3% Native American, 0.04% Native Hawaiian or Pacific Islander, 3.7% other race, 10% two or more races, 13.8% Hispanic or Latino - of any race; (U.S. Census Bureau 2020). Gainesville has a subtropical climate and is in USDA Hardiness Zone 9a (USDA NRCS 2012). The average high summer and winter temperatures are 32 °C and 20 °C, respectively. The average low summer and winter temperatures are 21 and 6 °C, respectively (U.S. Climate Data 2021). Gainesville's dominant soils consist of sand, loamy sand, and fine sand, and range from moderately well drained to poorly drained (USDA NRCS 2019). Historically, Gainesville's plant communities included pine flatwoods, sandhills, upland mesic hardwood forests, and freshwater marshes (Davis 1967).
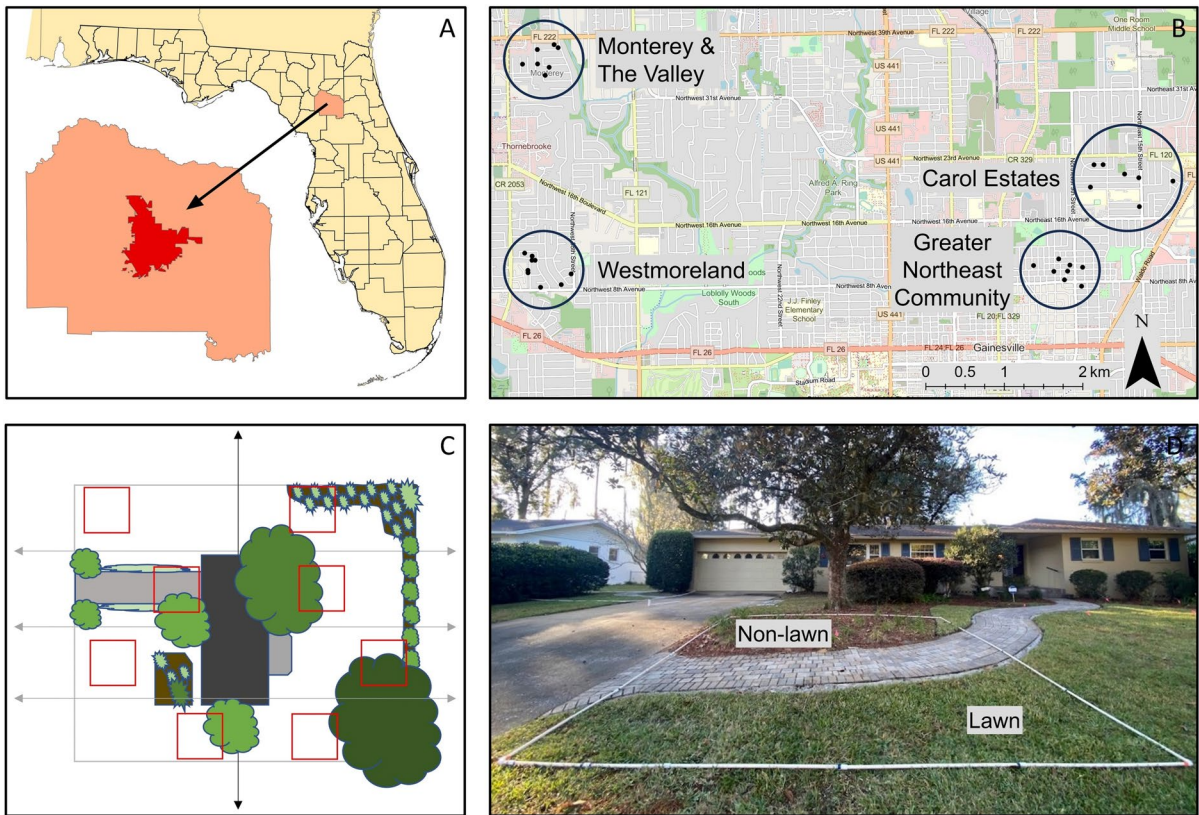
Sampling design and neighborhood selection

We sampled plant communities using a nested, hierarchical sampling design, enabling us to estimate alpha and beta plant diversity at four distinct spatial scales: plot, half-yard (front and back yard), whole-yard, and neighborhood (Fig. 2). We used a series of criteria to select neighborhoods for sampling that exhibited variation in landscaping irrigation and property value among households. First, we collected appraised property values and irrigation estimates for 2019 from UF/IFAS H$_2$OSAV: Water Savings, Analytics, & Verification Program (UF/IFAS PREC 2020). We then visited neighborhoods to visually confirm variation among yards in landscaping styles (e.g., plant choice, lawn height) to limit potential homogenizing effects caused by community norms (Minor et al. 2016), and to ensure representation of variation in landscaping styles. The four selected neighborhoods included: Monterey & The Valley (considered one neighborhood due to geographic proximity and interconnectedness), Westmoreland, Carol Estates, and Greater Northeast Community, none of which were regulated by HOAs.

