



Environmental equity and urban afforestation in the extreme northeastern Brazilian Amazon

L. A. Jimenez¹ · S. M. Silvestre² · J. A. Aquino³ · L. M. Freire¹ · J. J. Toledo⁴

Accepted: 29 July 2024

© The Author(s), under exclusive licence to Springer Science+Business Media, LLC, part of Springer Nature 2024

Abstract

Urban areas with low tree cover in Brazil are found mainly in the Amazon, highlighting the disparities in access to benefits of urban greening. The goal of this study was to analyze how urban afforestation is associated with sociodemographic factors to understand the distribution of trees (and their benefits) among different social segments in the city of Macapá, in the extreme Northeastern Amazon. We conducted a floristic inventory in 217 blocks belonging to 32 neighborhoods, and classified species according to origin (exotic or native) and fruit use (fruit-bearing or non-fruit-bearing). Additionally, we selected 12 variables to represent sociodemographic gradients among neighborhoods. We reduced the dimensionality of floristic and sociodemographic data using ordination techniques, and used their first axes to represent gradients of species composition and sociodemographic factors. The effect of sociodemographic gradients on tree abundance, species richness and composition was tested using generalized linear models. The results indicate that most urban trees are exotic, non-fruit-bearing, and of medium size (10–30 cm in diameter). Older neighborhoods with higher human population age and income showed higher tree abundance and higher species richness. The species composition was related to the gradient of human population density, proportion of old people, income, and household occupancy. Large non-fruit trees were more frequent in neighborhoods with low population density. Exotic trees were more frequent in older neighborhoods with older people and high income, and large fruit trees showed higher abundance in neighborhoods with low household occupancy rates. Ensuring that low-income neighborhoods and historically marginalized communities have equal access to green spaces and adequate tree coverage is a key element to promote environmental justice and to develop healthier and more sustainable cities.

Keywords Amazon · Exotic species · Urban afforestation · Sociodemographic factors

✉ L. A. Jimenez
lorejimenez@gmail.com

S. M. Silvestre
saulomsilvestre@gmail.com

J. A. Aquino
jaa@ufc.br

L. M. Freire
laisapa@gmail.com

J. J. Toledo
jjuliotoledo@gmail.com

¹ Laboratório de Limnologia, Programa de Pós-Graduação em Ciências Ambientais e Conservação, Universidade Federal do Rio de Janeiro, Avenida São José do Barreto, 764, Macaé, Rio de Janeiro 27965-045, Brasil

² Instituto Chico Mendes de Conservação da Biodiversidade, Rua Hamilton Silva, 1570, Santa Rita, Macapá-Amapá 68.906-440, Brasil

³ Departamento de Ciências Sociais, Universidade Federal do Ceará, Av. da Universidade 2995, Benfica, Fortaleza-Ceará 60020-181, Brasil

⁴ Departamento de Meio Ambiente e Desenvolvimento, Universidade Federal do Amapá, Rod. Juscelino Kubitschek, Km 02, Jardim Marco Zero, Macapá-Amapá 68903-419, Brasil

Introduction

The distribution of tree cover in urban areas is related to sociodemographic factors such as income, neighborhood age, population density, household occupancy rate, and proportion of young people (e.g., dos Santos et al. 2010; Fan et al. 2019; Useni et al. 2021; Bille et al. 2023; Fang et al. 2023; Lin and Guneralp 2024). These relationships have shown how these factors can shape the diversity and distribution of tree species in urban environments. For example, in the USA, urban tree cover was negatively correlated with population density (Nowak and Greenfield 2012), and in Australia, trees in areas with low household occupancy rates were more likely to survive (Ren et al. 2023). Additionally, tree cover is related to socioeconomic factors. Urban areas predominantly occupied by low-income communities and ethnic minorities often face inequalities in tree distribution compared to areas occupied by high-income communities and predominantly white populations (Heynen et al. 2006; Li and Weng 2007; Lin et al. 2021).

These relationships point to a lack of equity that can result in disparities in access to the benefits of urban greening, such as shading, high air quality, stress reduction, and enhanced quality of life. In this context of environmental inequalities, it is crucial to consider the needs of marginalized communities when planning urban greening (Heynen et al. 2006; Fleming and Steenberg 2023). For example, conditions of excessive heat, known as “urban heat islands,” are particularly harmful to low-income communities, which have fewer resources to cope with the impacts of heat (Roman et al. 2020). An adequate street greening in areas dominated by socioeconomically disadvantaged populations can help mitigate these inequalities and improve environmental and health conditions.

Urban afforestation in Brazil exhibits diverse characteristics due to the vast extent of the country and the different climatic regions and ecosystems (dos Santos et al. 2010; Álvarez et al. 2015; Silva et al. 2020; Soares et al. 2021). However, urban areas with less tree cover are located in the northern region of the country, in the Brazilian Amazon. In a study conducted with the 15 Brazilian cities with over one million inhabitants, the two largest capitals in the Brazilian Amazon region – Belém and Manaus – had the lowest percentage of tree cover in the country (IBGE 2010). However, methodologies for measuring vegetation cover vary (e.g., counting; satellite imagery), which can lead to discrepancies in the results (Rocha and Nucci 2018).

The lack of urban afforestation in tropical regions can have a significant negative impact on people’s well-being, as the presence of trees and vegetation in urban environments is associated with a range of benefits. For example, trees improve air quality (Beckett et al. 2000; Nowak et

al. 2014), store carbon (Nowak et al. 2014), mitigate noise (Mao et al. 1993; Gaudon et al. 2022), contribute to the conservation of biodiversity (Cornelis and Hermy 2004), and are an important source of food and health benefits (McLain et al. 2012; Poe et al. 2013). Additionally, urban afforestation promotes intangible benefits such as stress reduction and social connections (Arnberger and Eder 2012; Peckham et al. 2013; Vujcic and Tomicevic-Dubljevic 2018). In this sense, in tropical regions where high temperatures can be challenging, urban afforestation plays a crucial role in creating healthier and more enjoyable urban environments.

The state of Amapá is one of the nine states encompassing the Brazilian Amazon and stands out as the most environmentally preserved in Brazil, with over 70% of its territory consisting of protected areas (twelve conservation units and five indigenous lands) (CI-BRASIL 2009). Despite being the second least populous state in the country, with around 734 thousand inhabitants (IBGE 2022), Amapá has the fourth highest urban population density in the Northern region (5.15 inhabitants per square meter), with approximately 90% of the population living in cities and more than half in the capital, Macapá (IBGE 2022). Although Amapá has the highest density and richness of tree species in urban environments among the states of the Brazilian Amazon (Soares et al. 2021), its capital, Macapá, has a high deficit of tree cover, with an average of approximately one tree per 100 m of sidewalks – less than one -tenth of a tree (0.03) per inhabitant (LAJ, unpublished data). Although estimates are scarce and methods differ (Rocha and Nucci 2018), the number of trees per capita in Macapá is lower than that reported by Álvarez et al. (2015) for the city of Campinas (Brazil) (0.17–0.6 trees per inhabitant) and by McPherson et al. (2016) for California (0.26 trees per inhabitant).

In this context of climate change (IPCC 2023), with a trend of increasing average global temperatures and intensification of extreme weather events, the current situation of urban afforestation in Macapá is of high concern. Therefore, studies are needed to support the development of strategies to ensure socio-environmental sustainability in the city. In this sense, we conducted an inventory to determine the density, diversity, and composition of urban tree species. Further, using generalized linear models we determined the influence of sociodemographic variables on tree abundance, species diversity and composition to understand how characteristics of human community affect urban tree cover.

Materials and methods

Study area

The study area encompassed the urban area of Macapá city, the capital of the Amapá State, situated on the left bank of the Amazon River delta in the extreme northeast of the Brazilian Amazon. Macapá covers an area of 6,563.85 km² and has an estimated population of 522,357 inhabitants. The population density is 62.14 inhabitants per square kilometer (IBGE 2018). The region has an Equatorial climate with an annual precipitation of ~2600 mm, characterized by a rainy season from December to July and a dry season from August to November. Average minimum and maximum temperatures range between 22.9°C and 32.6°C, respectively, with an annual average of 27°C. The original vegetation types covering the area were dense floodplain forests, Amazonian savannas, and patches of semi-deciduous forest with a high diversity of tree species (Tavares 2004).

Sampling design

Sampling was conducted on the sidewalks surrounding the blocks of neighborhoods. Information about distribution and number of blocks per neighborhoods was obtained from the cartographic database of Amapá. The sample size was 10% of the blocks per neighborhood, totaling 217 blocks across thirty-two neighborhoods, ranging from 1 to 19 blocks per neighborhoods (6.7 ± 4.2 ; mean \pm standard deviation). The blocks were numbered, and a random selection was performed using the ‘sample’ function in R software, version 4.2.0 (R CORE TEAM 2022). This sample size (10%) is far above the recommendation (1-ha plot per 1000 ha or 0.1%) for floristic inventories in natural forests (Alder and Synnott 1992), which are more complex in structure and more diverse than urban forests. The perimeter of the blocks was measured using the Google Maps application with the ‘Measure Distance’ function, and it varied from 257.8 m to 692.5 m (446.1 ± 98.1 m). We summed the perimeters of the blocks within neighborhoods to provide the total perimeter per neighborhood, which ranged from 447.5 m to 10064.7 m (2997.5 ± 1972.2 m).

Sampling was conducted on the sidewalks surrounding the blocks of neighborhoods. Information about distribution and number of blocks per neighborhoods was obtained from the cartographic database of Amapá. The sample size was 10% of the blocks per neighborhood, totaling 217 blocks across thirty-two neighborhoods, ranging from 1 to 19 blocks per neighborhoods (6.7 ± 4.2 ; mean \pm standard deviation). The blocks were numbered, and a random selection was performed using the ‘sample’ function in R software, version 4.2.0 (R CORE TEAM 2022). This sample

size (10%) is far above the recommendation (1-ha plot per 1000 ha or 0.1%) for floristic inventories in natural forests (Alder and Synnott 1992), which are more complex in structure and more diverse than urban forests. The perimeter of the blocks was measured using the Google Maps application with the ‘Measure Distance’ function, and it varied from 257.8 m to 692.5 m (446.1 ± 98.1 m). We summed the perimeters of the blocks within neighborhoods to provide the total perimeter per neighborhood, which ranged from 447.5 m to 10064.7 m (2997.5 ± 1972.2 m).

Floristic inventory

All street trees with a diameter at breast height (dbh) ≥ 1 cm, found on the sidewalks surrounding each block, were inventoried. We collected data regarding (1) tree locations (tree code, street, neighborhood, block number) and (2) tree characteristics (vernacular name, height, dbh), and a picture was often taken to help in the comparison with species images on specialized platforms (see next section).

