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Article

Species Richness, Stem Density, and Canopy in Food Forests: Contributions to Ecosystem Services in an Urban Environment

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Abstract
Food forests expand the traditional concepts of urban forestry and agriculture, providing a broad diversity of tree-related ecosystem services and goods. Even though food forest systems bridge an obvious gap between agriculture and forestry, their potential value in the urban landscape is often undervalued. The inclusion of edible species in urban forest stands can enhance nutrition and well-being in the urban landscape, where food deserts are common. The potential for ecosystem services is especially pronounced in subtropical and tropical regions, where there is a heightened need for shade due to climate change-related heat waves. For this study, we investigated the tree species richness, stem density, and canopy cover provided by food forest gardens in 10 Miami-Dade County, Florida public schools located in the urban landscape. We compared results with neighboring properties around the schools and discovered that the food forest canopy was comparable with neighborhood urban tree cover. Additionally, we established that arborescent species richness (including an increase in edible taxa) and stem density was higher in food forests than in adjacent neighborhood plots. We posit that local food production could be enhanced by planting edible species in small spaces (e.g., empty lots or residential yards), as opposed to focusing on just ornamental taxa or recommended street trees. Our study highlights the importance of using mixed edible tree species plantings (especially with consideration to provisioning, regulating, and supporting services), potentially meeting urban forestry and agricultural goals proposed by urban planners and managers.

Keywords
agroforestry; environmental services; green infrastructure; urban ecology; urban forestry; urban planning; urban sustainability

Issue
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1. Introduction

Urbanization has accelerated over the last few decades, with an estimated 55% of the global population now living in cities (Gao & O'Neill, 2020). Multiple socioeconomic benefits are associated with the urbanization process, yet questions remain about the feasibility of creating sustainable urban spaces where population density is high. As an example, urban forestry has gained considerable traction as an essential component of urban planning in the last decade (Escobedo et al., 2019; Miller et al., 2015). Increased forest canopy (or other types of green infrastructure; see Meléndez-Ackerman et al., 2018) in an otherwise artificial environment ensures the maintenance of several important ecosystem services, including mitigation of urban heat islands (Bowler et al., 2010; Moll, 1989) and carbon storage (Escobedo et al., 2010; Nowak & Crane, 2002). Indeed, in the last decade, there has been an intensified global effort to increase forest cover in the urban landscape, and with good reason. For instance, a recent study looking at 37 metropolitan areas in the US determined that tree canopy coverage in minority neighborhoods averaged only 23%, compared to 43% in predominately US-born white neighborhoods (Locke et al., 2021). Other researchers corroborate this inequity of tree cover distribution across lower income neighborhoods (e.g., Flocks et al., 2011; Landry & Chakraborty, 2009), including the strong correlation between urban biodiversity and neighborhood wealth (the so-called "luxury effect"; see Hope et al., 2003; Leong et al., 2018; Schell et al., 2020). This type of socioecological disparity has led to the development of environmental justice movements (see Campbell, 2014), conceptual models (e.g., Johnson et al., 2020), as well as citizen science tools (e.g., Tree Equity Score; Vibrant Cities Lab: Resources for Urban Forestry, Trees, and Green Infrastructure).

Such discussions have important implications for sustainability and resilience in the urban landscape, not a minor consideration in this era of global climate change (see Ahern, 2013). Community resilience has been defined by other authors as the ability of community members to manage and use communal resources (including food) in order to thrive in an unpredictable and dynamic environment (see Food and Agriculture Organization of the United Nations, 2013; Magis, 2010; Tendall et al., 2015). Resilience is typically viewed as a key factor in determining sustainability (or the ability to meet our needs without compromising the needs of future generations; Berkes et al., 2008; Brundtland & Khalid, 1987; Wu, 2010). Accordingly, one way to reduce a community's ecological and economic vulnerability is to encourage a diversification of natural resources, including economically and culturally important plants (Brown & Jameton, 2000; Buchmann, 2009; Clark & Nicholas, 2013). Other studies have highlighted the important role that locally produced food plays in social networking, health, and community autonomy, particularly during times of economic and environmental strife (e.g., Buchmann, 2009; Meléndez-Ackerman et al., 2018; Shimpo et al., 2019). An increasingly popular trend in urban landscapes includes the cultivation of edible species in multi-storied home gardens, or "food forests" (FFs; Jacke & Toensmeier, 2005). FFs (in the permaculture lexicon, an edible agroforest with an emphasis on perennial plant taxa; see Park & Higgs, 2018; Park et al., 2018) expand the traditional concepts of urban forestry. Despite its ties to the relatively recent permaculture community, these types of multi-storied home gardens are some of the oldest agroforestry systems in existence, particularly in the pantropical regions of the world (Michon et al., 1986; Miller & Nair, 2006; Soemarwoto, 1987). The inclusion of high value edible tree species in gardens has the potential to enhance nutrition and well-being in urban areas, where "food deserts" are common (see Jensen & Orfila, 2021), perhaps explaining their increasing popularity in temperate areas (Lovell et al., 2017).

The ecological design of a FF mimics the structure and biodiversity of a natural forest system (Clark & Nicholas, 2013), including the high species richness, nutrient cycling, and multiple canopy layers typically found under natural conditions (McCoy et al., 2021). Even though FF systems are thought to deliver a broader perspective on the concepts of urban forestry and agriculture (McLain et al., 2012; Park et al., 2018), and contribute to urban food security (Albrecht & Wiek, 2021), their long-term impact in the urban landscape is still uncertain. While the expansive body of scientific studies on tropical rural agroforestry systems dates back several decades, empirical evidence on ecosystem services provided by FFs in the Northern Hemisphere is still in the early stages of development. Ecosystem services are typically defined as falling under distinct categories, including provisioning, regulating, supporting, and cultural services (see Escobedo et al., 2011; Zhang et al., 2018). In the case of FFs, these systems have the potential to provide a broad diversity of tree-related services under all four classifications: provisioning (food security, medicinal resources), regulating (carbon storage, nutrient cycling, shade, erosion mitigation, etc.), supporting (habitat, biodiversity), and cultural (environmental education, sense of place, aesthetic appeal; Eiden, 2021; Thiesen et al., 2022).