We then screened for the following criteria: (1) homes built before 1990 to ensure that among-yard variation in vegetation composition and structure was due to management decisions rather than differences between older and newer landscaping; (2) resident-owned homes, as owners likely are the decision-makers regarding their yards (Hale and Morzillo 2020); (3) homes without swimming pools, which replace a large proportion of potentially vegetated area in yards; (4) properties large enough to fit four 25m^2 plots in both the front

Fig. 2 Nested sampling design. **A** The study location; Gainesville, Alachua County, Florida, USA. **B** Four neighborhoods containing 30 total sampled yards. **C** Four plots were sampled in both front and back yards (*n*=238). **D** A 5×5 m plot used to sample lawn and non-lawn plant species

and back yards; and (5) properties not adjacent to areas of unmanaged vegetation (e.g., wooded lots, stormwater ponds, nature preserves) to control for potential effects on yard plant communities (Hobbs 1988). No selected yards were closer than ~ 1.5 km to a natural area.

Of the 1282 properties in the four neighborhoods, 629 met these criteria. We mailed the qualifying homeowners an invitation to participate in an online survey. From respondents, 30 yards were selected for sampling across gradients of irrigation usage, as determined via UF/IFAS PREC (2020), and management intensity (i.e., complexity and frequency of landscaping practices like mowing and fertilizing) as self-reported by participants. We visited each yard to visually confirm variability in plant community composition.

Perceived plant species richness

To measure perceived plant species richness, we asked the following question in our online survey: "About how many different types of plants do you have growing in your FRONT/BACK yard (be sure to include the weeds as well)? Guess as best as you can if you do not know." Respondents could choose one of the five following ordinal classes, "< 5, 5–10, 10–20, 20–40, > 40 species" to prevent unquantifiable responses (e.g., "too many to count"). We asked for front and back yard estimates separately due to potential differences in human interaction with each (Locke et al. 2018).

Vegetation sampling

To evaluate actual plant species richness, alpha diversity, and beta diversity, we recorded all plant species in all plots. Sampling occurred in September-November 2020, when plants still retained foliage and were identifiable. Front and back yards were delineated by any existing structures (e.g., fences) or were estimated using the lateral midpoint of the house. Two of the four plots' edges were placed 15 cm away from the property edges to capture potential edge effects in these commonly landscaped areas. The other two were placed at 3/8 and 5/8 the width of the yard. All plots were placed random distances from the house using a random number generator to determine the percent length of the yard where the plot center would be placed (Fig. 2C). By randomly placing plots in four separate sections of the front and back yards, our sampling design helped to ensure that we captured the proportionality of different yard cover types (e.g., lawn, non-lawn, paved areas). Plots were constructed as 5×5 m squares, except for 14 plots shaped as 2.5×10 m rectangles when faced with yard shape constraints (e.g., longer, more narrow yard halves). One property only had space for two plots in the back yard, therefore, across 30 properties 238 total plots were sampled (4 plots/yard half $\times$ 2 yard halves $\times$ 30 properties = 240–2). In each plot, all species taller than 0.5 m were classified as "non-lawn" species. Plants shorter than 0.5 m were then classified as either "lawn" (grasses and small, self-recruiting plants, which are usually mowed) or "non-lawn" species (typically ornamental landscaping plants, herbaceous plants, shrubs, etc.) based on functionality. Species growing less than 0.5 m that could not be defined as exclusively lawn or non-lawn plants (e.g., *Lantana camara* is a landscaping plant that also survives in mowed lawns), were placed in their own category, "either-or" species.

We used Wunderlin and Hansen (2011) as our primary taxonomic authority for native and naturalized species, and several secondary references used to identify less-common, typically ornamental species (USDA NRCS 2023; WFO 2023; Wunderlin et al. 2023). The UF/IFAS Herbarium Plant Identification and Information Service also provided assistance. When identification to species level was not possible, specimens from the same genus or family were lumped into a single taxon

(e.g., *Citrus* spp. and Bromeliaceae, respectively). Follow-up censuses of the entire yard were conducted to document species not in plots, which were excluded from analyses but are included in our list of total observed species (Supplementary Table A2). References used to classify plant species as native, non-native, or invasive (Iannone et al. 2020) include FISC (2019), UF/IFAS (2023), and Wunderlin et al. (2023).