We measured the dbh at 1.30 m above the ground (or at another height when irregularities were present on the trunk) for all trees using a diametric tape (Forestry Suppliers, Jackson, MS). Tree identification was performed in the field using guides such as Lorenzi (2002) and Silva Junior (2005) and with the assistance of botany experts from the Federal University of Amapá. It was not necessary to collect botanical vouchers and compare with Herbarium specimens because the species we found are commonly found in urban areas and easily identifiable. Further, most native species found in urban area are those commonly used by locals for fruit production (such as *Pouteria caimito*, *Spongias mombin*, *Euterpe oleracea*), which makes it easier to access the genus level through vernacular names, for further consultation on the literature to reach the species level. Circumscriptions for flowering plants followed APG IV (Angiosperm Phylogeny Group 2016); Smith and Brown (2018) for gymnosperms. Botanical nomenclature was revised using Tropicos database (<https://www.tropicos.org/home>), REFLORA (<https://reflora.jbrj.gov.br/consulta/#CondicaoTaxonCP>) and The World Flora Online (WFO) Plant List (<https://wfo-plantlist.org/>).

Species classification

We classified the species according to their origin (native or exotic) and fruit use (fruit-bearing or non-fruit-bearing). We checked the origin of the species in different databases such as Global Biodiversity Information Facility (www.gbif.org), REFLORA Program, IBFlorestas (www.ibflorestas.org.br), and on the specialized literature (Lorenzi 2002; Marto 2005; Patro 2018). We designated a species as native

if they are naturally found in the Amazon biome. Species from other biomes (such as the Atlantic Forest or Brazilian Cerrado) were classified as exotic. We designated as fruit-bearing only those species used for human consumption if according to information from the literature above and through observation in local markets and on streets where local fruits are sold.

Sociodemographic data

Twelve variables were selected to represent sociodemographic differences among Macapá's neighborhoods: neighborhood age was obtained from the municipal government's database; average annual per capita income (converted from Brazilian R\$ to \$USD at an exchange rate of 5.1:1.0), population density (individuals/km²), proportion of occupied households, and proportion of men and women in different age groups (< 18 years, ≥ 18-<30 years, ≥ 30-<60 years, and ≥ 60 years) were obtained from the Brazilian Institute of Geography and Statistics (IBGE) platform (Table 1).

Table 1 Descriptive statistics of predictor variables in neighborhoods of Macapá, AP, Brazil. SD: standard deviation. Per capita income was obtained in Brazilian real (R\$) and converted to United States Dollar (\$USD) at an exchange rate of 5.1:1.0

Variables	Minimum	Maximum	Median	Mean ± SD
Neighborhood Age (years)	11	39	29	29.94 ± 7.64
Annual Per Capita Income (\$USD)	1678.82	7135.01	3184.54	3592.42 ± 1300.83
Density (ind./km ²)	545.94	17393.74	4766.24	4840.16 ± 3661.5
Occupied Homes (%)	51.2	95.47	91.11	89.56 ± 7.87
Men ≥ 60 years (%)	1.15	5.13	2.15	2.44 ± 1.07
Women ≥ 60 years (%)	0.73	9.2	2.24	3.01 ± 1.91
Men ≥ 30 and < 60 years (%)	14.47	21.29	16.69	17.01 ± 1.71
Women ≥ 30 and < 60 years (%)	12.79	26.12	17.61	18.46 ± 2.99
Men ≥ 18 and < 30 years (%)	7.91	15.17	11.52	11.78 ± 1.8
Women ≥ 18 and < 30 years (%)	10.76	15.42	12.51	12.83 ± 1.27
Men < 18 years (%)	10.7	23.84	17.9	17.46 ± 3.54
Women < 18 years (%)	11.41	22.56	17.55	17.01 ± 3.29

Neighborhood age is an indicative of human occupation span and about the installed infrastructure (paved streets, sidewalks, electricity network).

Data analysis

Analyses were conducted at the neighborhood level, as sociodemographic information was provided at this level by the municipality and IBGE. We aggregated the data of the species collected on sidewalks surrounding the blocks by neighborhood and used the sum of the perimeters to control for the sampling effort per neighborhood. Tree density was calculated by dividing the number of sampled trees by the sum of the sampled perimeter (sampled sidewalks) of each neighborhood.

The dimensionality of the sociodemographic variables was reduced through Principal Component Analysis (PCA). The variables were transformed into variance units before running the PCA. The first three principal components (axes) were used as predictor variables in regression analyses.

We conducted the subsequent analyses for all species together, and separately for exotic, native, fruit-bearing, and non-fruit-bearing species, as well as for different size classes (all stems ≥ 1 cm dbh; small stems 1 ≤ dbh < 10 cm; medium-size stems 10 ≤ dbh < 30 cm; and large stems ≥ 30 cm dbh) to assess overall patterns as well as patterns within each group of species and size classes.

To assess how well our sampling effort captured tree diversity in the study area, we constructed interpolated and extrapolated species accumulation curves (Chao et al. 2014) using species abundance lists. Extrapolated species richness up to double the reference sample size (Hsieh et al. 2020) was used to compare richness (q=0) with observed richness.

Modeling of tree distribution

We calculated the Bray-Curtis dissimilarity index between neighborhoods using species abundance data. To reduce the dimensionality of data we applied a Principal Coordinate Analysis (PCoA) and used the first axis to represent species composition.

The effect of sociodemographic gradients (PCA Axes 1, 2, and 3) on tree abundance and estimated species richness in the neighborhoods was tested using generalized linear models (GLM) with negative binomial distribution (Zuur et al. 2009). We used this type of distribution error family to account for the overdispersion (ratio between residual deviance and degrees of freedom > 1) detected when a Poisson distribution error was used. The relationship between species composition (PCoA Axis 1) and sociodemographic gradients was assessed with GLM using the Gamma error

distribution and logarithmic link functions (Zuur et al. 2009). Since the gamma error distribution does not allow for negative values, which are frequent in PCoA axes, we rescaled the PCoA scores to positive values by summing the absolute of the minimum value in the PCoA scores plus a ten-thousandth of a unit, making all scores greater than zero, while preserving the order and distance among themselves. As sampling effort differed between neighborhoods, we included the length of sidewalks (in km) in the models to control for this type of variation. As income data were unavailable for one neighborhood the analyses were run for 31 neighborhoods. The analysis for large stems was conducted using only 12 neighborhoods where large street trees were present.

We used the R platform, version 4.2.0 (R Core Team 2022), for all analyses. We used the package ‘stats’ for the PCA and GLM (with Gamma family), and the package ‘MASS’, version 7.3.56 (Venables and Ripley 2002), for the GLM with negative binomial family. Species richness was estimated using the package ‘iNEXT’, version 2.0.20 (Hsieh et al. 2020). The ‘vegan’ package (Oksanen et al. 2022), version 2.6-2, was used to calculate Bray-Curtis dissimilarities, and the ‘ape’ package (Paradis and Schliep 2019), version 5.6.2, was used to run the PCoA.

We used the R platform, version 4.2.0 (R Core Team 2022), for all analyses. We used the package ‘stats’ for the PCA and GLM (with Gamma family), and the package ‘MASS’, version 7.3.56 (Venables and Ripley 2002), for the GLM with negative binomial family. Species richness was estimated using the package ‘iNEXT’, version 2.0.20 (Hsieh et al. 2020). The ‘vegan’ package (Oksanen et al. 2022), version 2.6-2, was used to calculate Bray-Curtis dissimilarities, and the ‘ape’ package (Paradis and Schliep 2019), version 5.6.2, was used to run the PCoA.

Results

A total of 1623 trees were sampled, belonging to 53 species (including one unidentified morphotype) distributed across 46 genera and 22 botanical families (Table 2). Most stems (86.2%) were of exotic species, and a small portion (13.5%) was native (0.3% were unclassified); 45.5% are fruit-bearing species, and 54.5% are non-fruit-bearing species. The most represented botanical families were Fabaceae (11 species), Arecaceae (9), Anacardiaceae (4), Myrtaceae (4), and Bignoniaceae (3), and the most abundant species were *Carpentaria acuminata* (Arecaceae, with 21% of stems), *Mangifera indica* (Anacardiaceae, 20%), *Syzygium malaccense* (Myrtaceae, 8%), *Moquilea tomentosa* (Chrysobalanaceae, 7%), and *Andira inermis* (Fabaceae, 5%).

The species accumulation curve was near reaching an asymptote (Supplementary Material: Figure S1), but the observed species richness (53) fell within the confidence interval of estimated richness (60.2 ± 12.1 ; estimated richness $\pm 95\%$ CI; 99.7% sample coverage), indicating that the sampling effort was sufficient to capture tree species diversity in the urban area of Macapá. Separating the sample into categories, the observed richness also fell within confidence intervals, and sample coverage (SC) was high for the following: exotic trees (37 observed species, 41.7 ± 7.1 estimated, and 99.8% SC), native trees (15 and 17.5 ± 4.9 ; 99.3% SC), fruit-bearing trees (27 and 30.8 ± 6.7 ; 99.7%), and non-fruit-bearing trees (26 and 29.4 ± 5.8 ; 99.7%).

The size of the trees varied significantly, from 1 cm to 175.07 cm dbh (19.42 ± 19.76 cm; mean \pm standard deviation; median = 12.62 cm). The size distribution was skewed, with 37.1% of stems being small-sized ($1 \leq \text{dbh} < 10$ cm), 45.5% being medium-sized ($10 \leq \text{dbh} < 30$ cm), and 17.4% being large-sized (≥ 30 cm dbh) (Figure S2).

Density and tree richness by neighborhoods

The density of medium-sized trees was higher than that of small and large-sized trees (Table S1). For all species combined, the density of medium-sized trees was 7.9 ± 4.6 ind./km (mean \pm standard deviation), while small and large-sized trees were 6.04 ± 4.37 ind./km and 1.82 ± 3.62 ind./km, respectively. The overall mean density for all species was 15.7 ± 7.74 ind./km. The density of exotic species (13.33 ± 6.84 ind./km) was much higher than that of native species (2.32 ± 1.82 ind./km), and the density of non-fruit-bearing species (9.2 ± 5.25 ind./km) was higher than that of fruit-bearing species (6.51 ± 4.37 ind./km).

The estimated richness (Hill number – $q=0$) varied from 2.00 to 36.54 species (15.13 ± 8.39 species) among neighborhoods (Table S2). Exotic species richness (11.42 ± 6.27) was over three times higher than native species richness (3.34 ± 2.65), and the richness of fruit-bearing species (7.07 ± 4.85) was similar to non-fruit-bearing species (7.57 ± 4.23).