Certainly, an added benefit of the FF design (especially in tropical climates) is the patchy shade conditions provided by the multiple canopy layers. The upper and mid-canopy layers, as well as the high stem density, inevitably provide protection for plant species in the lower strata that might be more vulnerable to drought and heat. Additionally, well-designed mixed-species gardens that are considered "closed systems" (i.e., few to no external inputs; see Hart, 1996) are likely to improve soil health, with added benefits to the overall ecological sustainability of home gardening. Nitrogen loss in particular is reduced, due to the enhanced nutrient uptake by tree and crop roots from varying soil depths, a feature much more prominent in mixed-species communities (Nair & Graetz, 2004). The relatively small size of FFs also lends well to encouraging the presence of beneficial insects, including those responsible for pollination and predation services, something that has been noted in diverse, smaller gardens (but not yet studied in FF gardens; see Philpott & Bichier, 2017).

To date, few studies in this region have linked high plant diversity with food security, but it is a logical conclusion that agroforestry systems will augment nutrition levels in a given community, something that has already been documented in rural communities in the tropics (see Jose, 2009; Mburu et al., 2016; Mellisse et al., 2018). Even though edible tree species are often absent from municipal urban tree plans (see Brito & Borelli, 2020), their inclusion in these plans could help offset the low tree diversity often seen in urban areas, where city planners may select the most popular urban forest (UF) tree species, hoping to avoid certain risk factors, such as breakage, maintenance costs, public ire, etc. (Barron et al., 2016; Castro et al., 2018; Kowalski & Conway, 2019; Paquette et al., 2006). Inevitably, this lack of diversity can put UFs at risk and reduces additional ecosystem services and benefits for community members.

We proposed to assess the tree species (or arborescent taxa; e.g., *Carica papaya*) richness, stem density, and canopy coverage of FF and neighboring UF plots in Miami-Dade County. Our aim with the study was to identify the potential contributions of these tree-based systems to provisioning (food production via the inclusion of edible taxa), regulating (canopy coverage), and supporting (species richness) services. Specifically, we ask:

1. Is species richness of arborescent taxa (≥5 cm diameter at breast height [dbh]) greater in the FF plots when compared with the species richness in UF plots, including a greater number of edible taxa?
2. Is stem density of arborescent species (≥5 cm dbh) higher in FF plots when compared with tree density in UF plots?
3. Is the percentage of canopy cover of FF plots comparable with those of UF plots?

2. Methods

2.1. Study Site

The study was conducted in Miami-Dade County, Florida (US), an area of approximately 6,300 km^2 that includes a diverse mix of metropolitan sprawl, natural areas, and agricultural lands (see Figure 1). The region is characterized by wet (May–October) and dry (November–April) seasons, with warm subtropical summers and mild winters, similar to other areas of the Caribbean. Economic and ecological challenges (e.g., rapid development, sea-level rise, vulnerability to hurricanes, and a high diversity of invasive tropical plant and animal species) are prevalent in the region (see Dawson, 2017; Groves et al., 2019; Keenan et al., 2018; Staudhammer et al., 2015). For example, sea-level rise has begun to push wealthy homeowners from locations such as Miami Beach and Fisher Island to the less-affluent neighborhoods sitting on the mainland's oolitic limestone Miami Rock Ridge. Long-term residents in these neighborhoods (many of them immigrants from the Caribbean and Latin America) are then dispersed, often to the outer reaches of the city, where housing prices are more affordable (Keenan et al., 2018).

Flocks et al. (2011) highlight the need for increased tree canopy cover in these disenfranchised neighborhoods, pointing to the higher tree diversity and density in wealthier neighborhoods, such as Coconut Grove and Coral Gables. Currently, the urban center of Miami-Dade County claims an overall canopy coverage of 20% (Hochmair et al., 2020), with recent canopy loss noted in some of the incorporated cities where our study sites are located (e.g., Hialeah). This amount is well below the 40–60% goal previously proposed by urban tree advocates like American Forests (Nowak & Greenfield, 2018), a trend that will be hard-pressed to curb, given the rapid population growth in the Miami-Dade County. According to a recent USDA Forest Service study (Nowak & Greenfield, 2018), Florida claims some of the highest rates of urban growth in the US, much of it centered in the southern portion of the state. Rapid urbanization in this subtropical urban landscape makes the