Statistical analyses

Perceived versus actual plant species richness Perceived plant species richness was modeled in response to actual species richness using cumulative link mixed models (ordinal regression) with the ordinal package (Christensen 2019) in R v.4.1.2 (R Core Team 2022). Front and back yards were included in the model separately due to "yard-half" being our observational unit for analysis ($N = 60$). Using a logistic link function (Agresti 2013), these models related ordinal survey answers that vary in range, such as our survey answers (e.g., 5–10 vs. 10–20), to actual species richness of lawn and non-lawn plants in front and back yards (Agresti 2010). These models yield odds of going from one ordinal class to the next higher class per unit of explanatory variable. Odds ratios higher than 1 show a direct, positive relationship, while odds ratios lower than 1 show a negative relationship where the odds of going from one ordinal class to the next lowest is the inverse of the odds value (1/x). Each individual yard was incorporated into the model as a random effect to account for potential spatial autocorrelation in plant community composition and landscaping practices. We estimated goodness of fit as Somers' Delta (Somers' D) (Somers 1962; Agresti and Tarantola 2018) using the DescTools package (Signorell et al. 2021). This parameter varies between -1 and 1 (perfect negative and positive relationship, respectively) and is accompanied by a 95% CI (Newson 2006). These models were fitted using Restricted Maximum Likelihood (Corbeil and Searle 1976). Initial models contained neighborhood as a random effect and yard half as a fixed effect. However, we removed these terms from models, as AIC values revealed no benefit of their inclusion (ΔAIC = 2–18), and yard half was not statistically significant ($p = 0.94$).

Differences in diversity To determine if lawn and non-lawn plants differed in diversity-area relationships, we estimated alpha and beta diversity for lawn and non-lawn plants at the plot, half-yard (front/back yard), whole-yard, and neighborhood scales. Species community matrices were constructed for each nested spatial scale using the BiodiversityR package (Kindt and Coe 2005). Alpha diversity was estimated as the mean species richness found across all sample units of a given scale. We then graphed this value (mean ± SE) at each spatial scale and used a paired *t*-test (assuming unequal variance) to test for differences between lawn and non-lawn plants within each sample unit (Welch 1938).

We estimated beta diversity as Simpson-based multiple-site dissimilarity (Eq. 1; hereafter β_{SIM}), as proposed by Baselga (2010), from species absence or presence in species community matrices of each spatial scale using the betapart package (Baselga et al. 2021).

$$\beta_{SIM} = \frac{\left[\sum_{i<j} \min\left(b_{ij}, b_{ji}\right)\right]}{\left[\sum_i (S_i - S_t)\right] + \left[\sum_{i<j} \min\left(b_{ij}, b_{ji}\right)\right]}, \quad (1)$$

where S_i is the total number of species in site i, S_t is the total number of species across all sites, and b_{ij} and b_{ji} are the number of species exclusive to sites i and j, respectively. We used this index because it is independent from alpha diversity and it varies across scales (Baselga 2010).

Independent of alpha diversity, β_{SIM} is influenced by species pool size; there is no consensus on best approach control for this influence (Ulrich et al. 2017, 2018). Thus, we utilized a simpler approach that employs bootstrapping. We first estimated β_{SIM} for lawn and non-lawn species across each spatial scale. To confirm these differences are not driven by differences in species pool size, we constrained species pool size of non-lawn species to the lower species pool size of lawn species, and randomly selected that number of species to estimate a new value of β_{SIM} from the resulting species community matrix. We repeated this procedure 1,000 times for each spatial scale and graphed our results, connecting lines across the scales of the same iteration. We then estimated the mean and 95% CI of the bootstrapped values. To confirm differences, we then calculated the cumulative probability that the actual β_{SIM} value of lawn plants would belong to the resulting bootstrapped distribution of non-lawn β_{SIM} values, assuming a normal distribution.

We tested for differences in diversity, both including and excluding species that could not be distinguished as lawn or non-lawn species from estimates for each species group. We also tested for the sensitivity of our bootstrap analysis to species pool size by repeating the analysis with constraints of 20, 50, and 100 species. All analyses revealed similar patterns; thus, we only present those including the "either-or" species and constraining bootstrapping to lawn plus "either-or" species.

Results

Survey response rates and socioeconomic characteristics

Of the 629 invited homeowners, 113 responded to the survey (~18% response rate). Further inspection of respondents' yards to confirm screening criteria resulted in 106 valid participants, of which 30 were sampled across four neighborhoods varying in average property value, house size, parcel size, and irrigation (Table 1).

