

The urban forest as a mirror of inequality: plant diversity, ecosystem services, and social valuation in Xalapa, Mexico

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Abstract

Urban forests in the Global South are shaped by socioeconomic inequalities conditioning their composition, ecological function, and social valuation. This study examined how a marginalization gradient structures tree diversity, vascular epiphyte (VE) richness, ecosystem services (ES), and community perception across 20 sites in Xalapa, Mexico. A total of 685 trees (115 species) and 60 VE species were recorded; ES were quantified with i-Tree Eco; and 164 structured interviews were conducted. Native tree richness was statistically equivalent throughout the gradient, whereas exotic richness was 2–2.7× greater in the least marginalized strata—a pattern termed the “garden effect.” A negative binomial generalized linear mixed model (GLMM) showed that tree diameter at breast height (DBH; $IRR = 2.94$; 95% CI : 2.81–3.08) is the primary determinant of VE richness, and that exotic trees harbored 17% lower VE richness than natives ($IRR = 0.83$; 95% CI : 0.71–0.97). Marginalization exerted no direct linear effect on VE richness owing to spatial confounding with land use. ES provision showed no significant gradient differences ($p > 0.05$; marginal $R^2 \leq 0.032$); however, social valuation shifted from aesthetic-recreational at low-marginalization sites to utilitarian and subsistence-oriented at the most vulnerable site. Three disservice categories were identified: (1) perceived physical risk from branch or tree collapse; (2) misclassification of VEs as parasitic plants by 15.3% of respondents, promoting unnecessary phytosanitary pruning; and (3) elevated biogenic Volatile Organic Compounds (VOC) emissions at less marginalized sites, constituting potential tropospheric ozone formation hotspots. These findings underscore the need for differentiated, equitable urban forest management in cities of the Global South.

1. Introduction

Rapid urbanization constitutes one of the most consequential land transformation processes affecting biodiversity and ecosystem services at the global scale (McKinney 2002). In this context, the urban forest—understood as the ensemble of trees and woody cover present in public and private spaces along the urban–peri-urban–rural continuum—has gained increasing relevance as both a research subject and a focus of public policy (Konijnendijk et al. 2006; Salbitano et al. 2016; Endreny 2018). Its ecological functions encompass local climate regulation, carbon storage and sequestration, stormwater interception, and the sustenance of biotic communities including vertebrate fauna, invertebrates, and epiphytic flora (Roy et al. 2012; de Groot et al. 2002). The distribution of these benefits within cities, however, is not homogeneous: it is conditioned by the socioeconomic structure of the territory, which renders the urban forest a material expression of socio-environmental inequality.

The relationship between socioeconomic status and urban biodiversity has been documented under the “luxury effect,” whereby higher-income strata concentrate greater richness and diversity of woody plants in their residential environments (Hope et al. 2003; Leong et al. 2018). This pattern, initially identified in cities of North America and Europe, has begun to be examined in cities of the Global South, where socioeconomic gradients are more pronounced, the urban–rural transition is continuous, and pre-existing ecological heterogeneity imposes distinct dynamics on plant communities (Escobedo et al.

2015; Dobbs et al. 2019). Urbanization is moreover a primary driver of biotic homogenization (McKinney 2006; Grimm et al. 2008; Pickett et al. 2011), and in Global South cities, urban forest composition reflects not only inequalities in resource access but also the divergent values and needs that distinct social groups project onto vegetation—ranging from aesthetic and ornamental criteria to direct provisioning functions linked to subsistence (Shackleton et al. 2015, 2024). The mismatch between existing and community-desired vegetation thus constitutes a form of socio-environmental injustice that the literature has only incipiently characterized in Latin American contexts. Indeed, quantitative assessments have confirmed that urban tree canopy cover and access to green vegetation correlate more closely with income and educational attainment than with ecological need (Schwarz et al. 2015; Nesbitt et al. 2019).

This dimension of socio-environmental injustice extends beyond tree composition and the mismatch between existing and desired vegetation: it encompasses the biodiversity that trees themselves sustain as structural hosts. Vascular epiphytes (VEs)—ferns, bromeliads, orchids, and allied plants that complete their life cycle on trees without parasitizing them—constitute one of the groups most sensitive to host tree structural quality and to management practices (Zotz 2013; Krömer et al. 2016). Their richness in urban environments has been documented in tropical cities as an indicator of tree disturbance and landscape permeability toward natural vegetation fragments (Izuddin & Webb 2015; Jarquin-Pacheco et al. 2025). Nevertheless, the integrated analysis of tree health condition, native versus exotic species composition, pruning practices associated with residents' socioeconomic level, and VE richness along socioecological gradients has not been addressed previously.

Alongside the biodiversity it sustains, the urban forest provides a suite of ecosystem services whose distribution also responds to the socioeconomic structure of the territory. Trees regulate microclimate, store and sequester carbon, intercept stormwater, and contribute to air quality; the magnitude of these benefits depends on species composition, canopy structure, and tree condition (Roy et al. 2012; Endreny 2018). In Global South cities, access to these services is stratified: higher-income sectors tend to concentrate greater provision, while marginalized sectors face deficits that compound their vulnerability to climate extremes (Escobedo et al. 2015; Dobbs et al. 2019). The standardized quantification of these services has relied on tools such as i-Tree Eco; however, their application in tropical contexts is subject to documented methodological limitations arising from the use of allometric equations calibrated for temperate species, which introduces uncertainty into the estimation of absolute values (Nowak 2021; Sharma et al. 2025). Beyond the biophysical dimension, community valuation of ecosystem services is heterogeneous and depends on local needs and cultural frameworks, such that ecological provision and its social recognition do not necessarily coincide (Shackleton et al. 2015, 2024).