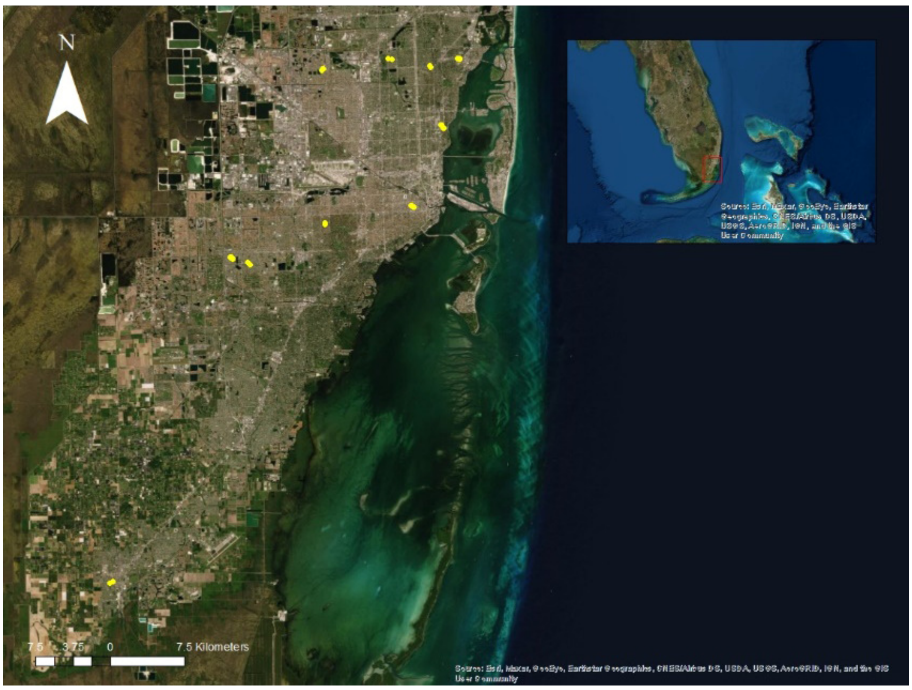

Figure 1. Miami-Dade County, Florida. Yellow dots indicate sites (10) where FF and UF plots were installed.

need to define and implement management plans for resilient UFs and urban growing systems even more critical (Barron et al., 2016; Ferreira et al., 2018; Ordóñez & Duinker, 2014).

2.2. Data Collection

Data were collected from February 2018 to October 2021 in FF gardens located in 10 Miami-Dade County public schools (see Figure 1). The size of the FF gardens in this study ranges from 0.10 to 0.40 ha, while the age of the gardens varies from one to six years. The FF gardens were designed and installed by The Education Fund's "Food Forests for Schools" program (https://www.educationfund.org/what-we-do/programs/food-forests-for-schools/food-forests-for-schools.html). Since 2015, the "Food Forests for Schools" program engages students at 26 Miami-Dade County public schools, elementary and K-8, to plant and maintain FFs on school grounds. The schools use the FFs to promote healthy eating habits and nutritional knowledge, and to create soothing outdoor sanctuaries while growing enough produce for school meals and homebound use. Typically, a rich variety of tropical edible species are cultivated in these perennial gardens (see Figure 2), including Filipino spinach (*Talinum fruticosum*), cranberry hibiscus (*Hibiscus acetosella*), papaya (*Carica papaya*), chaya (*Cnidoscolus aconitifolius*), katuk (*Sauropus androgynus*), sissoo spinach (*Alternanthera sissoo*), yuca (*Manihot esculenta*), bananas (*Musa* spp.), moringa (*Moringa oleifera*), longevity spinach (*Gynura procumbens*), and pigeon pea (*Cajanus cajan*; see McCoy et al., 2021).

Our field team established 10 20 × 20 m FF plots in the FF gardens, in which all arborescent (trees or tree-like) species with a dbh ≥ 5 cm were documented, mapped, and identified. Typically, the individuals in this size category were, on average, at least 4 m tall. Plot locations were selected based on a grid system, in which a plot location was chosen randomly using random number sequences (Laferrière, 1987). A potential plot site was only rejected if it centered on an impervious substrate (i.e., without vegetation). Nonetheless, due to the locations of the gardens within or adjacent to school buildings, some of the 20 × 20 m FF plots included parts of the schools' buildings (see Figure 3), a typical occurrence in garden studies (e.g., Philpott & Bichier, 2017). Neighboring 20 × 20 m UF plots were randomly located at least 100 m away from the FF plots. Similar to the FF plots, potential UF plot locations were rejected if they centered on an impervious substrate (e.g., only street substrate represented the plot). Locations of the plots were required to either have public access or (if on private property) to be of a reasonable distance from the street to ensure confident identification of the species in question. In the UF plots, all arborescent species with a dbh ≥ 5 cm were also documented, mapped, and identified. Species richness was determined to be the total number of taxa with a dbh ≥ 5 cm per plot (e.g., Gotelli & Colwell, 2001). Stem density was calculated using the total number of arborescent stems ≥ 5 cm across the entire 400 m^2 plot and multiplied by the conversion factor (25) to generate stem density ha^{-1}.

Canopy size estimates of the FF and UF plots were determined using the USDA Forest Service's web-based urban tree canopy assessment tool *i-Tree Canopy* V.7 (https://www.itreetools.org). The photo interpretation method of *i-Tree Canopy* uses a random point sampling protocol that interfaces with Google Maps™, enabling the user to estimate the percentage of different land cover types, including tree canopy (Hwang & Wiseman, 2020; Nowak et al., 2018). US Forest Service protocol recommends sample sizes of 500 and 1,000 points, assuming a standard urban municipal area coupled with an average tree canopy cover (US Forest Service, 2011). Boundaries were projected for each 400 m^2 plot area onto a Google Maps™ image of the study area. For our relatively small study areas, we opted to use 30 survey points that were randomly generated for each plot

Figure 2. One of the Miami-Dade County Public Schools' FF gardens surveyed in this study.

Figure 3. 20 × 20 m plots located in FF gardens at two of the participating Miami-Dade County Public Schools.

(Figure 4). Points were categorized as "tree" or "non-tree." For the purpose of this study, tall herbaceous plants (e.g., papaya and banana) were also included under the "tree" category, given their height, which was comparable to neighboring woody stems. Canopy from trees outside of the plots was not included since these stems were excluded from the species richness and stem density estimates.