Table 1 Summary statistics for the sampled properties within the four neighborhoods in our study where property value, house size (not footprint size), and parcel (lot) sizes are reported in mean (± SD), while monthly irrigation is reported in median (min., max)

Neighborhood	Yards sampled	Property value (US$)[a]	House size (m²)	Parcel size (ha)	Irrigation (kL/month)[a]
Monterey & The Valley	6	188,774 (± 39,940)	196 (± 37)	0.14 (± 0.02)	5.8 (0.0, 43.3)
Westmoreland	9	136,223 (± 15,896)	153 (± 15)	0.13 (± 0.02)	0.0 (0.0, 38.1)
Greater Northeast Community	8	150,728 (± 41,492)	117 (± 30)	0.09 (± 0.01)	0.0 (0.0, 1.9)
Carol Estates	7	103,059 (± 15,030)	125 (± 29)	0.09 (± 0.01)	0.0 (0.0, 5.6)

[a]Estimates from 2019

Overall plant biodiversity

We detected 500 plant species across all plots: 360 non-lawn species, 104 lawn species, and 36 "either-or" species that could have occurred in both lawn and non-lawn areas (Supplementary Table A1, A2); all 36 occurred in lawns, while only 26 occurred in non-lawn areas. Yard management practices (e.g., mowing) also prevented identification of 42 individual specimens, which were excluded from analysis. Of the 360 non-lawn species, 116 are native to Florida, 241 are non-native to Florida (28 of which are invasive), and three have unknown origins. Of the 104 lawn species, 54 are native to Florida, and 50 are non-native to Florida (four of which are invasive). Of the 36 "either-or" species, 23 are native to Florida, 12 are non-native to Florida (of which six are invasive), and one has unknown origins. Follow-up yard censuses revealed 173 additional species: 159 non-lawn species, 13 lawn species, and one "either-or" species (Supplementary Table A1, A2).

Perceived vs. actual species richness

Perceived plant species richness (Table 2) was positively correlated to non-lawn species richness, but negatively correlated to lawn species richness (Fig. 3A, B). The odd ratios from the cumulative link mixed models revealed that as actual non-lawn species richness increased by 1, survey respondents were 1.17 times more likely to report one ordinal category greater for perceived species richness (Table 3). In contrast, as actual lawn species richness increased by 1, survey respondents were similarly likely [1.12 times (1/0.89)] to report one ordinal category lower for the perceived species richness, i.e., the relationships were inverted (Table 3). Therefore, homeowners that perceived more species in their yards usually had more non-lawn plants and fewer lawn plants. The Somers' Delta shows that actual non-lawn and lawn species richness were 53% and 37% in concordance with ordinal intervals of perceived species richness, respectively (Table 3).

Patterns of alpha diversity

As hypothesized, alpha diversity increased more rapidly in relation to spatial scale for non-lawn plants than for lawn plants, although non-lawn alpha diversity was not greater across all spatial scales (Fig. 4). At the plot and half-yard scale, alpha diversity was 102% and 31% greater, respectively, for lawn plants than non-lawn plants (Fig. 4; plot level: 19.06 ± 0.43 for lawns vs. 9.43 ± 0.34 for non-lawns, $t = 18.56$, $df = 237$, $p < 0.0001$; half-yard scale: 35.55 ± 1.09 vs. 27.12 ± 1.31, $t = 5.55$, $df = 59$, $p < 0.0001$). Alpha diversity did not differ between lawn and non-lawn species at the full-yard scale

Table 2 Minimum, maximum, mode, median, and mean ($\pm$ SE) for perceived and actual plant species richness

|	Yard-half	Min., Max.	Mode	Median	Mean ($\pm$ SE)
Perceived species richness[a]	Front	<5, >40	10–20	10–20	–
	Back	<5, >40	20–40	20–40	–
Actual species richness (lawn)	Front	15, 50	–	38	37.7 ($\pm$ 1.5)
	Back	12, 47	–	33.5	33.4 ($\pm$ 1.5)
Actual species richness (non-lawn)	Front	3, 37	–	24.5	24.7 ($\pm$ 1.5)
	Back	8, 50	–	27.5	29.6 ($\pm$ 2.1)

[a]Perceived species richness ordinal categories included <5, 5–10, 10–20, 20–40, and >40

Table 3 Results of cumulative link mixed models relating perceived to actual plant species richness of non-lawn and lawn plants

Model[a]	Coefficient ($\pm$ SE)	Odds ratio	p-value	Somers' D (CI 95%)
Non-lawn species richness	0.157 ($\pm$ 0.051)	1.17	0.002	0.53 (0.29–0.76)
Lawn species richness	$-$ 0.114 ($\pm$ 0.053)	0.89	0.033	0.37 (0.09–0.58)