To the foregoing must be added the dimension of urban forest disservices—a component historically underrepresented in the literature relative to the emphasis placed on benefits. Trees generate perceived negative impacts including physical risk from falling branches or trunks, damage to infrastructure, and the presence of parasitic plants; these impacts vary according to socioeconomic context and housing vulnerability (Roman et al. 2021; Martin et al. 2024). Furthermore, trees produce biogenic emissions of volatile organic compounds (VOCs) that, in the presence of anthropogenic nitrogen oxides (NO_x) from

vehicular traffic, can contribute to tropospheric ozone formation—a secondary pollutant with documented adverse effects on human respiratory health and plant productivity (Calfapietra et al. 2013; Gebreyesus et al. 2024). This chemical disservice operates independently of public perception, creating a gap between biophysical reality and community narrative that has rarely been quantified in tropical cities.

Xalapa, Veracruz, Mexico, represents a suitable study system for examining these dynamics. It is a mid-sized city (~ 500,000 inhabitants) with high internal socioeconomic heterogeneity, located at approximately 1,400 m above sea level in a region of high biological diversity with remnants of cloud forest (bosque mesófilo de montaña), oak woodland (encinar), and tropical dry forest (Castillo-Campos 1991). Its accelerated urbanization has generated documented losses of natural vegetation cover (Lemoine-Rodríguez et al. 2019), yet numerous sites retaining tree cover in diverse configurations—parks, median strips, vacant lots, protected natural areas, and private gardens—persist within the urban matrix and harbor arboreal and epiphytic biodiversity (Jarquin-Pacheco et al. 2025), whose ecological function and social valuation in relation to socioeconomic context remain insufficiently characterized.

This study integrates four analytical components along a socioeconomic marginalization gradient across 20 sites in the municipality of Xalapa: (1) tree composition and diversity and its correspondence with citizen preferences; (2) the determinants of vascular epiphyte richness associated with urban trees; (3) the quantification of ecosystem services using i-Tree Eco and the characterization of their social valuation; and (4) the identification of perceived and overlooked disservices. The approach adopted challenges the assumption that socioeconomic marginalization entails a linear and homogeneous loss of ecological function, and provides empirical evidence for the design of differentiated, equitable, and ecologically grounded urban forest management strategies in tropical Latin American cities.

2. Materials and Methods

2.1. Study site

The municipality of Xalapa covers an area of 128.87 km² and is located between 19°29′–19°36′ N and 96°48′–96°58′ W, at elevations ranging from approximately 1,200 to 1,560 m above sea level (m asl), in the central portion of the state of Veracruz, Mexico (Gobierno del Estado de Veracruz 2021). The prevailing climate is humid to sub-humid semi-warm, with precipitation distributed throughout the year; mean annual precipitation ranges from 1,400 to 1,700 mm and mean annual temperature is approximately 18°C (City Adapt 2022). The original vegetation comprised cloud forest (bosque mesófilo de montaña), oak woodland (encinar), conifer forest, tropical dry forest, and riparian vegetation (Castillo-Campos 1991).

2.2. Land use map and site selection

A land use map was produced through supervised classification (Campbell et al. 2015) of Planet satellite imagery (3 m spatial resolution, May 2023), achieving an overall classification accuracy of 82.1%. Four

land use categories were distinguished: tree canopy cover, grassland/cropland, human settlements, and water bodies. Twenty 100 × 100 m plots were randomly selected across four tree canopy cover strata (0–25%, 25–50%, 50–75%, and 75–100%), with five sites per stratum and a minimum inter-site distance of 1 km. Spatial independence among sites was verified prior to analysis using Moran's I ($I = 0.020$, $z = 0.461$, $p = 0.64$), confirming the absence of significant spatial autocorrelation (Fig. 1).

For each plot, the normalized marginalization index (Gobierno de México 2021) was assigned according to the Basic Geostatistical Area (AGEB) in which the plot centroid fell (INEGI, 2020). This index served as an emergent characterization of the socioeconomic context of each site; the resulting distribution across strata was as follows: 6 sites of very low marginalization, 10 of low marginalization, 3 of medium marginalization, and 1 of very high marginalization (Col. Independencia). The very high marginalization stratum is treated as exploratory-descriptive throughout; values reported for this stratum are not generalizable, and future studies should incorporate multiple sites under conditions of high vulnerability.

2.3. Biodiversity, ecosystem services, and tree condition

2.3.1. Tree diversity and vascular epiphyte richness

Three 20 × 20 m subplots were established within each 100 × 100 m plot (60 subplots in total), placed to maximize coverage of wooded areas. All trees with height ≥ 1.5 m and diameter at breast height (DBH) ≥ 5 cm were inventoried. For each tree, the presence and identity of vascular epiphyte species were recorded up to a height of 8 m using direct visual observation (Krömer et al. 2007); the presence of hemiparasitic plant species (Loranthaceae and Viscaceae) was recorded separately from epiphytes. This protocol explicitly distinguished epiphytes from parasitic plants to avoid misclassification.

2.3.2. Ecosystem services using i-Tree Eco

For all inventoried trees, the following dendrometric variables were recorded: height, DBH, crown base height, percentage of missing crown, percentage of dieback, crown length and width, and crown exposure. These data were entered into i-Tree Eco to quantify five categories of urban forest effects: carbon storage, annual carbon sequestration, oxygen production, stormwater runoff control, and biogenic volatile organic compound (VOC) emissions. VOC emissions are treated herein as a potential disservice (see § 3.4), as they may contribute to tropospheric ozone formation in the presence of sufficient anthropogenic NO_x.

Although the application of i-Tree Tools has documented limitations for tropical taxa (Nowak 2021; Sharma et al. 2025), the United States Forest Service (USFS) adapted the suite for Mexico, covering 18 metropolitan areas (MAs), including Xalapa (i-Tree Tools n.d.). Crucially, because potential tool limitations affect all study sites and marginalization strata systematically, relative comparisons across strata remain internally valid despite uncertainties in the resulting values.

2.3.3. Tree condition

The phytosanitary and structural condition of each inventoried tree was assessed using 15 evaluative criteria encompassing canopy completeness, presence of decay, structural defects, and overall vigor (Martin et al. 2014). Each tree was classified into one of three condition categories: “excellent,” “good,” or “poor.”