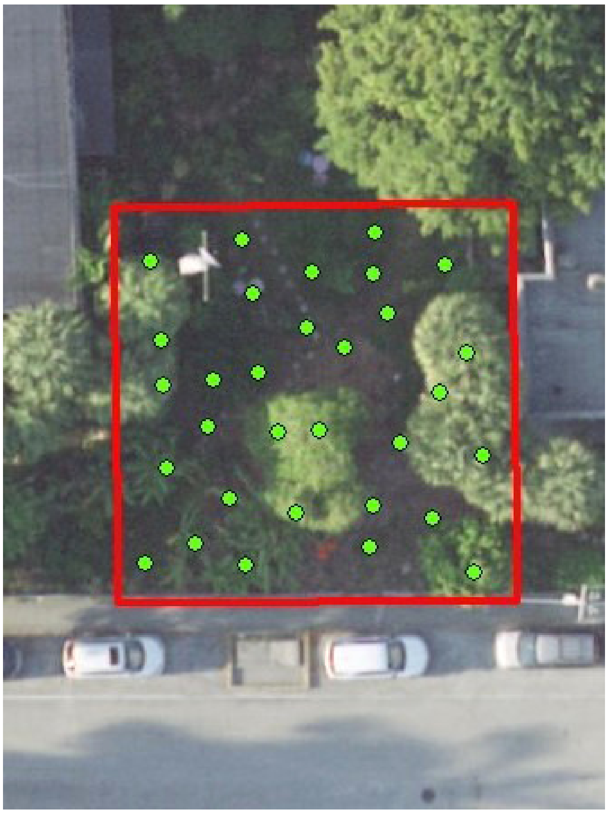

Figure 4. An example of the 30 points that were randomly generated by *i-Tree Canopy* tool in the 20 × 20 m plots.

2.3. Data Analysis

We compared differences in species richness, stem density, and canopy coverage across all 20 FF and UF plots using paired student t-tests in the R 3.4.2 platform (https://www.R-project.org). In addition to verification of plant species using the online New York Botanical Garden *C. V. Starr Virtual Herbarium* (http://sweetgum.nybg.org/science/vh), we also verified the native and invasive status of plant taxa using the Florida Plant Atlas (https://florida.plantatlas.usf.edu), as well as the Florida Invasive Species Council (FISC) website (https://floridainvasivespecies.org). The FISC list characterizes invasive plants as Category I (capable of altering native plant communities) or Category II (increased in abundance but not altering native plant communities).

3. Results

We documented 36 arborescent species across the FF and UF plots (see Table 1), with only 17 species associated with the UF plots, and 28 species in the FF plots. Of those taxa, four FISC Category I species (*Albizia lebbeck*, *Cupaniopsis anacardioides*, *Schefflera actinophylla*, and *Schinus terebinthifolia*) and three Category II species (*Cocos nucifera*, *Koelreuteria elegans*, and *Terminalia catappa*) were identified. While most of the individual invasive stems (n = 11) were located in UF plots, four (*A. lebbeck*, *C. nucifera*, *C. anacardioides*, and *T. catappa*) were found in FF plots. With the exception of the coconut palm, we assume that most of these invasive stems were presumably "volunteers" (or plants that occur naturally due to seed dispersal) that were left to grow and reproduce. Of the 36 species associated with this study, nine were determined to be South Florida natives (*Bursera simaruba*, *Carica papaya*, *Ficus aurea*, *Hamelia patens*, *Lysiloma latisiliquum*, *Pimenta racemosa*, *Quercus virginiana*, *Sabal palmetto*, and *Swietenia mahogani*), of which four were found in UF plots (*B. simaruba*, *F. aurea*, *Q. virginiana*, and *S. mahogani*), and eight were documented in the FF plots (*B. simaruba*, *C. papaya*, *H. patens*, *L. latisiliquum*, *P. racemosa*, *Q, virginiana*, *S. palmetto*, and *S. mahogani*). Only 16 of the 36 taxa recorded in this study are considered "edible," including the invasive *S. terebinthifolia*, which is commonly used as a spice in Caribbean cookery. While this aggressive species is typically present in the urban landscape via the easy dispersal of its seed (often through frugivorous birds), it is actively cultivated in some neighborhoods in Miami-Dade County (Cara A. Rockwell's personal observations). Of the 16 edible taxa, five were found in the UF plots (see Table 1). In only one case did we find an edible species in a UF plot that was absent in the FF plots (*Mangifera indica*). *Musa* spp. was the most abundant edible species (found only in the FF plots), with 51 identified stems (or as in the case of proper botanical terminology, "pseudo-stems"), although it is possible that some of these "individual" banana plants were actually offshoots of the original banana pseudo-stems. Even though cultivated bananas reproduce through "suckers" from the underground rhizome network, we counted these clonal genets as individual stems, rather than as one entire banana plant.

3.1. Species Richness

Average species richness was determined to be significantly higher in the FF plots ($p = 0.02$; see Table 2 and Figure 5), with approximately 5.5 arborescent species in each FF plot, and 2.8 in the UF plots. Given the total number of 28 species in the FF plots, this relatively low number of species per plot suggests that species composition varies significantly across the 10 FF sites, at least per unit area. Indeed, in some cases, the 400 m^2 surveyed represents a small fraction of the total area (e.g., the largest FF garden surveyed in this study is close to 4,000 m^2), so presumably, our sampling likely missed other arborescent species present in the gardens. Even though the more popular cultivated species (e.g., *C. papaya*, *Musa* spp.) are generally represented by multiple stems across the FF gardens, we did encounter clustering of certain species, potentially leading to underestimation (or overestimation in some cases) of some taxa within the 20 × 20 m plots.

Table 1. Identified species, South Florida native status, FISC category, and number of stems in 10 FF and 10 UF plots in Miami-Dade County, Florida.