Sociodemographic gradients

The average age and income of the neighborhoods in Macapá were 29.9 years and \$3,592.42, respectively. The average population density per km² was 4,840.16, and the average proportion of occupied households was 89.5%. The average proportion of men and women was similar, with a slightly higher percentage of women for all age groups ≥ 18 years: for those aged ≥ 60 years, the proportion was 2.4% and 3.0% for men and women, respectively; for those aged ≥ 30 and < 60 years, the proportion was 17.0%

Table 2 Abundance of tree species sampled in sidewalks of Macapá city (Brazil) according to origin, fruit use and different tree sizes

Family and species	Origin (Brazilian biome)	Fruit use	dbh size classes (cm)			Total
			≥ 1-<10	≥ 10-<30	≥ 30	
Anacardiaceae						
<i>Anacardium occidentale</i> L.	Native (Amazonia/Caatinga/Cerrado/Atlantic Forest/Pampa/Pantanal)	Fruit	11	20	1	32
<i>Mangifera indica</i> L.	Exotic	Fruit	64	118	143	325
<i>Spondias mombin</i> L.	Native (Amazonia/Cerrado/Atlantic Forest)	Fruit	2	0	0	2
<i>Spondias purpurea</i> L.	Exotic	Fruit	0	1	0	1
Annonaceae						
<i>Annona muricata</i> L.	Exotic	Fruit	1	0	0	1
<i>Annona squamosa</i> L.	Exotic	Fruit	4	1	0	5
Areaceae						
<i>Attalea excelsa</i> Mart. ex Spreng.	Native (Amazonia/Cerrado/Pantanal)	Fruit	0	0	2	2
<i>Bactris gasipaes</i> Kunth	Exotic	Fruit	0	4	0	4
<i>Carpentaria acuminata</i> (H. Wendl. & Drude) Becc.	Exotic	Non fruit	188	157	2	347
<i>Cocos nucifera</i> L.	Exotic	Fruit	36	33	2	71
<i>Dypsis lutescens</i> (H. Wendl.) Beentje & J. Dransf.	Exotic	Non fruit	1	2	0	3
<i>Euterpe oleracea</i> Mart.	Native (Amazonia/Cerrado)	Fruit	15	13	1	29
<i>Phoenix dactylifera</i> L.	Exotic	Fruit	0	1	0	1
<i>Phoenix roebelenii</i> O'Brien	Exotic	Non fruit	0	1	0	1
<i>Roystonea oleracea</i> (Jacq.) O.F. Cook	Exotic	Non fruit	3	21	6	30
Bignoniaceae						
<i>Crescentia cujete</i> L.	Exotic	Non fruit	1	0	0	1
<i>Tabebuia aurea</i> (Silva Manso) Benth. & Hook. f. ex S. Moore	Native (Amazonia/Cerrado)	Non fruit	1	0	0	1
<i>Handroanthus serratifolius</i> (Vahl) S.O. Grose	Native (Amazonia/Caatinga/Cerrado/Atlantic Forest/Pantanal)	Non fruit	15	27	0	42
Caricaceae						
<i>Carica papaya</i> L.	Exotic	Fruit	16	15	0	31
Chrysobalanaceae						
<i>Moquilea tomentosa</i> (Benth.) Fritsch	Exotic (Atlantic Forest)*	Non fruit	45	65	7	117
Combretaceae						
<i>Terminalia catappa</i> L.	Exotic	Non fruit	3	6	11	20
Cycadaceae						
<i>Cycas revoluta</i> Thunb.	Exotic	Non fruit	32	30	3	65
Fabacea						
<i>Adenanthera pavonina</i> L.	Exotic	Non fruit	1	1	1	3
<i>Anadenanthera falcata</i> (Benth.) Speg.	Exotic (Caatinga/Cerrado/Atlantic Forest)*	Non fruit	0	3	0	3
<i>Andira inermis</i> (Sw.) Kunth	Native (Amazonia/Cerrado/Atlantic Forest)	Non fruit	16	51	15	82
<i>Bauhinia variegata</i> L.	Exotic	Non fruit	1	0	0	1
<i>Caesalpinia</i> cf. <i>pluviosa</i> DC.	Exotic	Non fruit	0	1	0	1
<i>Caesalpinia ferrea</i> Mart.	Exotic (Caatinga/Cerrado/Atlantic Forest)*	Non fruit	1	2	0	3
<i>Clitoria fairchildiana</i> R.A. Howard	Native (Amazonia/Caatinga/Cerrado/Atlantic Forest)	Non fruit	1	6	5	12
<i>Inga edulis</i> Mart.	Native (Amazonia/Caatinga/Cerrado/Atlantic Forest)	Fruit	2	2	0	4
<i>Delonix regia</i> (Bojer ex Hook.) Raf.	Exotic	Non fruit	2	4	2	8
<i>Vachellia farnesiana</i> (L.) Wight & Arn.	Exotic	Non fruit	3	0	2	5
<i>Erythrina</i> sp. L.	Exotic	Non fruit	1	1	0	2
Lauraceae						
<i>Persea americana</i> Mill.	Exotic	Fruit	3	4	1	8
Malpighiaceae						
<i>Malpighia emarginata</i> DC.	Exotic	Fruit	3	1	0	4
Meliaceae						
<i>Azadirachta indica</i> A. Juss.	Exotic	Non fruit	46	26	0	72

Table 2 (continued)

Family and species	Origin (Brazilian biome)	Fruit use	dbh size classes (cm)			Total
			≥ 1-<10	≥ 10-<30	≥ 30	
Moraceae						
<i>Artocarpus heterophyllus</i> Lam.	Exotic	Fruit	0	4	1	5
<i>Ficus benjamina</i> L.	Exotic	Non fruit	10	23	18	51
Musaceae						
<i>Musa</i> sp L.	Exotic	Fruit	2	4	0	6
Myrtaceae						
<i>Psidium guajava</i> L.	Exotic	Fruit	23	18	0	41
<i>Syzygium cumini</i> (L.) Skeels	Exotic	Fruit	7	3	1	11
<i>Syzygium malaccense</i> (L.) Merr. & L.M. Perry	Exotic	Fruit	20	61	53	134
<i>Eucalyptus</i> sp L'Hér.	Exotic	Non fruit	2	0	0	2
Oxalidaceae						
<i>Averrhoa carambola</i> L.	Exotic	Fruit	1	1	1	3
Pinaceae						
<i>Pinus</i> sp L.	Exotic	Non fruit	3	0	1	4
Rubiaceae						
<i>Morinda citrifolia</i> L.	Exotic	Fruit	1	2	1	4
<i>Genipa americana</i> L.	Native (Amazonia/Caatinga/Cerrado/Atlantic Forest/Pantanal)	Fruit	0	0	1	1
Rutaceae						
<i>Citrus aurantifolia</i> Swingle	Exotic	Fruit	8	1	0	9
<i>Citrus sinensis</i> (L.) Osbeck	Exotic	Fruit	2	0	0	2
Sapindaceae						
<i>Talisia esculenta</i> (A. St.-Hil.) Radlk.	Native (Amazonia/Caatinga/Cerrado/Atlantic Forest)	Fruit	0	1	0	1
Sapotaceae						
<i>Pouteria caimito</i> (Ruiz & Pav.) Radlk	Native (Amazonia/Cerrado/Atlantic Forest)	Fruit	0	1	0	1
Urticaceae						
<i>Cecropia</i> sp Loeffl.	Native (Amazonia/Caatinga/Cerrado/Atlantic Forest/Pantanal)	Non fruit	3	1	0	4
Indeterminate						
	-	-	3	2	0	5

*Species exotic for the Amazonian biome but native to any other biome found in Brazil

Table 3 Loadings (Pearson correlation) of the first three principal component analysis axes relating to sociodemographic variables

Variables	PC1	PC2	PC3
Age (years)	0.71	-0.30	0.17
Average Income (USD/neighborhood)	0.82	0.18	-0.05
Density (individuals/km ²)	0.22	-0.77	0.06
Occupied Homes (%)	0.07	-0.26	-0.76
Men ≥ 60 years (%)	0.86	0.03	0.29
Women ≥ 60 years (%)	0.75	0.11	0.37
Men ≥ 30 – < 60 years (%)	0.45	0.62	-0.34
Women ≥ 30 – < 60 years (%)	0.84	0.31	-0.27
Men ≥ 18 – < 30 years (%)	0.27	-0.80	0.04
Women ≥ 18 – < 30 years (%)	0.19	-0.75	-0.24
Men < 18 years (%)	-0.94	0.02	0.16
Women < 18 years (%)	-0.94	0.02	0.01

and 18.4%, respectively; in the age group of ≥ 18 and < 30 years, the proportion was 11.7% and 12.8%, respectively; and for those under 18 years, it was 17.4% and 17.0% for men and women, respectively (Table 1).

The first axis of the PCA explained 44% of the variation in sociodemographic data and described a gradient of increasing neighborhood age, average income, proportion of elderly men and women (> 60 years), and middle-aged women (≥ 30 and < 60 years) and was negatively related to the proportion of men and women under 18 years (Table 3). The second axis explained 21% of the variation and was negatively related to population density and the proportion of young men and women (≥ 18 - < 30 years) but was positively related to the proportion of middle-aged men. The third axis explained 9% of the variation and described a gradient of decreasing proportion of occupied households.

The older neighborhoods received the highest scores on the first axis of the PCA (Figure S3), which coincides with higher income and older population, while the more recently founded neighborhoods received the lowest scores on the first axis, being associated with lower per capita income and with a younger population (Table 3).

Relationship of tree abundance and richness with sociodemographic gradients

The tree abundance on sidewalks was positively related to PC1 (Negative Binomial Regression: Wald's test - $Z=2.19$, $P=0.03$; Table S3), indicating an increase in the number of trees in older neighborhoods with older populations and higher income (Fig. 1A). This same pattern was observed for large trees ($Z=4.19$, $P<0.001$; Fig. 1B), but the relationship between the number of small trees ($Z=0.52$, $P=0.60$) and medium-sized trees ($Z=0.91$, $P=0.37$) with PC1 was not significant (Table S3). The number of trees, regardless of size, was not significantly related ($Z < |1.6|$, $P>0.09$) to PC2 and PC3 (Table S3).

The total number of exotic trees ($Z=2.19$, $P=0.03$) and large exotic trees ($Z=3.96$, $P<0.001$) was positively related to PC1 (Table S3), indicating an increase in the abundance of exotic trees in older neighborhoods with older population and higher income (Fig. 1C and D). There was

no significant relationship between the number of small ($Z=0.71$, $P=0.48$) and medium-sized stems of exotic trees ($Z=0.80$, $P=0.42$) with PC1, and there was no significant relationship between the number of exotic trees of any size with PC2 or PC3 ($Z < |1.63|$, $P>0.10$; Table S3). For native trees, there was a marginally significant relationship ($Z=1.86$, $P=0.06$) between the number of large trees and PC1 (Fig. 1E), but there was no relationship between the total number of trees ($Z=0.87$, $P=0.38$), small ($Z=-1.55$, $P=0.12$), or medium-sized trees ($Z=0.30$, $P=0.78$) with PC1 (Table S3). The number of native trees of any stem size was not related ($Z < |1.03|$, $P>0.3$) to PC2 and PC3 (Table S3).