2.4. Social perception of the urban forest

A structured interview instrument was designed to assess: (a) the use and perceived value of immediately surrounding wooded areas; (b) perceived benefits of trees; (c) management practices carried out by residents; and (d) the ability to distinguish parasitic plants from vascular epiphytes. Using a universe of 933 households ($N = 933$), the minimum required sample size was calculated as 164 interviews ($Z = 1.96$, precision $e = 0.07$, $p = 0.5$; Cochran 1977), which were distributed proportionally across the 20 sampled sites, with one interview per household.

The instrument was validated through a pilot test applied to 5% of the projected sample prior to full deployment (Heinemann 2008). To ensure maximum consistency in application and eliminate inter-interviewer variability, all interviews were conducted exclusively by the lead author between April and October 2024 (Rosenblatt 2012). Where complications of access or resident availability prevented probabilistic household selection, final selection was carried out using non-probabilistic convenience sampling (Etikan et al. 2016); inferences based on interview data are therefore interpreted at the descriptive level.

The floristic species list identified by residents as preferred was constructed from open-ended responses during the interviews and subsequently matched to the taxonomic inventory. The degree of overlap between the field inventory and citizen preferences was quantified using the Jaccard similarity index ($J = |A \cap B| / |A \cup B|$), applied separately for each marginalization stratum.

2.5. Statistical analyses

Alpha tree diversity (orders $q = 0, 1, 2$) was estimated using Hill numbers standardized by sampling coverage, with individual-based rarefaction and extrapolation curves (95% bootstrap confidence intervals) computed in iNEXT (Hsieh et al. 2016). Prior to modeling, pairwise independence and multicollinearity among site-level predictors were evaluated using Spearman rank correlations (ρ) and Variance Inflation Factors ($VIF < 5$ threshold).

Vascular epiphyte richness per tree ($n = 685$) was analyzed using a negative binomial GLMM (variance/mean = 4.1; proportion of zeros = 59%), fitted in R (R Core Team 2024) using the glmmTMB package (Brooks et al. 2017). The natural logarithm of DBH [$\log(\text{DBH})$], phorophyte origin (native vs. exotic), and tree condition (excellent/good/poor) were included as tree-level fixed effects. Sampling site and an observation-level random effect were included as random terms to account for pseudoreplication and residual overdispersion, respectively. Results are expressed as Incidence Rate Ratios ($IRR = e^{\beta}$), interpreted as the multiplicative change in expected VE richness per unit change in the predictor.

The provision of four i-Tree Eco ecosystem services (carbon storage, carbon sequestration, oxygen production, and stormwater runoff control) was modeled using GLMMs with a Gamma distribution and logarithmic link, fitted in glmmTMB, with degree of marginalization as a fixed effect and site as a random effect. The significance of the fixed factor was determined using the Type III Wald chi-square test. Marginal R^2 (variance explained by fixed effects alone) and conditional R^2 (variance explained by the full model) were calculated following Nakagawa & Schielzeth (2013), and the Intraclass Correlation Coefficient (ICC) was derived to partition within-site and between-site variance. All statistical analyses were conducted at $\alpha = 0.05$.

3. Results

3.1. Tree diversity, community composition, and citizen preferences along the socioeconomic gradient

A total of 685 tree individuals were recorded, distributed across 115 species (86 native, 29 exotic), 85 genera, and 46 families (Appendix A).

3.1.1. Tree diversity by canopy cover stratum and degree of social marginalization

Tree diversity responded strongly to canopy cover stratum. The 75–100% canopy cover stratum exhibited markedly lower species richness ($q_0 = 35.2$; 95% CI : 29.6–40.7) than the three lower-cover strata (q_0 range: 62.3–69.9), with non-overlapping confidence intervals (Table 1). No statistically significant differences were detected among the 0–25%, 25–50%, and 50–75% strata.

Table 1
Tree diversity by tree canopy cover stratum. n , individuals; S , observed richness; $\hat{C}$, sample coverage; q_1 and q_2 , observed Hill numbers; $Chao1$, asymptotic richness; q_0 standardized to common coverage with 95% confidence interval.

Cover	n	S	$\hat{C}$	q_1	q_2	$Chao1$	q_0 stand. [95% CI]
0–25%	155	53	0.846	37.2	28.0	78.1	69.2 [46.2, 92.2]
25–50%	141	58	0.824	42.9	31.1	75.6	69.9 [49.6, 90.2]
50–75%	189	54	0.895	39.5	32.2	73.0	62.3 [39.7, 85.0]
75–100%	200	38	0.960	26.1	19.5	42.0	35.2 [29.6, 40.7]

Along the socioeconomic gradient, the very low and low marginalization strata ($q_0 = 43.5$ and 49.9, respectively) significantly exceeded the medium marginalization stratum ($q_0 = 35.0$), with confidence intervals excluding zero (low – medium: 14.9 [9, 21]; very low – medium: 8.5 [4, 13]), while the very low and low strata were statistically equivalent to each other (difference: 6.4 [– 1, 14]). The very high

marginalization stratum ($n = 1$ site) is reported descriptively and excluded from inferential comparisons (Table 2).

Table 2
Total tree diversity by marginalization level. n , individuals; S , observed richness; $\hat{C}$, sample coverage; q_1 and q_2 , observed Hill numbers; $Chao1$, asymptotic richness; q_0 standardized to common coverage with 95% confidence interval.

Marginalization	n	S	$\hat{C}$	q_1	q_2	$Chao1$	q_0 stand. [95% CI]
Very low	211	57	0.906	37.4	25.9	71.6	43.5 [39, 48]
Low	328	82	0.906	52.3	38.5	109.4	49.9 [44, 56]
Medium	109	35	0.892	25.5	20.0	41.0	35.0
Very high ^a	37	13	0.871	9.6	7.5	15.0	—
^a Stratum represented by a single site (descriptive).							