Species	Common Name	Family	Edible	South Florida Native	FISC Category	FF	UF	Total
Adonidia merrillii	Christmas palm	Arecaceae				7	2	9
Albizia lebbeck	Golden silk tree	Fabaceae			I	1	1	2
Averrhoa carambola	Starfruit	Oxalidaceae	✓			1	1	2
Bursera simaruba	Gumbo limbo	Burseraceae		✓		5	1	6
Carica papaya	Papaya	Caricaceae	✓	✓		19		19
Chrysophyllum cainito	Caimito	Sapotaceae	✓			1		1
Citrus hystrix	Kaffir lime	Rutaceae	✓			1		1
Cnidoscolus aconitifolius	Chaya, Mayan spinach	Euphorbiaceae	✓			2		2
Cocos nucifera	Coconut palm	Arecaceae	✓		II	1		1
Cupaniopsis anacardioides	Carrotwood	Sapindaceae			I	1		1
Diospyros digyna	Black sapote	Ebenaceae	✓			1		1
Eriobotrya japonica	Loquat, Japanese plum	Rosaceae	✓			1	1	2
Ficus aurea	Florida strangler fig	Moraceae		✓			1	1
Ficus religiosa	Sacred fig	Moraceae				1		1
Hamelia patens	Firebush	Rubiaceae		✓		4		4
Handroanthus sp.	Trumpet tree/ipê	Bignoniaceae				1		1
Koelreuteria elegans	Flamegold rain tree	Sapindaceae			II		2	2
Ligustrum sp. *	Privet	Oleaceae					1	1
Lonchocarpus sp.	Lancepod	Fabaceae					1	1
Lysiloma latisiliquum	False tamarind	Fabaceae		✓		1		1
Mangifera indica	Mango	Anacardiaceae	✓				2	2
Moringa oleifera	Moringa	Moringaceae	✓			9	1	10
Morus nigra	Black mulberry	Moraceae	✓			3		3
Muntingia calabura	Jamaican cherry, strawberry tree	Muntingiaceae	✓			2		2
Musa spp.	Banana	Musaceae	✓			51		51
Peltophorum pterocarpum	Yellow poinciana	Fabaceae					1	1
Pimenta racemosa	Bay rum	Myrtaceae	✓	✓		1		1
Quercus virginiana	Live oak	Fagaceae		✓		2	12	14
Sabal palmetto	Sabal palmetto	Arecaceae		✓		12		12
Schefflera actinophylla	Queensland umbrella tree	Araliaceae			I		1	1
Schinus terebinthifolia	Brazilian pepper	Anacardiaceae	✓		I		3	3
Sesbania grandiflora	Hummingbird tree	Fabaceae	✓			2		2
Swietenia mahagoni	West Indian mahogany	Meliaceae		✓		1	3	4
Terminalia buceras	Black olive	Combretaceae				2	6	8
Terminalia catappa	Tropical almond	Combretaceae			II	1		1
Veitchia arecina	Montgomery palm	Arecaceae				6		6
Total						140	40	180

Notes: Category I—capable of altering native plant communities; Category II—increased in abundance but not altering native plant communities. * There are two FISC-listed Category I invasive *Ligustrum* species in Florida (*L. lucidum* and *L. sinense*), but we have refrained from listing this individual as an invasive, given that we were unable to identify it to species without flowers.

Table 2. Student t-test results for comparison of 10 FF and 10 UF plots in Miami-Dade County, Florida.

	Sample Mean		Student t-Test Results
	FF (pre-hurricane)	UF (post-hurricane)	
Species richness	5.5	2.8	df = 9; t = 2.8; p = 0.02
Stem density ha^{-1}	350	100	df = 9; t = 4.5; $p \leq 0.01$
Canopy (%)	51.3	46.7	df = 9; t = 0.6; p = 0.57

3.2. Stem Density

Stem density between the FF and UF plots differed significantly ($p \leq 0.01$; see Table 2 and Figure 5). The total number of stems across the ten FF plots was calculated to be 140 (dbh ≥ 5 cm), and the total number of stems across the 10 UF plots was 40. However, it must be noted that the average girth of the UF trees tended to be larger than the FF plants (the most common UF tree was the large canopy species, *Q. virginiana*), thus allowing for fewer trees within the 400 m^2 area, given above- and belowground competition limitations. The mean number of stems in the FF plots was found to be 14 (350 stems ha^{-1}), and four (100 ha^{-1}) in the UF plots.

3.3. Canopy Coverage

Canopy coverage did not differ between FF ($\overline{x}$ = 51.3%) and UF ($\overline{x}$ = 46.7%) plots (p = 0.57; see Table 2 and Figure 5), despite the higher number of stems and species richness in the FF plots.

4. Discussion and Conclusions

Our research represents an important case study about urban FF systems and their importance in urban landscapes. While we did not specifically measure long-term food security in these neighborhoods as a function of high species diversity, we did confirm that our FF plots had a high number of edible arborescent species (14 of the 28 FF species, or 50%), as well as a significant number of edible taxa stems (95 of the total 140 stems found in the FF plots, or 68%). One could therefore make a strong case that the inclusion of edible species in a front yard or an urban park (as opposed to a UF with none) could benefit food security (and potentially nutrition). In the UF plots, we documented several edible taxa (*Averrhoa carambola*, *Eriobotrya japonica*, *M. indica*, *Moringa oleifera*, and *S. terebinthifolia*), but four of these were found only in the front yard of one private residence. This lack of edible species in the UF plots (particularly in the case of plots that were located in the public right-of-way) suggests that there may be some reticence on the part of local governments to plant edible tree species (see Hajzeri & Kwadwo, 2019; Kowalski & Conway, 2019; Ortez, 2021). Certainly, data collection from Florida International University's Grove ReLeaf UF project (https://pg-cloud.com/ictb) confirms that few edible trees exist in the public right of way in Coconut Grove, a prominent neighborhood in the center of the city of Miami. According to their unpublished data set, only 45 of the total 319 arborescent species (which includes the herbaceous taxa *C. papaya* and *Musa* spp., as well as multiple palm species) are considered edible taxa. Of these taxa, *C. nucifera* (or coconut palm) is the most common edible species (326 occurrences in the database), although planting of *C. nucifera* is now prohibited by the City of Miami, due to the hazard it poses from falling fruits. Additionally, it has been identified by FISC as a Category II invasive plant. As another local example, the Miami-Dade County Street Tree Master Plan lists 63 recommended street trees, but only six taxa (*Celtis laevigata*, *Coccoloba diversifolia*, *Coccoloba uvifera*, *Noronhia emarginata*, *Pimenta dioica*, *Podocarpus* sp.) have edible or medicinal properties (Miami-Dade County, 2007).