The total number of non-fruit trees ($Z=2.34$, $P=0.02$) and the number of large non-fruit trees ($Z=3.09$, $P=0.002$) was positively related to PC1 (Table S3), indicating that the abundance of non-fruit trees increased with neighborhood age and with the age and income of residents (Fig. 1I and J). The same pattern was observed for large fruit trees

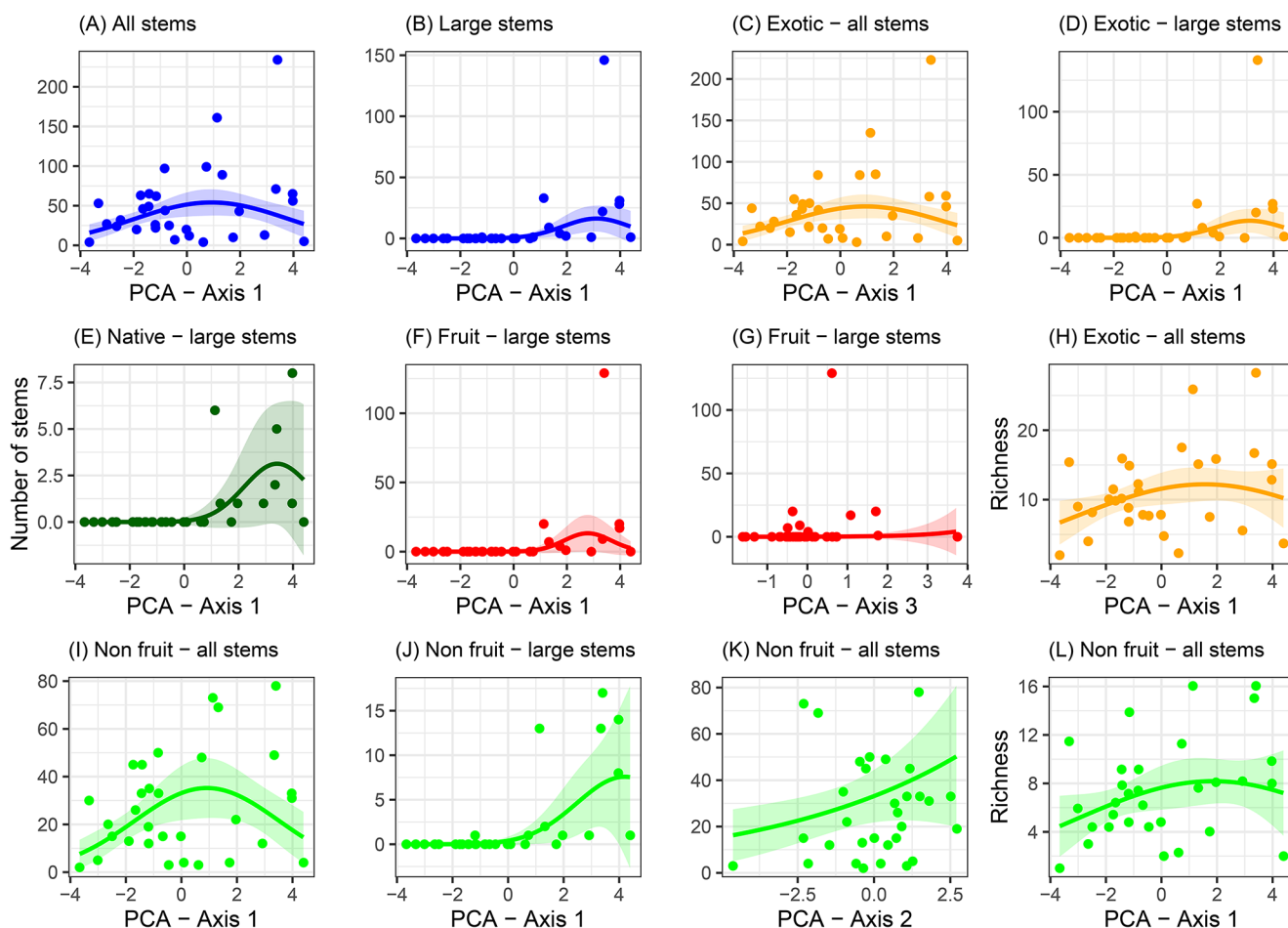


Fig. 1 Relationships between the number of trees and estimated richness (Hill number - $q=0$) with sociodemographic gradients (PC1, PC2, and PC3) in Macapá, Amapá, Brazil. The graph titles indicate the species categories (all, exotic, native, fruit, or non-fruit) and size classes (all, small, medium, and large). The lines represent significant

relationships (or marginally significant in E and L), and shading represents the 95% confidence interval. The results of the regression models are in the Supplementary Material: Table S3. The analysis for large stems was conducted using data from twelve neighborhoods

($Z=2.76$, $P=0.006$), which increased with PC1 (Fig. 1F). In addition, the total number of non-fruit trees was positively related ($Z=1.98$, $P=0.047$) to PC2, indicating that the abundance of non-fruit trees tends to be higher in neighborhoods with low population density and with a low proportion of young people (Fig. 1K). The number of large fruit trees was positively related ($Z=2.70$, $P=0.007$) to PC3, indicating that the abundance of large fruit trees increases in neighborhoods with a low rate of occupied households (Fig. 1G).

The total number of fruit trees and the number of small and medium-sized stems of fruit trees were not significantly related ($Z < |1.32|$, $P > 0.19$) to any of the gradients (Table S3). The total number of non-fruit trees was not related to PC3 ($Z=1.20$, $P=0.23$). The number of small and medium-sized stems of non-fruit trees was not significantly related ($Z < |1.72|$, $P > 0.09$) to any of the gradients, and the number of large stems of non-fruit trees was not significantly related ($Z = -2.27$, $P=0.79$) to PC2 (Table S3).

The species richness of exotic trees was positively related ($Z=2.06$, $P=0.04$) to PC1 (Table S3), indicating that the number of exotic species increases with neighborhood age, and with the age and income of the human population (Fig. 1H). There was also a marginally significant positive relationship ($Z=1.85$, $P=0.06$) between the richness of non-fruit species and PC1 (Fig. 1L). The richness of exotic and non-fruit species was not significantly related ($Z < |0.84|$, $P > 0.4$) to PC2 or PC3, and the total richness, native species richness, and fruit species richness were not significantly related ($Z < |1.66|$, $P > 0.09$) to any of the gradients (Table S3).

Relationship of species composition with sociodemographic gradients

The species composition of trees of all sizes (axis 1 of PCoA captured 21.7% of variation of the original distance matrix) was significantly related to PC2 (GLM: $t=3.03$, $P=0.006$; Table S4), indicating that the species composition of trees changes along the gradient of population density and the proportion of young people (Fig. 2A). The composition of all trees was marginally significant related ($t=2.01$, $P=0.056$) to PC3, indicating that the composition changes according to the household occupancy (Fig. 2B). There was no relationship between the composition of all trees and PC1 ($t=1.11$, $P=0.28$).

The composition of small-sized trees (axis 1 of PCoA explained 19.9% of variance) was related to PC2 ($t=2.91$, $P=0.008$; Table S4), showing that differences in the species composition of recently planted trees are associated with population density and with the proportion of young

people (Fig. 2C). The composition of small-sized trees was not related to PC1 or PC3 ($t < |1.1|$, $P > 0.32$).

The composition of medium-sized trees (axis 1 of PCoA explained 15.9% of variance) was related to PC3 ($t=2.08$, $P=0.048$; Table S4), indicating that the composition of well-established species changes according to the household occupancy (Fig. 2D). There was no relationship between the composition of medium-sized trees and PC1 or PC2 ($t < |1.43|$, $P > 0.16$).

The composition of large-sized trees (axis 1 of PCoA explained 32.7% of variance) was related to PC1 ($t=6.45$, $P < 0.001$; Table S4), showing that there is a change in the composition of well-established species associated with the age of the neighborhoods and with the age and income of the human population (Fig. 2E). The composition of large-sized trees was not related to PC2 or PC3 ($t < |2.12|$, $P > 0.08$).

Among the 16 most representative species (which contributed to $> 1\%$ of stems in the sample), a quarter (*Ficus benjamina*, *Anacardium occidentale*, *Euterpe oleracea*, and *Terminalia catappa*) showed an asymmetric distribution, with higher abundance ($> 60\%$) in neighborhoods with high population density ($> \text{median of } 4766 \text{ ind./km}^2$; Fig. 3). The remaining species (*Carpentaria acuminata*, *Mangifera indica*, *Syzygium malaccense*, *Moquilea tomentosa*, *Andira inermis*, *Azadirachta indica*, *Cocos nucifera*, *Cycas revoluta*, *Handroanthus serratifolius*, *Psidium guajava*, *Carica papaya*, and *Roystonea oleracea*) showed similar abundances ($> 40\%$) regarding human population density.

Among the 16 most abundant species regarding small-sized stems, *E. oleracea*, *Citrus aurantifolia*, and *Syzygium cumini* showed higher abundance in neighborhoods with high population density, while *M. indica* and *A. occidentale* were more abundant in neighborhoods with low population density (Figure S4A). The remaining species (*C. acuminata*, *Azadirachta indica*, *M. tomentosa*, *C. nucifera*, *C. revoluta*, *P. guajava*, *S. malaccense*, *C. papaya*, *A. inermis*, *H. serratifolius*, and *F. benjamina*) showed a symmetrical distribution associated with human population density.

Among the 15 most abundant species regarding medium-sized stems, *C. nucifera*, *C. revoluta*, *A. indica*, and *A. occidentale* were more abundant in neighborhoods with high household occupancy rates ($> \text{median of } 91\%$), while *R. oleracea*, *P. guajava*, and *E. oleracea* were more abundant in neighborhoods with lower occupancy rates (Figure S4B). The most representative species with symmetrical distribution were *C. acuminata*, *M. indica*, *M. tomentosa*, *S. malaccense*, *A. inermis*, *H. serratifolius*, *F. benjamina*, and *Carica papaya*. All nine most representative species regarding large-sized stems (*M. indica*, *S. malaccense*, *F. benjamina*, *A. inermis*, *T. catappa*, *M. tomentosa*, *R. oleracea*, *C. fairchildiana*, and *C. revoluta*) had higher abundance in neighborhoods with higher income (Figure S4C).