Partitioning species richness by origin revealed that native species diversity was statistically invariant across all degrees of social marginalization ($q_0 = 18.9$ – 22.3 at a common coverage of $n = 34$; confidence intervals for q_0 , q_1 , and q_2 fully overlapped across strata). In contrast, exotic species diversity declined markedly with increasing marginalization: the very low and low strata harbored 2.0–2.7 times greater exotic diversity ($q_0 = 8.8$ – 10.9 at $n = 18$) than the medium stratum ($q_0 = 4.0$), decreasing to a single exotic species at the very high marginalization site (Fig. 2). This pattern—the addition of exotic flora in affluent sectors without displacement of the native species base—is hereafter termed the “**garden effect**.”

3.1.2. Forest supply versus citizen preference

The overall Jaccard similarity index between the floristic inventory and resident-preferred species was low ($J = 0.20$), indicating limited correspondence between what is present and what the population desires. Similarity varied systematically with socioeconomic context (Table 3): the least marginalized strata showed the highest values (very low: $J = 0.28$; low: $J = 0.22$), with approximately equal preference for native and exotic species ($\sim 50\%$ each). The medium marginalization stratum exhibited marked dissimilarity ($J = 0.15$), with a rising preference for native species. At the very high marginalization site, the low Jaccard value ($J = 0.11$) reflects a reduced local inventory and a pronounced shift toward native species preference (75%), consistent with a demand for utilitarian and subsistence-oriented provisioning services that is not met by the existing urban forest.

Table 3

Forest supply versus citizen preference along the socioeconomic marginalization gradient.

Marginalization level	Citizen preference (% native vs. exotic)	Jaccard similarity index (sampled trees vs. mentioned trees)
Very low (n = 6 sites)	Native: 53% Exotic: 47%	0.28
Low (n = 10)	Native: 54% Exotic: 46%	0.22
Medium (n = 3)	Native: 58% Exotic: 42%	0.15
Very high (n = 1) ^a	Native: 75% Exotic: 25%	0.11
^a <i>Stratum represented by a single site (descriptive).</i>		

3.2. Vascular epiphyte richness along the urban tree gradient

A total of 60 VE species belonging to 35 genera and 10 families were recorded (Appendix B). According to the negative binomial GLMM, VE richness was overwhelmingly determined by host tree size: for each one-standard-deviation increase in log(DBH), expected richness nearly tripled ($IRR = 2.94$; 95% CI : 2.81–3.08). Exotic phorophytes harbored 17% fewer VE species than native phorophytes ($IRR = 0.83$; 95% CI : 0.71–0.97). With respect to tree condition, apparent positive effects were observed for both the “good” ($IRR = 1.21$; 95% CI : 1.05–1.40) and “poor” ($IRR = 2.12$; 95% CI : 1.45–3.08) categories relative to “excellent”; however, these effects were substantially attenuated after controlling for DBH, indicating that the apparent condition effect reflects the larger girth of older, structurally compromised trees rather than a direct facilitative effect of deteriorated bark.

Land use could not be estimated as a tree-level fixed effect owing to structural zero separation in one land use category and confounding with sampling site; VE richness by land use is therefore reported descriptively (Table 4). Livestock grazing sites recorded the highest mean VE richness per tree (4.11 ± 3.79), while residential areas (0.75 ± 1.34), parks (0.42 ± 0.77), and sports courts showed the lowest values.

Table 4
Epiphyte richness per tree by land-use category (mean $\pm$ SD).

Land use	N (trees)	Mean $\pm$ SD (spp./tree)
Livestock farming	93	4.11 $\pm$ 3.79
Institutional	28	2.71 $\pm$ 3.07
Roadway	12	2.33 $\pm$ 3.82
Agriculture	174	1.41 $\pm$ 1.96
Green area	15	1.33 $\pm$ 2.72
School	33	1.30 $\pm$ 1.83
Vacant lot	66	0.96 $\pm$ 1.82
Residential	121	0.75 $\pm$ 1.34
Natural Protected Area (NPA)	108	0.59 $\pm$ 1.41
Park	19	0.42 $\pm$ 0.77
Sports field	16	0.00 $\pm$ 0.00

Tree canopy cover percentage showed a positive but weak association with VE richness ($IRR = 1.2$ per standard deviation increase). The degree of marginalization showed an unstable effect that changed in sign and significance depending on whether land use was included in the model (with land use: $IRR = 1.03$, $p = 0.70$; without land use: $IRR = 0.86$, $p = 0.009$), demonstrating spatial confounding between marginalization and land use type. Accordingly, no independent causal effect of marginalization on VE richness can be inferred.

3.3. Ecosystem services and social valuation

According to the Gamma GLMMs, the degree of marginalization was a non-significant predictor of all four evaluated ecosystem services (carbon storage: $\chi^2 = 2.56$, $p = 0.465$; carbon sequestration: $\chi^2 = 1.18$, $p = 0.757$; oxygen production: $\chi^2 = 1.18$, $p = 0.758$; avoided stormwater runoff: $\chi^2 = 2.82$, $p = 0.421$) (Appendix C). Carbon storage exhibited the highest ICC (0.524), indicating that 52.4% of its total variability is attributable to between-site differences rather than to within-site tree-level variation. Avoided stormwater runoff showed a moderate ICC (0.286), while carbon sequestration and oxygen production displayed the lowest $ICCs$ (0.175 each). Marginal R^2 values across all four services were very low (0.009–0.032), confirming that marginalization level possesses minimal explanatory power over biophysical ES provision in this system.

Social valuation of urban forest benefits differed systematically across socioeconomic strata (Table 5). When resident perceptions were classified within the TEEB/MEA ecosystem service framework, “regulation services” predominated across all strata and increased progressively from the very low

marginalization stratum (46.84%) to medium (56.94%) and very high marginalization (56.86%), driven by unanimous recognition of air quality improvement and shade provision. “Cultural services” (health, aesthetic value, recreation) showed a marked decline from the very low stratum (33.89%) to the very high marginalization site (9.80%). Conversely, “provisioning services” (water supply, fruit production) were mentioned least frequently at very low marginalization sites (8.97%), rising steadily toward higher-marginalization strata (15.69%–18.06%). “Supporting and biodiversity services” reached their greatest representativeness at the very high marginalization stratum (17.65%), where residents notably emphasized soil conservation and local biodiversity protection.