Given the potential contributions of FF gardens to food security, lack of emphasis on edible species cultivation may be missing an important opportunity to address local food production, especially given the increased levels of food insecurity due to Covid-19 (see Gundersen et al., 2021; Niles et al., 2020). Indeed, edible tree species are often overlooked for urban canopy enhancement recommendations by municipal governments, despite the inclusion of FFs in the Food and Agriculture Organization's Guidelines on Urban and Peri-urban Forestry, which highlights their role in addressing hunger (Salbitano et al., 2016). In the case of the FFs in this study, certain species are known for high levels of production, depending on local site conditions, weather, management prescription (e.g., fertilization), and variety of the species in question. As an example, researchers from University of Florida's Institute of Food and Agricultural Sciences have determined that a mature grafted mango tree is capable of producing up to 100–150 kg/year (Crane et al., 2020), and the herbaceous papaya plant of producing 27–36 kg/year (Crane, 2018). Despite these obvious benefits, some of the hesitancy in planting edible species may have to do with concerns of maintenance, as well as urban pests, such as rats. A recent study from Brazil points to the low number of municipalities that encourage edible species, despite the increasing levels of food insecurity in Brazilian cities. Only five of the 49 municipalities surveyed considered the positive aspects of planting edible species in the UF; the rest of the UF management plans actively prohibited the planting of edible taxa (see Brito & Borelli, 2020).

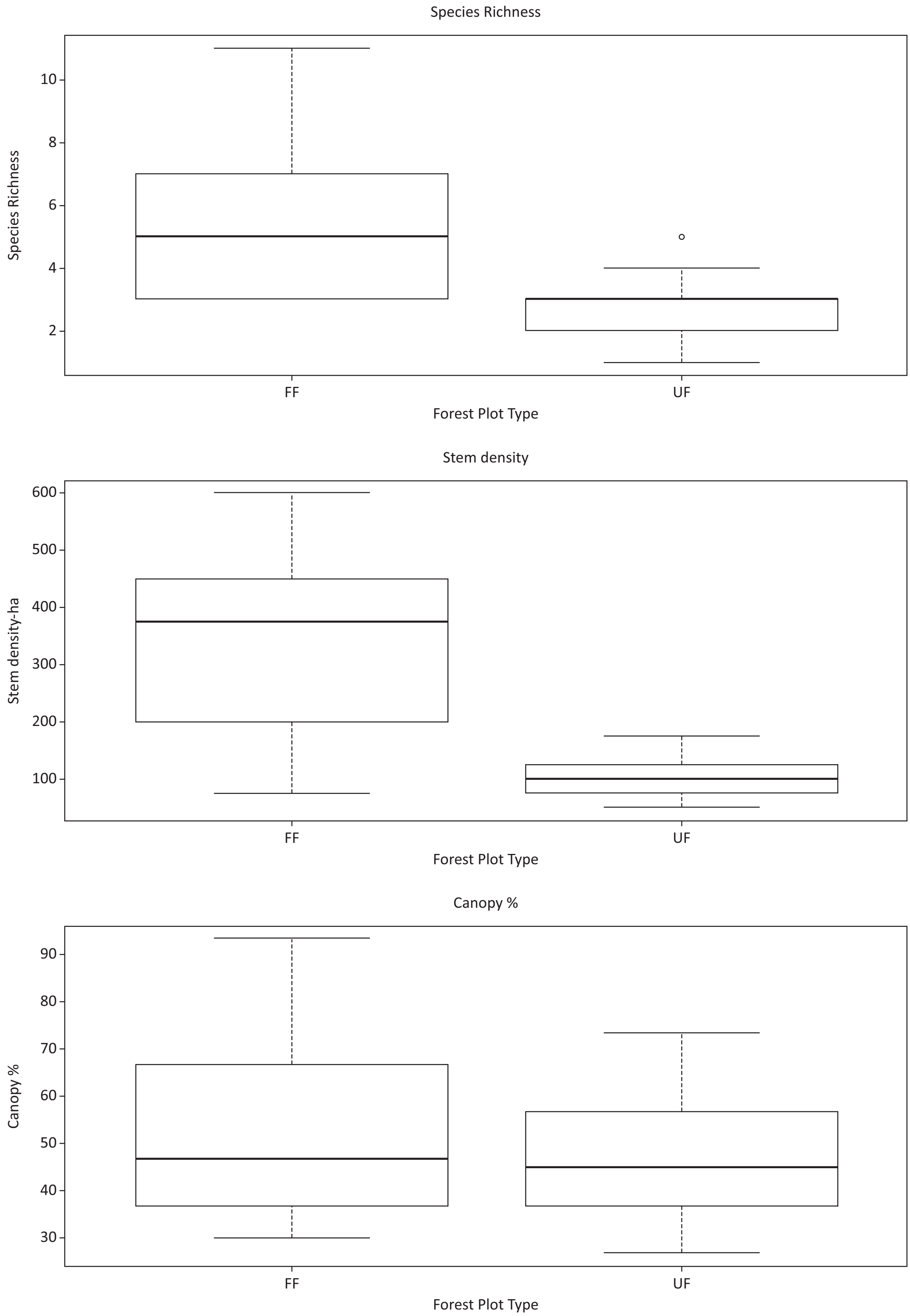

Figure 5. Paired student t-test results for species richness (number of species per hectare; $p = 0.02$), stem density (number of stems dbh ≥ 5cm per hectare; $p \leq 0.01$), and canopy coverage (in %; $p = 0.57$) for 10 FF plots and 10 UF plots in Miami-Dade County, Florida.