[a]Non-lawn and lawn species richness were treated as fixed effects, while the sampled lot was included as a random effect
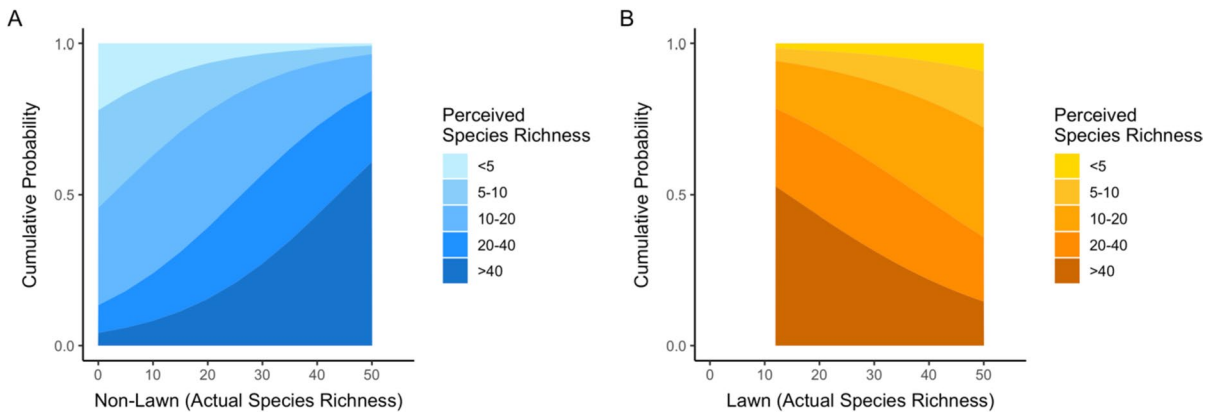
Fig. 3 Relationships between perceived and actual plant species richness revealed by cumulative link mixed models for **A** non-lawn and **B** lawn plants. Figures show the cumulative probability of being within a given ordinal category for perceived species richness (y-axis) for a given value of actual species richness (x-axis). Differences in x-axis ranges are based on min-max species richness values between non-lawn and lawn species