Table 5

Relative frequency of mentions of urban forest benefits grouped by macro-categories of ecosystem services (ES) along the marginalization gradient. ES grouping: Regulating: air quality, climate, shade, rainfall, noise, and climate protection; Cultural: health, recreation, aesthetic, sentimental, cultural, and historical; Provisioning: water, fruit, wood, medicine, and fencing; Supporting/Biodiversity: conservation, animal/plant refuge, soil quality and retention, and crop support.

Ecosystem service category	Marginalization level			
	Very low (%)	Low (%)	Medium (%)	Very high (%)
Regulating services	46.84	49.89	56.94	56.86
Cultural services	33.89	27.18	16.67	9.8
Supporting and Biodiversity services	10.3	7.01	8.33	17.65
Provisioning services	8.97	15.92	18.06	15.69
Total mentions (n)	301	471	73	51
<i>Note: Percentages represent the proportion of mentions relative to the total accumulated per stratum, as respondents could provide multiple answers.</i>				

3.4. Perceived and overlooked environmental disservices

Three categories of disservices were identified across the gradient (Table 6):

- (1) Structural conflict and physical risk.** Concern over “falling risk (branches/trees)” was most frequently reported in the very low (30.8% of respondents) and very high marginalization strata (27.3%), though driven by distinct underlying factors (see § 4.4). “Infrastructure damage” was reported primarily in the low (18.5%) and very low (17.3%) marginalization strata, while “obstruction/interference” was confined to low (9.9%) and medium (5.3%) strata. “Insecurity” was reported by only 1.9% of respondents in the very low stratum.
- (2) Biological disservice (hemiparasitic plants).** The hemiparasitic species *Psittacanthus schiedeanus*, *Struthanthus* sp., and *Phoradendron nervosum* (Loranthaceae/Viscaceae) were recorded, collectively affecting 11.0%–14.7% of sampled trees across the gradient. At the perception level, recognition of phytosanitary damage attributable solely to mistletoe was highest in the low (44.4%) and very high strata

(36.4%). A subset of respondents (15.3% overall) erroneously classified VEs as parasitic plants, with the highest proportion in the medium stratum (21.1%); this misperception was absent at the very high marginalization site.

(3) Biogenic VOC emissions. This chemical disservice was universally unrecognized by residents across all strata. Mean per-tree VOC emissions ranged from a minimum in the low stratum ($134.6 \text{ g year}^{-1} \text{ tree}^{-1}$), associated with low-emitting native species (*Inga vera*, *Trichilia havanensis*), to a maximum at the very high marginalization site ($1,229.3 \text{ g year}^{-1} \text{ tree}^{-1}$), driven by remnant vegetation dominated by *Quercus* spp. Cross-referencing biophysical VOC flux with census-derived vehicular pressure proxies (INEGI 2020) revealed a spatial mitigation pattern: the very low marginalization stratum combines moderately high emissions ($593 \text{ g year}^{-1} \text{ tree}^{-1}$, associated with *Quercus* and *Liquidambar*) with high vehicular density ($0.624 \text{ vehicles household}^{-1}$; 55.4% of households owning a motor vehicle), generating conditions conducive to tropospheric ozone formation (Calfapietra et al., 2013). By contrast, the very high marginalization site—despite its highest absolute VOC flux—is exposed to sparse traffic ($0.129 \text{ vehicles household}^{-1}$; 8.7% of households with a motor vehicle), locally limiting the availability of NO_x required for ozone synthesis.

Table 6

Perceived disservices of the urban forest along the socioeconomic marginalization gradient.

Disservice dimension and category	Marginalization level			
	Very low	Low	Medium	Very high
1. Structural conflict and physical risk^a				
Risk of falling (branches/trees)	30.8%	16.1%	21.1%	27.3%
Damage to infrastructure	17.3%	18.5%	15.8%	9.1%
Obstruction / hindrance	0.0%	9.9%	5.3%	0.0%
Insecurity	1.9%	0.0%	0.0%	0.0%
2. Biological disservice: parasitic plants^a				
Perceived impact from mistletoe only	28.8%	44.4%	31.6%	36.4%
Perceived impact from mistletoe, epiphytes, or other plants	9.6%	12.4%	21.1%	0%
Sampled trees with presence of parasitic plants	31 of 211 (14.7%)	46 of 330 (13.9%)	13 of 109 (11.9%)	5 of 35 (14.3%)
3. Chemical disservice and anthropogenic context				
Average actual VOC emissions (g/year/tree)	592.9	134.6	203.2	1,229.30
Relative vehicular pressure (vehicles/household) ^b	0.624	0.556	0.435	0.129
Household car availability (% of dwellings) ^b	55.4%	46.6%	35.0%	8.7%
Reference samples by stratum				
Total disservice mentions (dimensions 1 and 2) (n)	30	51	10	5
Total respondents per stratum (N)	52	81	19	11
^a Values for the categories in the indicated disservice dimension represent the percentage of respondents relative to the total surveyed in that same stratum. ^b Data estimated as a proxy for potential vehicular intensity based on the national population census (INEGI, 2020).				

4. Discussion

4.1. The “garden effect” and socioecological mismatch in tree composition

Tree composition and diversity in the Xalapa urban forest exhibit a complex socioecological stratification. Sites with lower socioeconomic marginalization show greater total species richness—a result partially consistent with the “luxury effect” documented by Hope et al. (2003) and Leong et al. (2018) primarily in cities of the Global North—but the underlying mechanism in this tropical system differs qualitatively: native species richness remained statistically invariant throughout the entire marginalization gradient. Ecological inequality is instead expressed through the capacity of less marginalized strata to introduce and maintain greater exotic species diversity, a pattern we term the “garden effect.” This mechanism contrasts with patterns reported for Santiago de Chile, where socioeconomic segregation entails the concentration of native species richness in higher-income areas (Guevara et al. 2024), underscoring the context-dependence of urban biodiversity gradients across the Global South.