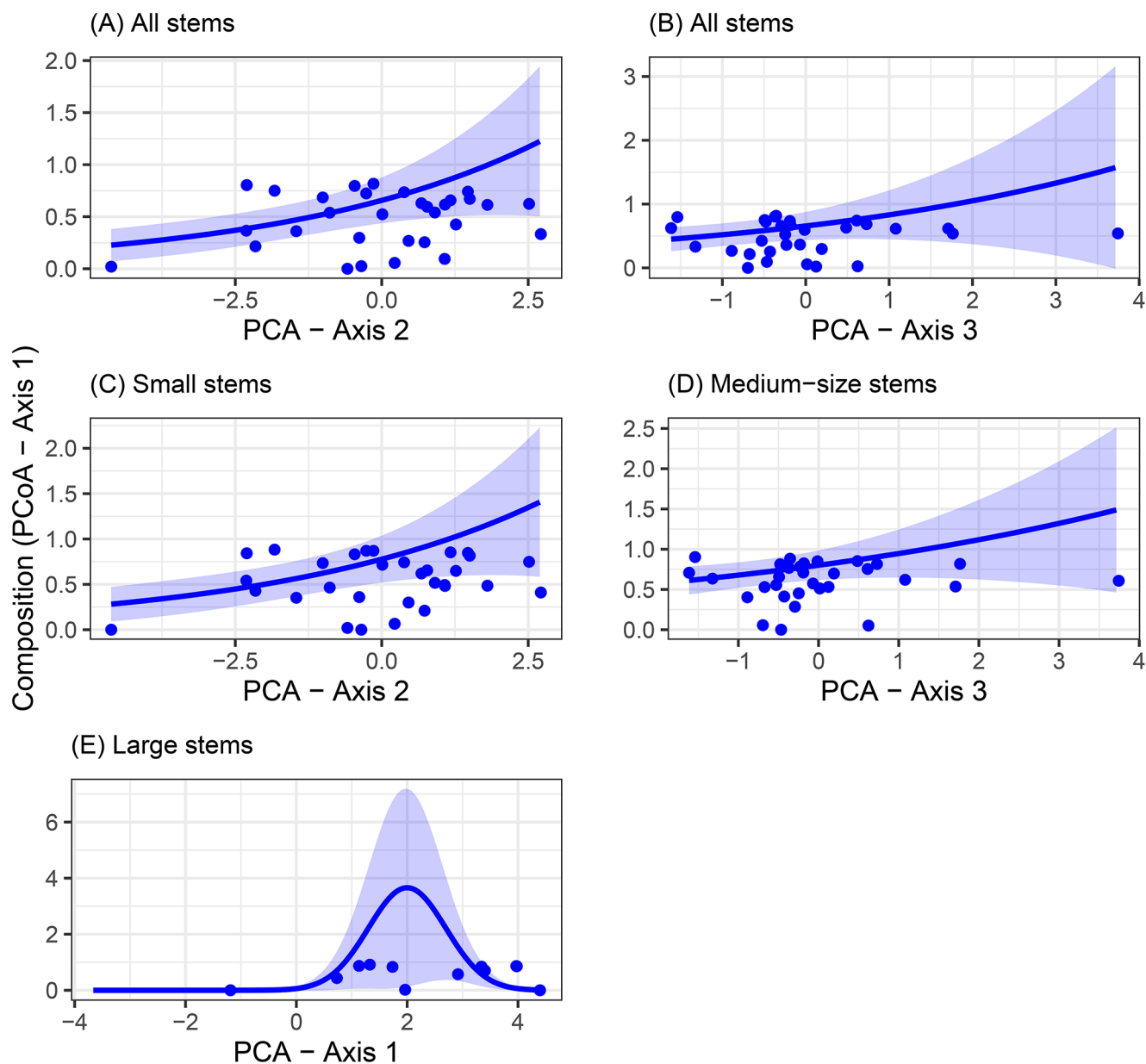


Fig. 2 Relationships of tree species compositions (Axis 1 of PCoA) with sociodemographic gradients (Axes 1, 2, and 3 of PCA) in Macapá, AP, Brazil. The graph titles indicate the size classes (all, small, medium, and large). The lines represent significant relationships

(or marginally significant in B), and the shading represents the 95% confidence interval. The results of regression models are available in the Supplementary Material: Table S4. The analysis for large stems was conducted using data from twelve neighborhoods

Discussion

The results of this study indicate that in Macapá, urban trees are mostly exotic, non-fruit-bearing, and encompass predominantly medium-sized stems. These findings may reflect a balance between aesthetic aspects, ease of maintenance, and space limitations in the urban area. Medium-sized trees can provide shade and improve the thermal environment without taking up too much space or interfering with power lines and buildings.

Regarding the use of exotic species, this seems to be a pattern not only in Brazilian Amazonian cities (Vieira and Panagopoulos 2020; Soares et al. 2021), but also in other regions of Brazil (Silva et al. 2020; Santos et al. 2019; Moro and Castro 2015; dos Santos et al. 2010) and worldwide (e.g., Nagendra and Gopal 2011; Useni et al. 2018). However, although not all exotic species are invasive, negative impacts of these species have been widely documented, mostly related to the loss of local biological diversity (Burghardt et al. 2009; Heleno et al. 2009; Burghardt and Tallamy 2013; Pysek et al. 2020), increased risk of

Fig. 3 Direct ordination of exotic, native, fruit, and non-fruit tree species based on neighborhood population density in Macapá, Amapá, Brazil. Population density was used in the ordination because it is strongly related to PC2, which, in turn, was significantly related to species composition (see Fig. 2 and Table S4)



extinction of native species (Lorenzi 2002; Wilcove et al. 1998; Pimentel et al. 2005; Blum et al. 2008; Pysek et al. 2020), as well as economic losses. In the USA, exotic species caused losses of approximately US\$ 120 billion per year (Pimentel et al. 2005).

Native species can ensure ecological balance, attract tourists, and provide residents with well-being and pride, as the vegetation presents distinct and typical aspects of their regions. This creates an identity and appreciation for local fauna, reinforcing a sense of belonging (Osako et al. 2016; Cilliers et al. 2019), while also increasing awareness about native plants and the need to conserve them (Williams et al. 2005; Moro and Castro 2015). Considering that Macapá city is in a privileged region of Brazil regarding ecosystem conservation, containing a great richness of natural resources, it is extremely important that native species be well valued, and that urban spaces can also contribute to the conservation of endemic species.

Regarding the preference for non-fruit-bearing trees, this may be related to concerns about the mess caused by fruit drop and the possibility of attracting unwanted animals. In a study investigating the benefits and barriers to incorporating fruit-bearing plants on public lands in Minnesota (USA), concerns about fruit drop and attraction of pests were frequently mentioned by respondents (Walhowe 2022). In Canada, municipal managers cited risks related to public health and safety, such as slips due to fallen fruits and the potential to cause allergies (Kowalski and Conway 2023). Indeed, one of the main ecosystem disservices associated with the establishment of urban fruit forests is allergic reactions caused by plant allergens (Cariñanos et al. 2019). On the other hand, integrating fruit-bearing trees in public spaces can contribute to local food security and biodiversity conservation by providing healthy and fresh food for the local human and wildlife community. In terms of Brazil, this importance is even more pronounced when considering that food insecurity affects more than 33 million people in the country (Rede Penssan 2022) and that approximately 221 thousand people are homeless (IPEA 2022).

The tree density in Macapá (15.7 trees per linear km) is the highest among the states of the Brazilian Amazon (Soares et al. 2021). However, it is still a low density compared to other urban centers in Brazil (e.g., 24.8 trees/km in Campinas) and the rest of the world (e.g., 40.2 trees/km in London and 46.6 trees/km in California) (Álvarez et al. 2015; Taylor et al. 2015; McPherson et al. 2016). Considering that the presence of green areas in cities has been associated with a range of benefits to increase people's life quality (e.g., stress reduction, improvement of mental health and air quality, reduction of urban heat) (Tzoulas et al. 2007; Lee and Maheswaran 2011), it is essential to promote the presence and maintenance of urban vegetation.

Relationship of tree abundance and species richness with sociodemographic gradients

The distribution of tree cover in Macapá was related to sociodemographic gradients. Older neighborhoods with an older human population and higher income have higher tree abundance and species richness. This positive relationship between tree cover, income, and neighborhood age is a result frequently found in the literature (e.g., Shiraishi 2022; Useni et al. 2021; Shams et al. 2020; Escobedo et al. 2015; Dobbs and Nitschke 2014; Lowry Jr et al. 2012; dos Santos et al. 2010; Heynen et al. 2006; Pedlowski et al. 2002). Urban tree cover in Macapá reflects the spatialization of social inequality in the city and the lack of afforestation planning for city expansion, as we could have new planned neighborhoods with extensive tree cover considering current debates on sustainable and green cities (Vilanova et al. 2024; Pauleit et al. 2019).

The positive relationship between population income and urban tree cover has been termed “luxury effect” in the literature. As individuals' purchasing power increases, they occupy landscapes with greater richness and abundance of plants, either by planting themselves or by moving to areas with high tree cover (Hope et al. 2003). However, this pattern may also result from a “legacy effect”, with greater tree diversity in older and wealthier neighborhoods (Clarke et al. 2013). Over time, the positive effect of income on tree cover becomes more pronounced, because wealthy homeowners have the financial resources to invest in the maintenance and cultivation of trees (Lowry Jr et al. 2012).

The unequal distribution of tree cover in a city exacerbates inequalities associated with life quality. Trees provide various ecosystem services that are essential for human life (e.g., temperature regulation, noise attenuation, air quality regulation). In Colombia, a higher species richness was found in areas of high socioeconomic status, and as a result, the pollution removal potential of these areas was almost twice that of lower-income strata (Escobedo et al. 2015). In the US, higher incomes were associated with high urban vegetation cover, which promotes cooler temperatures (Jenerette et al. 2007). Deprivation of access to environmental amenities constitutes an issue of environmental injustice, deepening the inequalities that lead to this situation in the first place.

The abundance of exotic trees in Macapá is higher in older neighborhoods with older populations and higher income. Some studies suggest that areas with higher income tend to have a higher abundance of exotic species in urban green spaces (e.g., Schwarz et al. 2015; Shams et al. 2020). The introduction of exotic species into older urban areas can be attributed to historical factors, such as the influence of colonizers or aesthetic preferences during the period in which

the cities were established (McNeely et al. 2001; dos Santos et al. 2010). However, this is not a universal rule - in Rio de Janeiro, the opposite pattern was observed, with exotic trees being more abundant in newer neighborhoods (dos Santos et al. 2010). Additionally, access to exotic species probably increases with income due to the higher accessibility to purchase several species of plants and because of available financial resources to manage these plants. However, it is important to note that the relationship between income and the presence of exotic species may be influenced by other factors, such as urban planning policies, availability of public services, and cultural preferences.

The abundance of non-fruiting trees, large stems of fruiting trees, and overall large trees in Macapá is higher in older neighborhoods with older populations and higher income. A higher proportion of fruiting trees in older neighborhoods was also found in California (Avolio et al. 2015). It is expected that older neighborhoods have a higher abundance of trees and large trees, considering the longer time the trees have had to grow and the longer time the population has had to plant trees compared to more recently founded neighborhoods.