Fig. 4 Relationship between lawn and non-lawn alpha diversity (mean ± SE) across the plot (*n* = 238), half-yard (*n* = 60), whole-yard (*n* = 30), and neighborhood (*n* = 4) scales. The x-axis shows the approximate size of plots, half-yards, whole-yards, and neighborhoods on the log scale. * = denotes statistically significant difference. All p values for statistically significant differences were ≤ 0.02
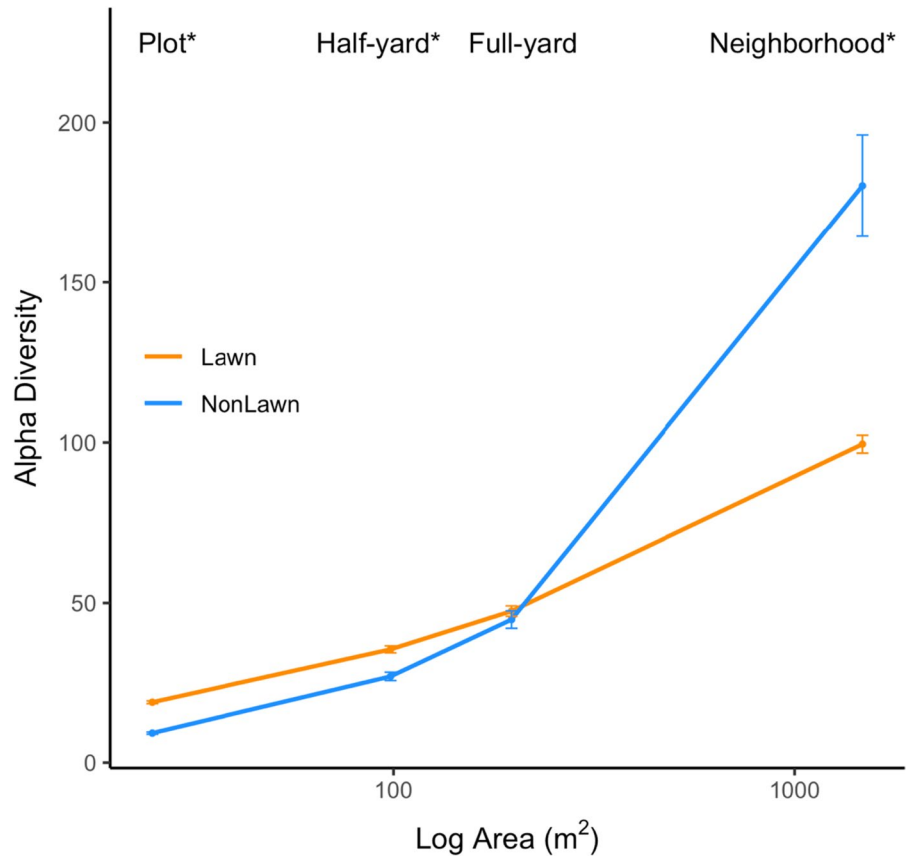
Fig. 5 Relationship between lawn and non-lawn beta diversity across plot ($n=238$), half-yard ($n=60$), whole-yard ($n=30$), and neighborhood ($n=4$) scales. The x-axis shows the approximate size of each observational unit on the log scale. Shown are actual values of beta diversity for lawn plant species (orange), actual values of beta diversity for non-lawn plant species, i.e., not controlling for differences in species pool size (dark blue), the mean $\pm$ 95% CI of bootstrapped beta diversity estimates controlling for species pool size at the 140 species level (mid-blue), and lines connecting beta diversity values of each bootstrap iteration (light blue). Bootstrapped and actual means were quite similar making it difficult to see both in the figure. * = denotes statistically significant differences at the $p<0.0001$ level
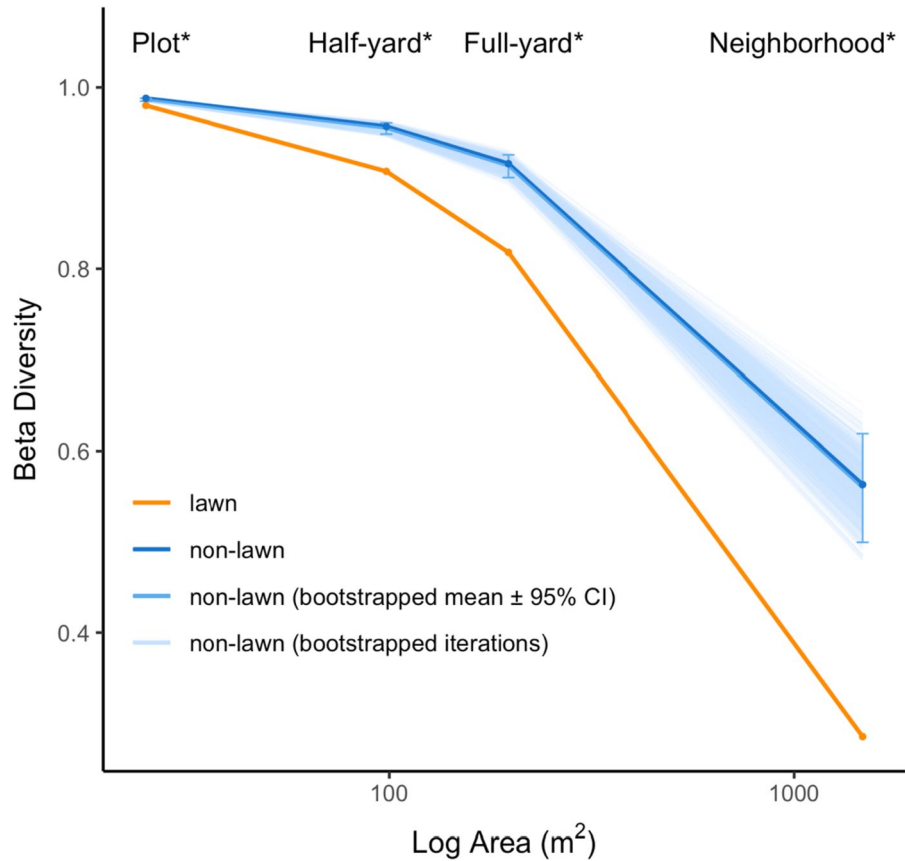
(t$=0.97$, df$=29$, $p=0.34$). At the neighborhood scale, patterns reversed with alpha diversity 81% higher for non-lawn than lawn species (Figs. 4 and 180.25 ± 15.86 vs. 99.50 ± 2.78, t $=-5.35$, df$=3$, $p=0.013$). Overall, there were more non-lawn species detected in the study area, and the additive number of species increased more slowly for lawns than non-lawn areas with increasing spatial scale.

Patterns of beta diversity

As hypothesized, non-lawn beta diversity was greater across all spatial scales, and declined more slowly, relative to lawn beta diversity (Fig. 5). The cumulative probability of the actual lawn beta diversity value being within the distributions of bootstrapped beta diversity values for non-lawn plants (constrained to 140 species) was extremely low across all spatial scales ($p<0.0001$; Fig. 5). The more rapid decline in beta diversity of lawn species compared

to non-lawn species caused differences between lawn and non-lawn species to increase from 1% at the plot level (0.98 vs. 0.99, respectively) to 97% at the neighborhood level (0.29 vs. 0.56, respectively). Thus, lawn species are more similar than non-lawn species with increasing spatial scale.