In Xalapa, the enrichment of exotic species in less marginalized sectors is consistent with patterns reported in other urban contexts, where historical planning legacies (Roman et al. 2018) and household-level decisions driven by aesthetic motivations (Conway 2016), as reflected in the relatively high floristic correspondence between inventory and preferences at very low and low marginalization sites ($J = 0.28$ and $J = 0.22$, respectively). Conversely, in medium and very high marginalization strata, floristic dissimilarity intensifies and preferences shift markedly toward native species (up to 75% at the most vulnerable site). The low Jaccard value at this site ($J = 0.11$) does not indicate unfamiliarity with the local flora; rather, it reflects the exclusion of species capable of providing the subsistence and provisioning services demanded by residents—a pattern of socio-environmental injustice documented across settlements of the Global South (Shackleton et al. 2015, 2024).

4.2. Vascular epiphytes: determinants of richness in the urban tree matrix

The GLMM analysis demonstrated that VE richness is primarily determined by host tree diameter (DBH; $IRR = 2.94$), rather than by poor sanitary condition—a finding that qualifies previous reports attributing greater VE establishment on trees in poor condition to bark roughness, wounds, or cavities (Izuddin & Webb 2015; Zotz 2016). In the present study, trees in poorer condition harbored greater VE richness because they are typically older individuals with greater DBH, not because deteriorated bark per se constitutes a superior substrate. The 17% lower VE richness on exotic phorophytes relative to native ones is mechanistically consistent with the prevalence of smooth or exfoliating bark textures in common exotic urban species (e.g., *Eriobotrya japonica*, *Ligustrum lucidum*), which impede VE seed adhesion and seedling establishment. The preference of VEs for native phorophytes has been similarly reported in tropical urban settings across Africa and South America (Bhatt et al. 2015; Alex et al. 2021; Nieva et al. 2025), suggesting a generalizable pattern with implications for urban biodiversity management.

The absence of a direct linear effect of marginalization on VE richness—demonstrated by the sign reversal and significance shift of the marginalization parameter upon inclusion or exclusion of land use—constitutes evidence of spatial confounding rather than a genuine socioeconomic effect. VE richness at

the site level responds primarily to land use type, with livestock grazing sites functioning as “biodiversity scaffolds” (Gallego et al. 2025) owing to the persistence of large-DBH native phorophytes and the predominance of utilitarian over aesthetic pruning criteria. This is consistent with reports from the same municipality showing that peri-urban sites harbor greater bromeliad epiphyte richness than urban areas with lower canopy cover (Aoki-Gonçalves et al. 2023). In peri-urban spaces, native phorophytes predominate and pruning is driven by safety or productive criteria rather than aesthetics, thereby preserving the structural integrity and microhabitat complexity necessary for VE establishment and persistence.

4.3. Ecosystem services and their social valuation: a dissociation along the gradient

The absence of significant differences in biophysical ES provision (carbon storage and sequestration, oxygen production, and stormwater runoff control) across the marginalization gradient challenges the expectation—drawn from other Neotropical cities—that socioeconomic stratification entails proportional stratification of ecosystem service delivery (Escobedo et al. 2015; Dobbs et al. 2019; Meléndez-Ackerman et al. 2022). Even within the municipality of Xalapa itself, differences in ES provision associated with socio-environmental vulnerability have been reported for street tree alignments specifically (García Valencia et al. 2025). The non-significant marginalization effects detected in this study ($p > 0.05$, marginal $R^2 \leq 0.032$) likely reflect the inclusion of diverse land use configurations—including livestock grazing and peri-urban sites—within the 75–100% canopy stratum at the most marginalized site, which maintains functionally equivalent tree biomass despite socioeconomic differences. The high ICC for carbon storage (0.524) confirms that most biophysical variability is attributable to between-site structural differences rather than to the socioeconomic gradient.

Despite this biophysical equivalence, social valuation of urban forest benefits diverged markedly across strata. The progressive displacement of cultural services (aesthetic, recreational, health-related) by provisioning services (fruit, water) and regulation services toward higher marginalization levels is consistent with dynamics documented in the African Global South (Shackleton et al. 2015, 2024) and underscores a fundamental tension: ecologically equivalent forests are valued—and therefore managed—according to socially contingent framings. This dissociation between biophysical provision and perceived value constitutes a critical argument against homogeneous, ornamentally oriented urban reforestation strategies, and in favor of context-specific, socioecologically informed management that aligns forest design with the use and subsistence demands of each community.

4.4. Environmental disservices: perception, misperception, and overlooked chemical risk

Assessment of the urban forest from the perspective of its negative impacts reveals a complex and socioeconomically stratified pattern of both perceived and unperceived disservices. The convergence of elevated “falling risk” concern at the very low and very high marginalization extremes (30.8% and 27.3%, respectively) obscures fundamentally different underlying drivers. In higher-income sectors, this

perception is associated with the protection of valuable urban infrastructure and property; at the most vulnerable site, it is conditioned by housing precariousness and limited institutional response capacity in the event of arboricultural accidents—a pattern consistent with what has been reported in historically marginalized neighborhoods in the United States (Martin et al. 2024). The concentration of infrastructure damage concerns in less marginalized strata (17.3%–18.5%) reflects the greater density of utility networks and residential structures that come into conflict with root systems and expanding canopies in those settings.