Our results indicate that the abundance of large fruiting trees increases in neighborhoods with a low rate of household occupancy, while the abundance of large non-fruiting trees increases with low population density. In Philadelphia, trees found in areas with higher percentages of vacant housing units have higher chance of surviving (Bigelow Jr et al. 2024), and in Melbourne, lower population density was associated with greater vegetation cover (Ren et al. 2023). Therefore, it is possible that as the number of people decreases, there is a reduction in the amount of pruning and cutting of the vegetation, which may allow trees to grow more freely. On the other hand, a higher number of people may influence tree preservation through maintenance activities (Useni et al. 2021), such as watering, pest control and fertilization. Our findings suggest that lower population density and lower household occupancy rates have a positive impact on vegetation growth over an extended period.

Additionally, the positive relationship between large trees and lower occupancy rates in urban areas may be associated with socioeconomic and environmental aspects - the lack of occupancy may result in the establishment of fewer commercial buildings, avoiding tree-cutting which would be a frequent activity to create these commercial spaces. In this sense, lower occupancy can contribute to the preservation of large trees, as the absence of commercial pressures may result in fewer tree removals to make way for commercial buildings.

Relationship of species composition with sociodemographic gradients

In our study area, small-sized stems of species such as *E. oleraceae* (açai palm), *C. aurantifolia* (lime tree), and *S. cumini* (java plum) showed higher abundance in neighborhoods with high population density. Population density can influence species choices for planting, with residents in densely populated areas often opting for smaller-sized species with lower maintenance requirements due to space constraints and limited access to sunlight (Qian et al. 2020).

The availability of fruit trees can promote a sense of connection with nature, improve psychological well-being, and enhance the perceived urban environment quality by residents (Cilliers et al. 2019). The *E. oleracea* (açai) is a native palm tree of the Amazon region, widely valued for its edible fruit – the açai berry. Açai is traditionally consumed by local communities as a staple food and holds high nutritive and cultural value. Although it does not provide much shade – a highly favorable attribute for cities in tropical regions – it does not conflict with urban structures (e.g., electrical wiring and sidewalk), and perhaps for this reason, it has been the most abundant native palm tree in the urban tree cover of Brazilian Amazon cities (Soares et al. 2021).

On the other hand, *M. indica* (mango tree) and *A. occidentale* (cashew tree) were more abundant in neighborhoods with low population density. The mango tree, native to India, is one of the most cultivated and appreciated fruit trees worldwide (Morton 1987; Singh and Sharma 2020) and is one of the main species of urban tree cover in the Amazon (Soares et al. 2021). In the capital of the state of Pará (Brazilian Amazon), mango trees have been declared as historical and cultural heritage (Koury et al. 2017; IPHAN 2020; PMB 2021). In Macapá, mango trees have significant cultural and economic importance – the fruits are highly appreciated and are an important source of food and income for the local population - the sale of mangoes can generate income of US\$ 15 per day per seller (Pacheco 2019).

However, the mango tree can also pose challenges in the urban environment – it has a root system characterized by the presence of a taproot, which can deeply penetrate the soil, and a large size (they can reach up to 30 m in height) (Mouco 2015), leading to many conflicts with urban structures - conflicts with the electrical wiring system, sidewalk breakage, and falling fruit on the public walkway (Soares et al. 2021). Perhaps for these reasons, mango trees are more abundant in areas with lower population density. Infrastructure issues such as sidewalk width and paving, distribution networks for electricity, telephone, Internet, water, and sewage, can influence both the selection of species and the probability of planting and cutting trees (Balensiefer and Wiecheteck 1985; Shi et al. 2023).

The composition of well-established tree species in Macapá changes according to the household occupancy rate. Medium-sized stems of species such as *C. nucifera* (coconut palm), *C. revoluta* (sago palm), *A. indica* (neem), and *A. occidentale* (cashew tree) were more abundant in neighborhoods with high household occupancy, while *R. oleracea* (royal palm), *P. guajava* (guava tree), and *E. oleracea* (açai palm) were more abundant in neighborhoods with lower occupancy rates.

In Macapá, most neighborhoods have a high occupancy rate (median > 91%). Therefore, changes in composition may be a result of subtle variation in occupancy, and possible interference of other factors (e.g., tree maintenance requirements, preferences related to fruit production, growth rate). For example, the guava tree - a species related to lower occupancy rates - is easily dispersed by birds and highly adaptable to different types of soil and climate, able to tolerate both dry and humid conditions (Jettanacheawchankit et al. 2020; Sthapit 2019). It is a species that does not require human intervention to establish and tolerates adverse environmental conditions in its growth and development, which may be why it is associated with lower household occupancy rates. On the other hand, the neem tree - which is related to higher occupancy rates - is a fast-growing species, resistant to pruning, with a dense canopy that provides shade and shelter for several animal species, and requires little maintenance (Biswas et al. 2002; Sujarwo et al. 2016). It is a species with rapid growth, which may be preferred by people. In the Democratic Republic of Congo, people preferred to plant tree species believed to reach maturity at a relatively young age (Sikuzani et al. 2019).

All representative large-sized stems of tree species in Macapá were more abundant in neighborhoods with higher income. A similar pattern was also observed in the US (Wolf 2013). Some studies have explored the positive influence of large trees on property values and residents' income. For example, in the US, it was observed that large trees were associated with a significant increase in house value, and furthermore, these trees provided additional financial benefits, such as reducing energy costs (Donovan and Butry 2010). Studies like this highlight the importance of large trees in urban settings, not only for their aesthetic and environmental function but also for the economic benefits they bring. Mature and well-maintained trees can enhance property values, making them more attractive to potential buyers and tenants, which, in turn, can result in higher income for the owners.

Cities can combine urban development with environmental preservation, providing a higher life quality for citizens and ensuring harmony between human activities and the environment. To achieve these goals, policymakers need to value the local potentials of each region, recognize the

richness of local biodiversity, and establish public policies that promote it, especially regarding native trees, contributing to ecological restoration and to the conservation of urban ecosystems.

However, it is crucial to ensure that all city residents have equal access to the benefits provided by trees. The unequal distribution of green areas and trees in cities exacerbates socio-environmental disparities, with more vulnerable communities being deprived of the benefits of urban greening. Ensuring that low-income areas and historically marginalized communities have equitable access to green spaces and adequate tree coverage is key to promote environmental justice and for the development of healthier and sustainable cities.

The findings of this study provide a comprehensive view of the distribution of tree cover and urban green equity in the extreme northeastern Brazilian Amazon. The information generated is a fundamental basis to guide both the conceptual and practical understanding of tree cover distribution for different layers of the population, enabling public authorities to develop appropriate strategies for urban afforestation and promotion of socio-environmental sustainability.

The urban socio-environmental problems are challenges which must be addressed for the implementation of the 2030 Agenda for Sustainable Development - a plan of action prepared by the United Nations (UN) with 17 goals for achieving sustainable development (SDGs) (UN - General Assembly 2015). In this sense, this study provides a regional diagnosis and enhances the capacity to promote local sustainable development, also contributing to aligning local indicators with the goals of the Brazilian 2030 Agenda.

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s11252-024-01592-4>.

Acknowledgements We would like to thank Taissa Barros Lobato, Estefany Furtado Baia, Lucio Royer Trujillo Rodriguez and Charly Sanches for their assistance in the floristic inventory. JJT was supported by the National Council for Scientific and Technological Development-CNPq with a Research Productivity Scholarship (#316281/2021-2).

Author contributions L. A. J., S. M. S. S. and L.M.F. wrote the main manuscript text and J.J.T. prepared figures and tables. L.A.J, J.A.A and J.J.T. conducted the statistical analyses. All authors reviewed the manuscript.

Funding Author JJT was supported by the National Council for Scientific and Technological Development-CNPq with a Research Productivity Scholarship (#316281/2021-2).

Data availability Data is provided within the supplementary material.

Declarations

Competing interests The authors declare no competing interests.