Discussion

Given the increasingly recognized importance of urban landscapes for supporting biodiversity (Aronson et al. 2017; Lepczyk et al. 2017), and the many ways that humans shape this composition (e.g., Nassauer 1995; Gobster et al. 2007; Leong et al. 2018), it is important to consider these effects on basic ecological patterns and processes of urban and residential landscapes. By quantifying human perception of plant species richness and the actual patterns of plant diversity across spatial scales, our research elucidates these effects on basic

diversity-area relationships. Homeowner perception of plant species richness in yards reflected actual species richness of non-lawn plants more than that of lawn plants, providing evidence that humans tend to acknowledge larger, conventionally attractive plants relative to smaller, less-conspicuous plants (Nassauer 1995; Hardy et al. 2000; Gobster et al. 2007). Additionally, non-lawn alpha diversity increased more rapidly, and non-lawn beta diversity decreased less rapidly compared to lawn plant species with increasing spatial scale. These findings reveal how variation in human perception, preference, and management of different plant groups affect the patterns of biodiversity within urban and residential landscapes.

Overall plant biodiversity trends

We found high levels of plant species richness, much of which is attributed to non-lawn plants. Including "either-or" species, we detected 386 plant species in non-lawn areas and 140 plant species in lawns. In the full yard census, 173 additional species were detected, most of which were non-lawn, unique ornamental species, and 13 were lawn species. Similar levels (774 species) were found in a recent study of 34 Alachua County, Florida yards (Braatz et al. 2021), suggesting that Florida yards may harbor high levels of plant biodiversity. Given our conservative approach of grouping multiple plants into one taxa when identification to species was not possible, yard plant species richness may be even higher than reported.

Overall levels of plant biodiversity detected also yielded insights regarding the proportion of native and non-native plant species. In our plots alone, non-native and invasive species represented 61% and 8% of those identified, respectively. The non-lawn species pool had a greater percentage of confirmed non-native species (64%) than the lawn species pool (44%). The high proportion of non-native plant species found, for non-lawn plants in particular, may reflect the contributions of the ornamental plant trade and urbanization to plant introductions and potential invasions (Reichard and White 2001; van Kleunen et al. 2018). Despite the proportion of non-native plants, we also found many native species (39%), especially in lawns

(55%), reiterating the potential for residential landscapes to support, and be managed for, native biodiversity (Ives et al. 2016; Aronson et al. 2017; Lepczyk et al. 2017).

Perceived vs. actual species richness

Homeowner perception of plant species richness was positively related to actual non-lawn species richness and negatively related to actual lawn species richness, revealing that homeowners were less likely to recognize lawn species. Selective plant blindness may explain the positive relationship regarding perceived species richness and non-lawn plants (Nassauer 1995; Hardy et al. 2000; Gobster et al. 2007). People may be more connected to the non-lawn, ornamental plants, particularly if those plants were selected by the resident. People may also be more likely to recognize and name the plant species marketed as retail products and commonly discussed, as opposed to many self-recruiting lawn species.

Homeowners with lower species richness in their lawns perceiving a greater number of species was unexpected. While selective plant blindness could explain a weak positive or absent relationship, the negative relationship is less clear. Differences in how homeowners and scientists perceive species may play a role, considering that homeowners may be more perceptible to variety in plant traits (e.g., color, size, and texture) rather than taxonomic distinctions (Blanchette et al. 2021). Therefore, biodiverse lawns may simply appear homogeneous to homeowners. We also considered if homeowners with higher non-lawn plant species richness have lower lawn species richness, perhaps due to maintaining a more conventionally manicured landscape; however, we found no evidence of this, as follow-up analyses treating yards as a random effect found no relationship between actual lawn and non-lawn species richness ($p = 0.25$). Ultimately, homeowners regardless of their perceived species richness may have had similar levels of actual plant species richness due to overlooked species in their lawns.

Patterns of alpha and beta diversity

As predicted, non-lawn plant alpha diversity increased more rapidly than that of lawn plants with increasing spatial scale. Since most census species detected

were non-lawn, incorporating these species into our plot-based analyses would have only intensified the rate by which non-lawn alpha diversity increased relative to scale. Greater lawn than non-lawn plant alpha diversity at smaller spatial scales (plot and half-yard) was unexpected given the commonly assumed desire for monoculture lawns (Blanchette et al. 2021); however, that desire was not noted in our study. This finding could be partially due to more small-statured lawn plants being able to fit in a plot than larger non-lawn species can. Additionally, given the link between lawn management intensity and decreased plant diversity (Chollet et al. 2018), and the adoption of such lawn management approaches, policies, and norms (Sisser et al. 2016), particularly in newer residential developments of Florida, our alpha diversity results may be more representative of mature yards with self-recruiting species rather than recently installed, highly managed residential landscapes. Regardless our findings show that, at least in some residential areas, lawn plants contribute more to diversity at smaller scales, while non-lawn plants contribute more to diversity at larger scales (e.g., neighborhood).