In the biological dimension, the prevalence of hemiparasitic plants (11.0%–14.7% of sampled trees) falls within the global range (12.8%) reported for urban parks (Zheng et al. 2025) and is substantially below the mean documented in Brazilian urban plazas (28.38%; de Menezes et al. 2023), suggesting moderate parasitism pressure in the Xalapa urban forest. The misidentification of VEs as parasitic plants by 15.3% of all respondents constitutes a conservation risk of practical significance: erroneous phytosanitary pruning driven by this misperception may eliminate epiphytic microhabitats that sustain a disproportionately high share of urban biodiversity (Shackleton 2016). This finding argues for the inclusion of targeted environmental education components in urban greening programs.

The chemical disservice associated with VOC emissions was universally unrecognized across all strata—a finding that highlights a structural knowledge gap in urban forest management. The apparent paradox whereby the most marginalized site emits the highest per-tree VOC flux (1,229.3 g year⁻¹ tree⁻¹, dominated by *Quercus* spp.) yet represents the lowest chemical risk for tropospheric ozone formation is explained by the interaction with vehicular NO_x: sparse traffic at this peri-urban site (0.129 vehicles household⁻¹) limits the anthropogenic precursor supply necessary for ozone synthesis (Calfapietra et al. 2013; Thambiran et al. 2007). Conversely, the very low marginalization stratum combines moderately high emissions from *Quercus* and *Liquidambar* with dense vehicular loads (0.624 vehicles household⁻¹), constituting an overlooked chemical risk hotspot in one of the city's most resource-rich sectors. These findings underscore the importance of integrating species-specific VOC emission profiles, traffic density, and land use context into urban forest planning decisions, particularly in the selection and placement of species for roadside and high-traffic corridor planting (Dominguez-Taylor et al. 2007; Gebreyesus et al. 2024).

5. Conclusions

Socioeconomic marginalization does not entail a homogeneous loss of ecological function in the Xalapa urban forest. Native species richness is statistically equivalent throughout the gradient, while ecological inequality is expressed through the “garden effect”: the differential capacity of less marginalized strata to introduce exotic species without displacing the native species base. This finding nuances the luxury effect and reveals a marked floristic exclusion of the most vulnerable sectors from access to the utilitarian and subsistence-oriented species they demand ($J = 0.11$).

Host tree diameter (DBH; $IRR = 2.94$) is the primary determinant of vascular epiphyte richness, demonstrating that trees in poorer condition harbor more epiphytes by virtue of being older individuals of greater girth, and not as a direct consequence of bark deterioration. Marginalization does not exert a linear control over VE richness because its apparent effect is an artifact of spatial confounding with land use type; peri-urban livestock grazing environments function as biodiversity scaffolds by preserving large-DBH native phorophytes under utilitarian pruning regimes.

The relationship between biophysical ecosystem service provision and social valuation is differential and non-linear. ES provision (carbon storage and sequestration, oxygen production, and stormwater runoff control) does not decline with increasing marginalization ($p > 0.05$, marginal $R^2 \leq 0.032$). However, the dissociation between what the forest provides and what communities value—shifting from aesthetic-recreational appreciation in lower-marginalization strata to utilitarian and subsistence demand at the most vulnerable site—constitutes a strong argument against homogeneous reforestation approaches and in favor of context-specific, socioecologically grounded forest design.

Perceived and overlooked disservices follow socioeconomic patterns that demand differentiated responses. Concern over falling branch and tree risk reaches similar magnitudes at the extremes of the gradient but is driven by fundamentally different causes (infrastructure protection versus housing precariousness). The misidentification of VEs as parasitic plants by 15.3% of respondents threatens the conservation of tree-associated biodiversity through misguided phytosanitary pruning. Finally, the intersection of high VOC emissions and elevated vehicular density in less marginalized sectors constitutes an overlooked chemical risk hotspot, while the high VOC flux at the most vulnerable site is locally mitigated by sparse traffic.

Urban forest management in Xalapa requires differentiated approaches across the socioeconomic gradient. In high-marginalization zones, conservation of mature native trees with large DBH should be prioritized alongside the incorporation of fruit-bearing, medicinal, and shade-providing species that meet social demand for utilitarian benefits. In lower-marginalization zones, aesthetic pruning practices that eliminate epiphytic microhabitats should be critically reviewed and regulated, and the strategic placement of high-VOC exotic species in high-traffic corridors should be evaluated to prevent adverse contributions to local air quality through tropospheric ozone formation.

The primary limitation of this study is the exploratory nature of the very high marginalization stratum analysis, which is based on a single site (Col. Independencia) that combines peri-urban land use with high VOC-emitting native species. Future research should incorporate multiple high-marginalization sites to confirm the generalizability of the described patterns and to disentangle the effects of socioeconomic vulnerability from land use configuration in structuring both biophysical and social dimensions of the urban forest.

Declarations

Declaration of Competing Interest.

The authors declare no competing financial interests or personal relationships that could have influenced the work reported in this paper.

Supplementary files

Appendix A, B and C

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Author Contribution

All authors contributed to study conception and design. Material preparation, data collection, statistical analysis, funding acquisition, and the first draft were performed by EFV. Methodological supervision, statistical review, and manuscript editing were carried out by JCLA. All authors reviewed and approved the final manuscript.

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Data Availability

All data supporting the findings of this study are available within the paper and its Supplementary Information.