References

- Alder D, Synnott T (1992) Permanent sample plot techniques for mixed tropical forest. Oxford Forestry Institute, University of Oxford.
- Álvarez IA, Gallo BC, Garçon EAM, Oshiro OT (2015) Street tree inventory of Campinas, Brazil: an instrument for urban forestry management and planning. *Arboric Urban Forestry* 41(5):233–244
- Angiosperm Phylogeny Group (2016) An update of the Angiosperm Phylogeny Group classification for the orders and families of flowering plants: APG IV. *Bot J Linn Soc* 181(1):1–20. <https://doi.org/10.1111/boj.12385>
- Amberger A, Eder R (2012) The influence of green space on community attachment of urban and suburban residents. *Urban For Urban Green* 11(1):41–49. <https://doi.org/10.1016/j.ufug.2011.11.003>
- Avolio ML, Pataki DE, Gillespie TW, Jenerette GD, McCarthy HR, Pincetl S, Clarke LW (2015) Tree diversity in Southern California's urban forest: the interacting roles of social and environmental variables. *Front Ecol Evol*. <https://doi.org/10.3389/fevo.2015.00073>
- Balansiefer M, Wiecheteck M (1985) *Arborização de cidades*. 1st edn. Curitiba: Instituto de Terras e Cartografia.
- Beckett KP, Smith PF, Taylor G (2000) Effective tree species for local air quality management. *J Arboric*. 26
- Bigelow LM, Fahey RT, Grabosky J, Hallett RA, Henning JG, Johnson ML, Roman LA (2024) Predictors of street tree survival in Philadelphia: Tree traits, biophysical environment, and socioeconomic context, vol 94. *Urban Forestry & Urban Greening*, p 128284. <https://doi.org/10.1016/j.ufug.2024.128284>
- Bille RA, Jensen KE, Buitenwerf R (2023) Global patterns in urban green space are strongly linked to human development and population density. 86:127980. *Urban Forestry & Urban Greening* <https://doi.org/10.1016/j.ufug.2023.127980>
- Biswas K, Chattopadhyay I, Banerjee RK, Bandyopadhyay U (2002) Biological activities and medicinal properties of neem (*Azadirachta indica*). *Current Sci* 1336–1345.
- Blum J, Ayub RA, Barbosa MB (2008) Época de colheita e qualidade pós-colheita do caqui cv. Fuyu com a aplicação pré-colheita de ácido giberélico e aminoetoxivinilglicina. *Biotemas* 21(4):15–19.
- Burghardt KT, Tallamy DW (2013) Plant origin asymmetrically impacts feeding guilds and life stages driving community structure of herbivorous arthropods. *Divers Distrib* 19:1553–1565. <https://doi.org/10.1111/ddi.12122>
- Burghardt KT, Tallamy DW, Shriver WG (2009) Impact of native plants on bird and butterfly biodiversity in suburban landscapes. *Conserv Biol* 23:219–224
- Cariñanos P, Delgado-Capel M, Maradiaga-Marín MF, Benítez G (2019) Considerations on the allergy-risks related to the consumption of fruits from urban trees in Mediterranean cities, vol 45. *Urban Forestry & Urban Greening*, p 126303. <https://doi.org/10.1016/j.ufug.2019.03.007>
- Chao A, Gotelli NJ, Hsieh TC, Sander EL, Ma KH, Colwell RK, Ellison AM (2014) Rarefaction and extrapolation with Hill numbers: a framework for sampling and estimation in species diversity studies. *Ecol Monogr* 84:45–67. <https://doi.org/10.1890/13-0133.1>
- CI-BRASIL (Conservation International Brazil) (2009) *Corredor Da Biodiversidade do Amapá/Conservação Internacional do Brasil*. Conservation International, Belém
- Cilliers SS, Cilliers JJ, Lubbe R (2019) Assessing the value of fruit trees in urban green spaces: a South African case study. *Urban Ecosyst* 22(1):49–61
- Clarke LW, Jenerette GD, Davila A (2013) The luxury of vegetation and the legacy of tree biodiversity in Los Angeles, CA. *Landscape and Urban Planning*. 116:48–59. <https://doi.org/10.1016/j.landurbplan.2013.04.006>
- R Core Team (2022) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <https://www.R-project.org/>
- Cornelis J, Hermy M (2004) Biodiversity relationships in urban and suburban parks in Flanders. *Landscape and Urban Planning*, Volume 69, Issue 4, Pages 385–401. <https://doi.org/10.1016/j.landurbplan.2003.10.038>
- Donovan GH, Butry DT (2010) Trees in the city: valuing street trees in Portland, Oregon. *Landsc Urban Plann* 94(2):77–83
- dos Santos AR, da Rocha CFD, Bergallo HG (2010) Native and exotic species in the urban landscape of the city of Rio De Janeiro, Brazil: density, richness, and arboreal deficit. *Urban Ecosyst* 13:209–222. <https://doi.org/10.1007/s11252-009-0113-z>
- Escobedo FJ, Clerici N, Staudhammer CL, Corzo GT (2015) Socio-ecological dynamics and inequality in Bogotá, Colombia's public urban forests and their ecosystem services. *Urban Forestry & Urban Greening*, 14(4) 1040–1053. <https://doi.org/10.1016/j.ufug.2015.09.011>
- Fan C, Johnston M, Darling L, Scott L, Liao FH (2019) Land use and socio-economic determinants of urban forest structure and diversity. *Landscape and Urban Planning*. 181:10–21. <https://doi.org/10.1016/j.landurbplan.2018.09.012>
- Fang F, Greenlee AJ, He Y, Eutsler E (2023) Evaluating the quality of street trees. In: Washington DC (ed) *Implications for environmental justice*, vol 85. *Urban Forestry & Urban Greening*, p 127947. <https://doi.org/10.1016/j.ufug.2023.127947>
- Fleming A, Steenberg J (2023) The equity of urban forest change and frequency in Toronto, ON. *Urban Forestry Urban Green* 90:128153. <https://doi.org/10.1016/j.ufug.2023.128153>
- Gaudon JM, McTavish MJ, Hamberg J et al (2022) Noise attenuation varies by interactions of land cover and season in an urban/peri-urban landscape. *Urban Ecosyst* 25:811–818. <https://doi.org/10.1007/s11252-021-01194-4>
- Heleno RH, Ceia RS, Ramos JA, Memmott J (2009) Effects of alien plants on insect abundance and biomass: a food-web Approach. *Conserv Biol* 23:410–419
- Heynen N, Perkins HA, Roy P (2006) The Political Ecology of Uneven Urban Green Space: the impact of Political Economy on Race and Ethnicity in Producing Environmental Inequality in Milwaukee. *Urban Affairs Rev* 42(1):3–25. <https://doi.org/10.1177/1078087406290729>
- Hope D, Gries C, Zhu W, Fagan W.F., Redman C.L., Grimm N.B., Nelson A.L., Martin C, Kinzig A (2003) Socioeconomics drive urban plant diversity. *Proc Nat Acad Sci USA* 100:8788–8792
- Hsieh TC, Ma KH, Chao A (2020) iNEXT: iNterpolation and EXTrapolation for species diversity. R package version 2.0.20 URL: <http://chao.stat.nthu.edu.tw/wordpress/software-download/>
- IBGE Census (2022) Brazilian Institute of Geography and Statistics. 2022. <https://censo2022.ibge.gov.br/>
- IBGE. Census (2010) Brazilian Institute of Geography and Statistics. <https://censo2010.ibge.gov.br/resultados.html>
- IBGE Brazilian Institute of Geography and Statistics (2018) <https://cidades.ibge.gov.br/brasil/ap/macapa/panorama>
- IPCC (2023) In: Lee H, Romero J (eds) *Climate Change 2023: synthesis report. Contribution of Working groups I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* [Core writing Team. IPCC, Geneva, Switzerland, pp 35–115. doi: <https://doi.org/10.59327/IPCC/AR6-9789291691647>.

- IPEA Instituto de Pesquisa Economica Aplicada. 2022. *Estimativa da População em Situação de Rua no Brasil (2012–2022)*. Brasília: Ministério do Planejamento e Orçamento. <https://doi.org/10.38116/ntdisoc103>
- IPHAN Instituto do Patrimônio Histórico e Artístico Nacional. (2020). *Tombamento* <https://portal.iphan.gov.br/tombamento>
- Jenerette G, Harlan S, Brazel A et al (2007) Regional relationships between surface temperature, vegetation, and human settlement in a rapidly urbanizing ecosystem. *Landscape Ecol* 22:353–365. <https://doi-org.ez7.periodicos.capes.gov.br/https://doi.org/10.1007/s10980-006-9032-z>
- Jettanacheawchankit S et al (2020) Guava (*Psidium guajava* L.) Fruit Chemistry, Nutritional Value, and Health benefits. *Fruit crops: diagnosis and management of nutrient constraints*. Springer, pp 415–437
- Koury MF, de Souza Pereira L, Silva JRC (2017) História e memória das árvores de Belém: A mangueira e a castanheira. *Revista Amazônia Science & Health* 5(2):14–25.
- Kowalski JM, Conway TM (2023) The routes to fruit: Governance of urban food trees in Canada, *Urban Forestry & Urban Greening*, Volume 86, 128045, <https://doi.org/10.1016/j.ufug.2023.128045>
- Lee AC, Maheswaran R (2011) The health benefits of urban green spaces: a review of the evidence. *J Public Health* 33(2):212–222.
- Li G, Weng Q (2007) Measuring the quality of life in City of Indianapolis by Integration of Remote Sensing and Census Data. *Int J Remote Sens* 28:249–267. <https://doi.org/10.1080/01431160600735624>
- Lin W, Guneralp B (2024) Influence of urban-form and sociodemographic factors on street tree abundance transpire across scales in Houston, TX, *Urban Forestry & Urban Greening*. 94:128263. <https://doi.org/10.1016/j.ufug.2024.128263>
- Lin J, Wang Q, Li X (2021) Socioeconomic and spatial inequalities of street tree abundance, species diversity, and size structure in New York City, *Landscape and Urban Planning*. 206:103992. <https://doi.org/10.1016/j.landurbplan.2020.103992>
- Lorenzi H (2002) *Árvores brasileiras: manual de identificação e cultivo de plantas arbóreas do Brasil*. 2.ed. Nova Odessa, Instituto Plantarum 2:384.
- Lowry JH, Baker ME, Ramsey RD (2012) Determinants of urban tree canopy in residential neighborhoods: Household characteristics, urban form, and the geophysical landscape. *Urban Ecosystem* 15:247–266. <https://doi-org.ez7.periodicos.capes.gov.br/https://doi.org/10.1007/s11252-011-0185-4>
- Mao L, Yong Gao Y, Sun W (1993) Influences of street tree systems on summer micro-climate and noise attenuation in Nanjing city, China. *Arboricultural J* 17(3):239–251. <https://doi.org/10.1080/03071375.1993.9746972>
- Marto GBT (2005) Identificação de espécies florestais: *Tabebuia heptaphylla* (Ipê-roxo). IPEF - Instituto de Pesquisa e Estudos Florestais
- McLain R, Poe M, Hurley PT, Lecompte-Mastenbrook J, Emery MR (2012) Producing edible landscapes in Seattle's urban forest, *Urban Forestry & Urban Greening*, Volume 11, Issue 2, Pages 187–194, <https://doi.org/10.1016/j.ufug.2011.12.002>
- McNeely JA, Mooney HA, Neville LE, Schei P, Waage JK (eds) (2001) *A global strategy on invasive alien species*. IUCN Gland, Switzerland and Cambridge, in collaboration with the Global Invasive Species Programme, pp 50
- McPherson EG, Doorn NV, Goede J (2016) Structure, function and value of street trees in California, USA, *Urban Forestry & Urban Greening*, V.17, 2016, Pages 104–115, <https://doi.org/10.1016/j.ufug.2016.03.013>
- Moro MF, Castro ASF (2015) A checklist of plant species in the urban forestry of Fortaleza, Brazil: where are the native species. *Ctry Megadiversity? Urban Ecosystem* 18:47–71. <https://doi.org/10.1007/s11252-014-0380-1>
- Morton J (1987) Mangos. Pp. 221–239 in *Fruits of Warm Climates*. University of Miami, Miami, Florida.
- Mouco, MAC (2015) *Cultivo da mangueira*. 3rd edn. Petrolina: Embrapa Semiárido.
- Nagendra H, Gopal D (2011) Tree diversity, distribution, history and change in urban parks: studies in Bangalore, India. *Urban Ecosystem* 14, 211–223 (2011). <https://doi.org/10.1007/s11252-010-0148-1>
- Nowak DJ, Greenfield EJ (2012) Tree and impervious cover in the United States. *Landscape and Urban Planning*, v.107, 21–30. <https://doi-org.ez7.periodicos.capes.gov.br/https://doi.org/10.1016/j.landurbplan.2012.04.005>
- Nowak DJ, Hirabayashi S, Bodine A, Greenfield E (2014) Tree and forest effects on air quality and human health in the United States. *Environ Pollut* 193:119–129. <https://doi.org/10.1016/j.envpol.2014.05.028>
- Oksanen J, Blanchet FG, Kindt R, Legendre P, Minchin P, O'Hara RB, Solymos P, Stevens MHH, Szoecs E, Wagner H, Barbour M, Bedward M, Bolker B, Borcard D, Carvalho G, Chirico M, De Caceres M, Durand S, Evangelista H, FitzJohn R, Friendly M, Furneaux B, Hannigan G, Hill M, Lahti L, McGlenn D, Ouellette M, Ribeiro Cunha E, Smith T, Stier A, Ter Braak C, Weedon J (2022) *vegan: Community Ecology Package*. R package version 2.6-2. <https://CRAN.R-project.org/package=vegan>
- Osako LK, Takenaka EMM, da Silva PA (2016) Arborização urbana e a importância do planejamento ambiental através de políticas públicas. *Revista Científica ANAP Brasil* 9(14).
- Pacheco J (2019) 24 de Janeiro). *Venda de manga verde com sal a R\$ 2 em Macapá chama a atenção de pedestres e motoristas*. <https://g1.globo.com/ap/amapa/noticia/2019/01/24/venda-de-manga-verde-com-sal-a-r-2-em-macapa-chama-a-atencao-de-pedestres-e-motoristas.ghtml>
- Paradis E, Schliep K (2019) Ape 5.0: an environment for modern phylogenetics and evolutionary analyses. *R Bioinform* 35:526–528. <https://doi.org/10.1093/bioinformatics/bty633>
- Patro R (2018) Oiti – *Licania tomentosa*. Site oficial Jardineiro.net. Disponível em: <http://www.jardineiro.net/plantas/oiti-licania-tomentosa.html>. Acesso em 10/11/18
- Pauleit S, Andersson E, Anton B, Buijs A, Haase D, Hansen R, Kowarik I, Olafsson AS, Jagt SV (2019) Urban green infrastructure – connecting people and nature for sustainable cities, *Urban Forestry & Urban Greening*, Volume 40, Pages 1–3, <https://doi.org/10.1016/j.ufug.2019.04.007>
- Peckham SC, Duinker PN, Ordóñez C (2013) Urban Forest values in Canada: Views of citizens in Calgary and Halifax, *Urban Forestry & Urban Greening*, Volume 12, Issue 2, Pages 154–162, <https://doi.org/10.1016/j.ufug.2013.01.001>
- Pedlowski MA, Da Silva VAC, Adell JJC et al (2002) Urban forest and environmental inequality in Campos Dos Goytacazes, Rio De Janeiro, Brazil. *Urban Ecosystem* 6:9–20. <https://doi.org/10.1023/A:1025910528583>
- Pimental D, Zuniga R, Morrison D (2005) Update on the environmental and economic costs associated with alien-invasive species in the United States, *Ecological Economics*, Volume 52, Issue 3, Pages 273–288, <https://doi.org/10.1016/j.ecolecon.2004.10.002>
- PMB Prefeitura Municipal de Belém (2021) Patrimônio Cultural de Belém. Disponível em: <http://www.belem.pa.gov.br/pagina/78>
- Poe MR, McLain RJ, Emery M, Hurley PT (2013) Urban forest justice and the rights to wild foods, medicines, and materials in the city. *Human Ecol* 41:409–422.
- Pysek et al (2020) Scientists warning on invasive alien species. *Biol Rev* 95:1511–1534. <https://doi.org/10.1111/brv.12627>
- Qian Y, Zhou W, Pickett ST, Yu W, Xiong D, Wang W, Jing C (2020) Integrating structure and function: mapping the hierarchical spatial heterogeneity of urban landscapes. *Ecol Proc* 9:1–11.
- Rede Penssan *Rede Brasileira de Pesquisa em Soberania e Segurança Alimentar e Nutricional* (2022). 'II Inquérito Nacional sobre