Beta diversity of non-lawn plants remained greater than that of lawn plants at all spatial scales, as predicted, and also would be intensified with the inclusion of census species. The divergence of beta diversity was robust even when controlling for differences in lawn and non-lawn plant species pool size (Ulrich et al. 2017, 2018). The greater dissimilarity of non-lawn plants may reflect unique homeowner choices in landscaping plants. Additionally, the more rapid decline in lawn plant beta diversity may reflect that lawn species are more influenced and limited by natural processes of dispersal rather than homeowner species preferences. More similar lawn species composition may also reflect difference lengths of time in the species pool (Lososová et al. 2012), as many self-recruiting lawn species could have been present in the historic species pool longer than many ornamental, non-lawn species (van Kleunen et al. 2018).

Beta diversity of non-lawn plants did decline more rapidly with increasing spatial scale than we expected considering homeowners in our sampled communities have freedom of plant selection. While environmental factors like climate could limit regional urban plant diversity (Pearse et al. 2018), human influence (e.g.,

landscape plant selection) may be more important at the spatial scale of our investigation. For instance, limited commercially available plant palettes shown to affect urban plant diversity elsewhere (e.g., Avolio et al. 2018; Cavender-Bares et al. 2020) may be applicable to our study area, influencing the rate at which alpha diversity plateaus and beta diversity declines across spatial scales.

Divergence in diversity-area relationships

Differences in diversity-area relationships between lawn and non-lawn plants show the effects that human perception and management can have on patterns of urban plant diversity. The inadequacy of a single curve to describe urban diversity-area relationships is supported by different curves being detected among varying plant growth forms (Su et al. 2021). Future studies should consider how human preference and management for other specific plant traits (e.g. flowering habit) further affect diversity-area relationships. Additionally, the influence of landscaping codes (e.g., HOAs) and societal norms may have compounding effects on diversity-area relationships given their influence on plant community structure and composition (Burr et al. 2018). We may also benefit from determining how diversity-area relationships vary among urban green space types other than yards, particularly for urban plant communities less influenced by human preference (e.g., stormwater ponds, remnant natural areas, vacant lots).

Study limitations and further considerations

Future research could address some characteristics of urban areas not captured by our study. Follow-up inspection of the 30 homeowners' demographics revealed that most were white (27/30) with a college education (27/30), and over half (17/30) had incomes of over $US 50,000. Our sampling pool was limited to the homeowners who responded to our survey and is not representative of our study area's ethnic, cultural, and economic diversity. Therefore, future investigations are needed to confirm if divergence in species-area relationships of lawn and non-lawn plants also occurs across the many socioeconomic factors that affect urban biodiversity (Kinzig et al.

2005). Future studies might explore plant species diversity-area relationships across different residential landscapes, homeowner lifestyles, and economic gradients, including in residences belonging to HOA's, and in rental properties.

There are other factors to consider in future research, for instance, increasing the geographic scope of this study may allow for detection of geographic variability in diversity-area relationships, such as is found for different plant lineages (Patiño et al. 2014) and for the relationships between wealth and plant diversity (Leong et al. 2018). We also unexpectedly found no evidence of yard half affecting the relationships between perceived and actual plant species richness, despite known differences in plant diversity and management between front and back yards, i.e., *the mullet effect* (Locke et al. 2018); This lack of difference could be further investigated. The effects of management intensity on diversity-area relationships should also be considered in future research since it affects overall yard diversity (Chollet et al. 2018). Finally, given the overall high number of species found in yards, future surveys of perceived species richness may benefit by providing ranges in survey answer choices beyond those used in our survey, and using ordinal categories with equal intervals (e.g., 10–19, 20–29, 30–39).

Conclusion

Our findings contribute to knowledge about the aesthetics-ecology relationship, i.e., the disconnect between ecological processes and aesthetic qualities in urban landscapes. We found evidence both of selective plant blindness and of a divergence in fundamental diversity-area relationships among plant groups which experience differences in perception and management. We also found high urban plant biodiversity and evidence that many plant species are overlooked, particularly in lawns which have been referenced as depauperate "biodiversity deserts" (Sturm and Frischie 2020; Kawahara et al. 2021). However, our results illustrate that lawns with low plant diversity (i.e., turfgrass monocultures) are not synonymous with lawns with a great diversity of plant species. Considering the implications of unrecognized plant diversity and of managing urban ecosystems to maintain low plant diversity, such as is done in newer residences, we could strive to enhance the conservation value of urban areas by increasing appreciation for the many already present, yet overlooked, plant species of residential landscapes.

Declarations

Conflict of interest All authors declare that we have no conflict of interests, be they financial or non-financial, that is relevant to this paper's content.

Ethical approval Survey approval under IRB # 202002223 Relating plant diversity, structure, and management to landscaping costs and environmental benefits: identifying suitable yards to sample.

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