The potential contributions of FF-based ecosystem services to food security are merely one aspect of this urban agroforestry system. The mix of edible and native species that we documented in the FF gardens compels us to discuss the integration of the native landscape concept with an edible garden focus. Indeed, there very well may be a benefit for FF gardens to include native taxa. For example, the Florida native firebush (*Hamelia patens*, of which we found four examples in one of the FF sites) is known to attract a rich diversity of pollinators, including hummingbirds, butterflies, and bees. Most of our sites are located in former pine rockland, an endangered habitat that now only exists in small patches (outside of Long Pine Key in Everglades National Park; see Possley et al., 2014). Presumably, planting native taxa in an edible garden, or even encouraging some of the native weeds, such as Spanish needles (*Bidens alba*; see Kleiman et al., 2021), could provide other ecosystem services, such as sources of food for native pollinators. This aspect of food forestry has not been fully explored, although it has been noted in more recent articles on the subject. Park et al. (2018) expressed the importance of using FFs to enhance native habitat restoration, even though FFs have not historically relied on native plant taxa. At the same time, this inclusion of native species has the potential to integrate FFs into a sustainable urban green infrastructure framework that reaches beyond the food security benefits. In a sense, these relatively small spaces could be viewed as "ecological stepping stones" that provide a buffer for native habitat patches in the relatively artificial urban environment, as long as the cultivation of potentially invasive species is avoided.

Nevertheless, the presence of FISC-listed invasive plant taxa in both the FF and UF plots was notable (seven out of a total of 36 species identified in this study, or 19%). The subtropical climate and high levels of urbanization in South Florida lend well to the establishment and persistence of aggressive exotic taxa (Staudhammer et al., 2015). In most cases, the individual invasive plants documented in our study are likely volunteers that were not removed before they became reproductive. Some of this reticence to cut down invasive tree species could be due to lack of knowledge. Alternatively, the failure to act could be stemming from an actual appreciation of certain characteristics of the tree that led it to be introduced to the region in the first place. For example, *S. terebinthifolia* has long been favored by South American and Caribbean cultures for its medicinal properties (Dvorkin-Camiel & Whelan, 2008; Muhs et al., 2017) and for its spicy fruits, which can add a peppery flavor to traditional dishes (Jones, 1997). In at least one UF plot (an empty lot), the presence of *Schinus* is likely due to bird-related dispersal. In the other case (a middle school parking lot), it appeared as if the shrubs were planted as a hedge (see Figure 6). The school is located in a neighborhood known for its Haitian population, members of the community that would likely recognize the edible and medicinal properties of the species. Regardless, the importance of reducing the number of invasive taxa in urban areas cannot be overstated,

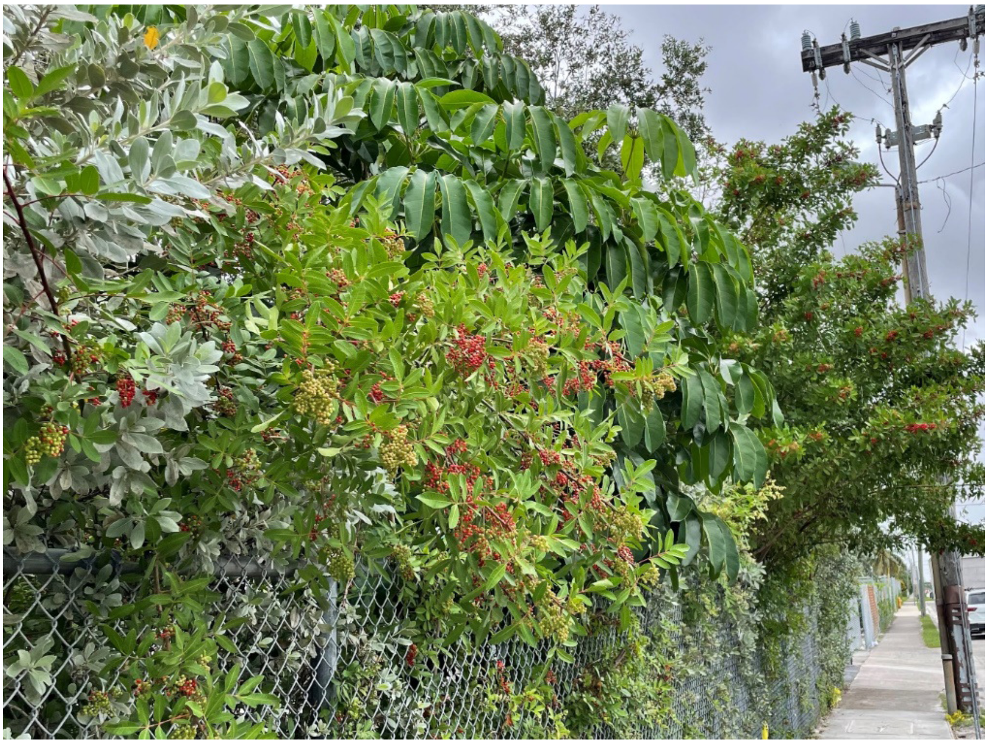

Figure 6. Planted hedge in one of the UF plots. Note the presence of the Brazilian pepper (red fruits, *Schinus terebinthifolia*), planted next to the Florida native, buttonwood (far left, *Cornocarpus erectus*), and the invasive Queensland umbrella tree (in between two stems of *S. terebinthifolia*; *Schefflera actinophylla*).

especially in the case of Category I species, which have the potential to outcompete (and displace) native plants and impact ecosystem services (Escobedo et al., 2010).