- Insegurança Alimentar no Contexto da Pandemia da COVID-19 no Brasil! [livro eletrônico]: II Vigisan: relatório final. São Paulo, SP: Fundação Friedrich Ebert, Rede Penssan. Disponível em: <https://olheparaafome.com.br/wp-content/uploads/2022/06/Relatorio-II-VIGISAN-2022.pdf> Acesso em: agosto, 2022
- Ren X, Torquato PR, Arndt SK, Forestry U (2023) & Urban Greening, vol 81, 127860, <https://doi.org/10.1016/j.ufug.2023.127860>
- Rocha MF, Nucci JC (2018) Índices de vegetação e competição entre cidades. *Geosp – Espaço e Tempo* (Online), v. 641–655
- Roman LA et al (2020) Disparities in urban tree canopy cover: an environmental equity perspective, vol 50. *Urban Forestry & Urban Greening*, p 126635
- Santos ROD, Soares RN, Silva BMDSE (2019) Compositional similarity of urban green areas in southeastern Brazil. *Floresta e Ambiente* 26(2):e20170747
- Schwarz K, Fragkias M, Boone CG, Zhou W, McHale M, Grove JM, et al. (2015) Trees grow on money: urban tree canopy cover and environmental justice. *PLoS one* 10(4):e0122051.
- Shams ZI, Shahid M, Nadeem Z, Naz S, Raheel D, Aftab D, Fraz TR, Roomi MS (2020) Town socio-economic status and road width determine street tree density and diversity in Karachi, Pakistan, vol 47. *Urban Forestry & Urban Greening*, p 126473. <https://doi.org/10.1016/j.ufug.2019.126473>
- Shi, F., Meng, Q., Pan, L., & Wang, J. (2023). Root damage of street trees in urban environments: An overview of its hazards, causes, and prevention and control measures. *Sci Tot Environ*, 166728.
- Shiraishi K (2022) The inequity of distribution of urban forest and ecosystem services in Cali, Colombia. *Urban For Urban Green* 67:127446
- Sikuzani YU, Malaisse F, Kaleba SC, Mwanke AK, Yamba AM, Khonde CN, Bogaert J, Kankumbi FM (2019) Tree diversity and structure on green space of urban and peri-urban zones: The case of Lubumbashi City in the Democratic Republic of Congo, *Urban Forestry & Urban Greening*, Volume 41, 2019, Pages 67–74, <https://doi.org/10.1016/j.ufug.2019.03.008>
- Silva JLS, Oliveira MTP, Oliveira W, Borges LA, Cruz-Neto O, Lopes AV (2020) High richness of exotic trees in tropical urban green spaces: Reproductive systems, fruiting and associated risks to native species. *Urban Forestry & Urban Greening*, 50, 2020, 126659, <https://doi.org/10.1016/j.ufug.2020.126659>
- Silva Junior, MC (2005) 100 árvores do cerrado: guia de campo/ Manoel da Silva Junior; colaboradores Gilmar Correia dos Santos ...[et al.]. – Brasília, Ed. Rede de Sementes do Cerrado, 278p.
- Singh RK, Das AK, Mishra A, Sharma V (2020) A comprehensive review on medicinal fruit trees. *Biocatal Agric Biotechnol* 24:101551
- Smith SA, Brown JW (2018) Constructing a broadly inclusive seed plant phylogeny. *Am J Bot* 105(3):302–314. <https://doi.org/10.1002/ajb2.1019>
- Soares ACS, Santos RO, dos, Soares RN, Cantuaria PC, de Lima RB, Silva BMS (2021) Paradox of Afforestation in cities in the Brazilian Amazon: an understanding of the composition and floristic similarity of these urban green spaces. 66:1618–8667. *Urban Forestry & Urban Greening* <https://doi.org/10.1016/j.ufug.2021.127374>
- Shapit P (2019) Guava: an ancient tropical fruit with modern benefits. *Int J Fruit Sci* 19(2):186–198
- Sujarwo W, Keim AP, Caneva G, Toniolo C, Nicoletti M (2016) Ethnobotanical uses of neem (*Azadirachta indica* A. Juss.; Meliaceae) leaves in Bali (Indonesia) and the Indian subcontinent in relation with historical background and phytochemical properties. *J Ethnopharmacol* 189:186–193
- Tavares JP (2004) N. Características Da Climatologia E Macapá-Ap. *Caminhos De Geografia* v 15:138–151
- Taylor MS, Wheeler BW, White MP, Economou T, Osborne NJ (2015) Research note: urban street tree density and antidepressant prescription rates—A cross-sectional study in London, UK, *Landscape and Urban Planning*. 136:174–179. <https://doi.org/10.1016/j.landurbplan.2014.12.005>
- Tzoulas K, Korpela K, Venn S, Yli-Pelkonen V, Kaźmierczak A, Niemela J, James P (2007) Promoting ecosystem and human health in urban areas using Green Infrastructure: a literature review. *Landsc Urban Plan* 81(3):167–178.
- Useni SY, Sambiéni KR, Maréchal J, Ilunga WIE, Malaisse F, Bogaert J, Munyemba KF (2018) Changes in the spatial pattern and ecological functionalities of Green spaces in Lubumbashi (the Democratic Republic of Congo) in relation with the degree of urbanization trop. *Conserv Sci* 11. <https://doi.org/10.1177/1940082918771325>
- Useni YS, Malaisse F, Yona JM, Mwamba TM, Bogaert J (2021) Diversity, use and management of household-located fruit trees in two rapidly developing towns in Southeastern D.R. Congo. *Urban Forestry Urban Green* 63:127220. <https://doi.org/10.1016/j.ufug.2021.127220>
- Venables WN, Ripley BD (2002) *Modern applied statistics with S*. fourth edition. Springer, New York. ISBN 0-387-95457-0
- Vieira TA, Panagopoulos T (2020) *Urban Forestry in Brazilian Amazonia*. Sustainability 12:3235. <https://doi.org/10.3390/su12083235>
- Vilanova C, Ferran JS (2024) Integrating landscape ecology in urban green infrastructure planning: A multi-scale approach for sustainable development, *Urban Forestry & Urban Greening*, Volume 94, 2024, 128248, <https://doi.org/10.1016/j.ufug.2024.128248>
- Vujcic M, Tomicevic-Dubljevic J (2018) Urban Forest benefits to the younger population: the case study of the city of Belgrade, Serbia, *Forest Policy and Economics*. 96:54–62. <https://doi.org/10.1016/j.forpol.2018.08.006>
- Walhowe JL (2022) Land manager decision-making practices when establishing public fruit-bearing plants in Hennepin county municipalities, Minnesota, *Urban Forestry & Urban Greening*. 74:127659. <https://doi.org/10.1016/j.ufug.2022.127659>
- Wilcove DS, Rothstein D, Dubow J, Phillips A, Losos E (1998) Quantifying threats to imperiled species in the United States. *BioScience* 48(8):607–615.
- Williams NS, McDonnell MJ, Seager TP (2005) Factors influencing plant selection in revegetation projects in an Australian urban landscape. *Landsc Urban Plan* 71(2–4):425–443.
- Wolf KL (2013) Tree cover, income inequality, and housing prices in an urban landscape. *Urban Forestry Urban Green* 12(4):500–508
- Zuur A, Ieno EN, Walker N, Saveliev AA, Smith GM (2009) *Mixed effects models and extensions in Ecology with R*. Springer, New York

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

Springer Nature or its licensor (e.g. a society or other partner) holds exclusive rights to this article under a publishing agreement with the author(s) or other rightsholder(s); author self-archiving of the accepted manuscript version of this article is solely governed by the terms of such publishing agreement and applicable law.