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Figures

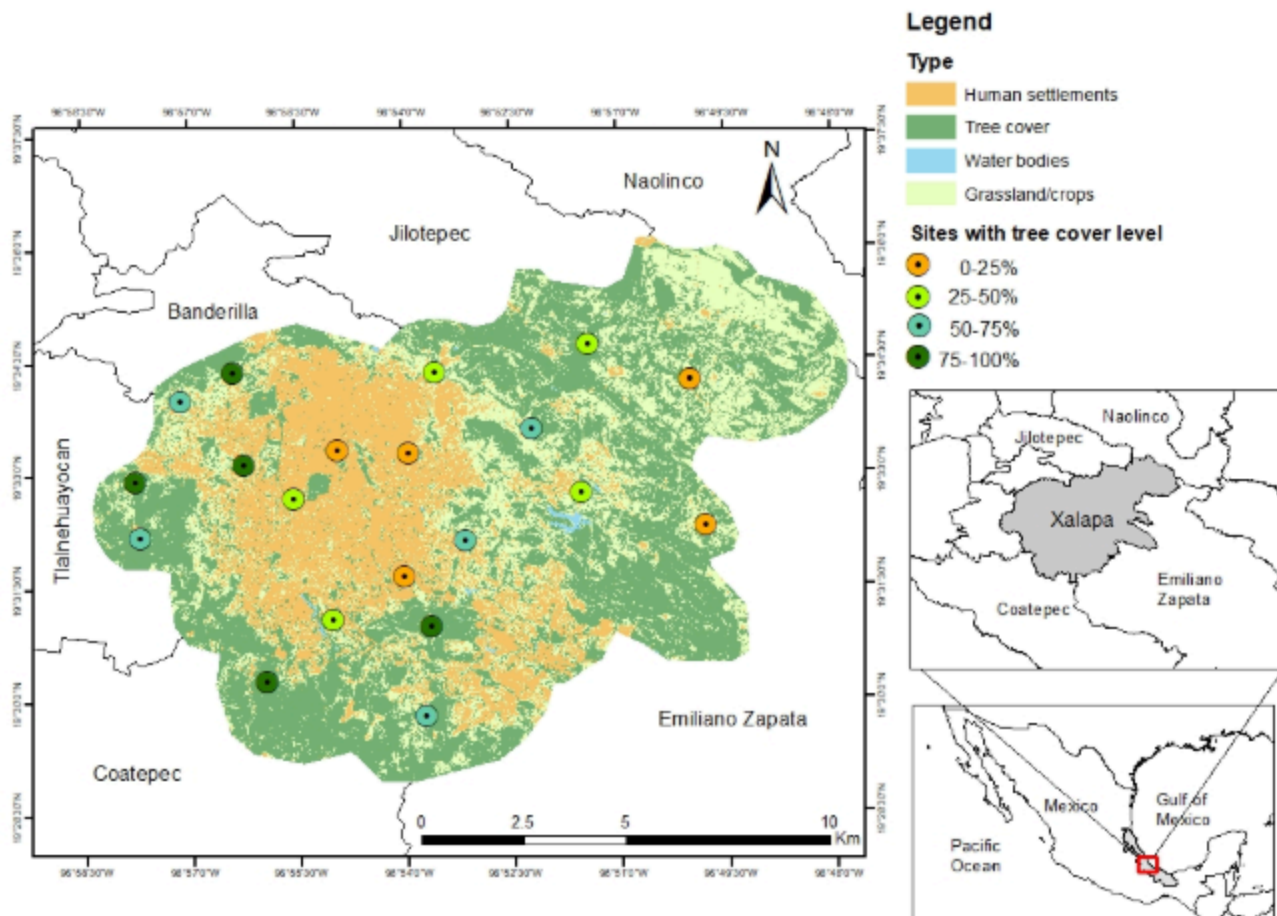


Figure 1

Land use and vegetation map of the municipality of Xalapa and location of the 20 sampled sites by tree canopy cover stratum

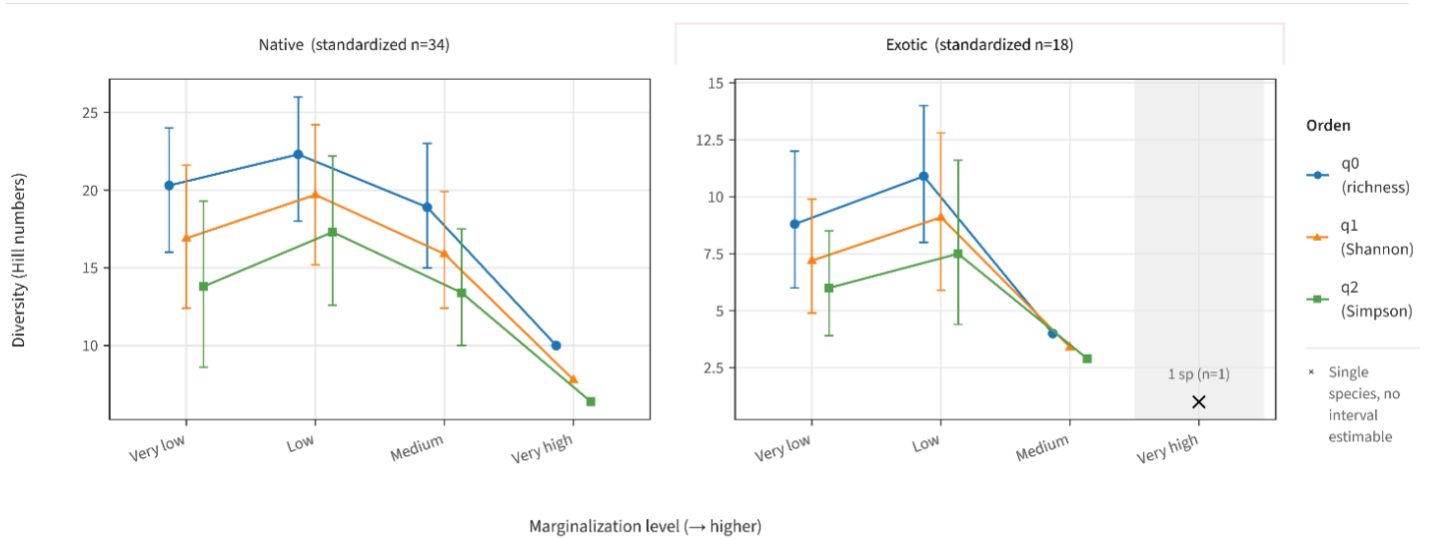


Figure 2

Diversity of native (left panels) and exotic (right panels) tree species along the marginalization gradient, at Hill orders q_0 , q_1 , and q_2 (individual-based rarefaction standardization; vertical bars = 95% bootstrap CI). The very high marginalization stratum (one site) is shown for descriptive purposes only

Supplementary Files

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