Additionally, we determined that canopy coverage in the FF plots was comparable to that of neighboring urban plots. While the FF canopy coverage did not surpass that of the neighboring UF plots, at the very least, our results suggest that FFs can potentially contribute towards the much-needed canopy cover in urban landscapes. Increased canopy cover in metropolitan areas has been demonstrated to reduce the urban heat island effect (Loughner et al., 2012; Ziter et al., 2019). As well, agroforestry studies in the pantropical regions of the world have long highlighted shade benefits of diverse edible tree-based systems, including links to sub-canopy plant health, water loss, and dietary diversity (e.g., Baudron et al., 2019; Tscharntke et al., 2011). Understandably, large crown woody species are typically favored for urban canopies, but for those residents seeking to gain both shade and food benefits around their house, other species could be considered. Additionally, many temperate and tropical sub-canopy species require partial shade conditions. For example, banana and papaya were common taxa in this study. While they are herbaceous plants, their large size allows them to be considered "canopy" species, at least in the FF system. While it is doubtful that as individual plants they could provide the same amount of canopy as a large, long-lived live oak tree, they do provide a certain amount of shade. Few studies have looked at the benefits of urban cooling as it relates to the height of the trees, but at least one recent tropical study (Blaser-Hart et al., 2021) determined that low and elevated-canopy trees in cacao agroforestry systems were equally effective at mitigating climate extremes. Certainly, we may look to more research examples in tropical agroforestry systems for insights into the benefits of tree-based systems that are characterized by variable canopy heights.

Along those lines, one of the major critiques of the photographic interpretation method utilized by *i-Tree Canopy* is the reliance on visual assessment of the image by the user. Admittedly, visual interpretation is prone to error, primarily due to the variable quality of the Google Maps™ image (especially when focusing on smaller subsets of the landscape), which can lead to misinterpretation (Hwang & Wiseman, 2020). In our case, given our familiarity with the ground data (e.g., number of stems, locations of impermeable surfaces), we believe that we mitigated this risk of misinterpretation. Nevertheless, canopy coverage in a FF garden is admittedly variable when compared with the crown cover of a more typical UF. While the specific traits of canopy coverage (e.g., height, continuity, age) were not a focus in this study, we did observe patchy shade conditions in the FF gardens. We believe that the non-contiguous FF shade in our plots is primarily due to a combination of factors: (a) the variable height of upper and mid-canopy species, (b) the diverse leaf traits of certain FF species (e.g., the small leaflets of the bipinnate or tripinnate leaves of *M. oleifera*), and (c) the design of the FF itself (i.e., heterogeneous distribution of multiple canopy layers; see Jacke & Toensmeier, 2005).

Few studies on urban FFs have explored the biological components (e.g., biodiversity, nutrient cycling, predator services, etc.) of these systems (but see Björklund et al., 2019; Park et al., 2018; Russo et al., 2017), even though their popularity is growing at a very rapid rate across global metropolitan areas. Indeed, several studies have highlighted the contributions of urban gardens and FFs to social resilience (see Chan et al., 2015; Shimpo et al., 2019), but the ability of urban agroforests to enhance ecological resilience and maintain ecosystem services in the urban landscape (especially in the face of climate change) is less certain. Recent studies have pointed to the importance of the FF design, which incorporates three-dimensional vegetation layers into the garden layout, facilitating the availability of multiple niches for both plants and associated organisms (Björklund et al., 2019; Cannell et al., 1996; Park et al., 2018). Additionally, there is a growing need to adapt agroforestry systems to extreme climate events (presumably already a significant factor in warmer climates; see Barona et al., 2020; Dawson et al., 2010; Luedeling et al., 2014). Providing alternate forms of small-scale food production under canopy cover will have extensive applicability to other grassroots efforts across the nation, informing policymakers, practitioners, and urban community members about the efficacy of urban food generating efforts. We know that annual gardens can mitigate urban heat islands and benefit food security (see Andersson et al., 2019), but these ecosystem services are likely to be magnified in a perennial system that incorporates trees.

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Conflict of Interests

The authors declare no conflict of interests.

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About the Authors

Cara A. Rockwell holds a PhD in tropical forest ecology from University of Florida. She is currently a research faculty member at Florida International University's Institute of Environment. Cara worked in South America for more than 15 years, investigating the impacts of timber harvesting on bamboo-dominated forest stands and Brazil nut-rich forests, as well as forest restoration through post-harvest silvicultural techniques. In addition to her ongoing projects in the Southwestern Amazon region, she has begun developing projects in South Florida, where she is focusing on ecosystem services in urban agroforestry systems, as well as recovery of native ecosystems post-disturbance.

Alex Crow graduated with a B.S. in environmental studies in 2019 from Florida International University's Department of Earth and Environment. Born and raised in Miami, he loves the unique blend of culture, people, and food that his city has to offer. He has a passion for the Everglades, and his research interests include native insect ecology, tropical and subtropical forestry, and apiculture. Alex worked in the Miami-Dade County Public School food forests from 2018–2021. He is now a restoration biologist with the Institute for Regional Conservation in Delray Beach, Florida.

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Eduardo Recinos, senior program manager for The Education Fund's Food Forests for Schools is an artist and teacher of 10+ years who earned a BFA and Teaching Certificate from Florida International University. He is an expert school gardener, having previously served as the lead garden teacher at Twin Lakes Elementary in Hialeah, Florida where he created a food forest that became integrated into all classroom lessons and where he led the innovative school-wide use of garden produce in the cafeteria. Eddie continues to apply his knowledge of curriculum, forest gardening, and nutrition to build the program, including the design of each Food Forest, integration of Food Forests in daily lessons across all subjects, and oversight of a dozen expert gardeners who visit schools to maintain and team-teach.

Deborah La Belle, program manager for The Education Fund's Food Forests for Schools, is a classically trained chef who worked with renowned James Beard awardee Michele Bernstein. A graduate of Johnson & Wales College of Culinary Arts, Deborah's knowledge of nutrition and our unique crops has propelled our program forward. Prior to joining The Education Fund, she taught cooking classes to Miami-Dade County Public Schools students, and ran her own home farm from which she created products for sale. Deborah's knowledge and skills allow her to work effectively with teachers, students, parents, as well as cafeteria personnel applying her knowledge of cooking, forest gardening, and nutrition to build the program.