SOCIAL-ECOLOGICAL FACTORS INFLUENCING URBAN YARD VEGETATION AND THE PROVISION OF SUSTAINABLE ECOSYSTEM SERVICES

by

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Abstract

Residential green spaces are increasingly gaining attention for their potential to contribute to ecosystem services of social and ecological value for cities. This research evaluated the potential of residential yards of San Juan, Puerto Rico, to contribute to urban sustainability through the provision of ecosystem services using a social-ecological approach. The study builds upon prior work at this site led by the San Juan Urban Long-Term Research Area (ULTRA) Collaborative Network and addressed the following overarching question: Which social-ecological factors could be influencing the vegetation structure and composition of the Río Piedras Watershed residential yards and their associated ecosystem services and disservices across the watershed? The work combines social and ecological data collected from household and yard surveys following ULTRA's long-term stratified sampling scheme of households via a convenient-based recruitment. Household surveys used semi-structured questionnaires implemented in 2011 and 2014 evaluated resident values and attitudes towards residential vegetation and their associated ecosystem services and how these may influence the structure and composition of yard vegetation across the watershed. This study took advantage of vegetation surveys implemented before and after the 2017 hurricane season to evaluate the influence of hurricane disturbances on yard vegetation. Main findings highlight that self-reporting of resident attitudes toward yard trees are generally positive with residents emphasizing ecosystem services over disservices, and varied according to differences in the spatial context of trees and residents. Models show that positive attitudes at the household scale may explain some of the variation in the number of yard trees. Residents also self-reported positive attitudes towards native plants mainly driven by sense of place,

and expressed preference towards certain plant traits (i.e., habit, size) and ecosystem services. Findings also show that large-scale hurricane disturbances can have immediate effects on yard vegetation structure and composition and be an important driver of the provision of ecosystem services in addition to the stated social factors. In this work it is argued that understanding how social and ecological factors interact locally to influence yard vegetation provides a better idea of what elements of the vegetation may provide functions of local value and promote sustainability.

Biography

Sofia Olivero Lora was born in San Juan, Puerto Rico on June 1, 1982. She started her studies in the University of Puerto Rico (UPR) at Humacao Campus in Wildlife Management Biology program and conducted research through the Mona Island Project at the CREST-CATEC (Center for Applied Tropical Ecology and Conservation) of the UPR Río Piedras Campus. After participating in an Ethnobiology Training Course from the Organization for Tropical Studies, she transferred and moved to Costa Rica to start a more interdisciplinary approach to her professional development in Biology at the Universidad Latina in San José, were she obtained a bachelor's degree in Biological Sciences with emphasis in Ecology and Sustainable Development. In 2009 she began the master's program in Forest Management and Biodiversity Conservation at the Tropical Agricultural Research and Higher Education Center (known as CATIE) in Turrialba, Costa Rica. She conducted her thesis studies with the Livestock and Environmental Management Group (GAMMA) under the project "FUNCiTree" using functional ecology approaches to evaluate isolated tree functions in silvopastoral systems located in the season dry forests of Rivas, Nicaragua. During her doctoral studies in Puerto Rico, she has received two National Science Foundation (NSF) doctoral fellowships, collaborated with the USDA Forest Service's International Institute of Tropical Forestry in research projects in the Caribbean region, communicated findings in a variety of conferences in Puerto Rico and internationally, and participated in multiple interdisciplinary collaborations. She has worked as an 'environmental protection specialist' at the Federal Emergency Management Agency in Puerto Rico after the devastating 2017 hurricane season.

Dedication

To Elsa and Fernando.

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Chapter 1. Introduction

1. Background

1.1 Urban green infrastructure and the mitigation of urbanization effects.

With more than half of the human population now living in urban areas, processes related to urbanization create big social and ecological challenges (Grimm et al., 2008; United Nations, 2018). The expansion and modification of urban areas, continues to alter the function of our earth systems at different spatial and temporal scales, increasing the vulnerability of urban social-ecological systems worldwide and calling for efficient coping mechanisms (Wu, 2014; United Nations, 2018). At the core of these functional differences is the loss of green cover in urban areas, which leads to reduced shading, evaporating cooling, rainwater interception and infiltration (among other effects), which exacerbate the effects of climate change on ecosystems, human health and wellbeing (Gill et al., 2007; Tzoulas et al., 2007). Cities are also more vulnerable and experience more losses to natural disasters compared to rural areas because of their high concentration of people, infrastructure and services (Dickson et al., 2012; McPhillips et al., 2018; Elmqvist et al., 2019). The improvement of green infrastructure, the distributed networks of green areas on cities, is increasingly being proposed as a realistic solution to mitigate the negative effects of environmental changes brought about by urbanization and to direct cities towards sustainable pathways (Benedict & McMahon, 2006; Gill et al., 2007; Tzoulas et al., 2007; Ahern, 2011; Lovell & Taylor, 2013; Gómez-Baggethun & Barton, 2013; Wu, 2014).

Indeed, the use of the green infrastructure framework has been viewed as an opportunity to operationalize urban sustainability given the social, economic and

ecological benefits that green spaces provide to cities (i.e., ecosystem services approach, (Benedict & McMahon, 2006). On the other hand, recent work also emphasizes that the ecosystem services framework also needs to consider the potential for vegetation disservices in green infrastructure planning as a way to identify social barriers to green infrastructure and consider them accordingly in the development and implementation of green infrastructure policies and plans (Lyytimäki et al., 2008; von Döhren & Haase, 2015; Lyytimäki, 2018). Tree planting is frequently included in green infrastructure strategical planning as a way to maximize the provision of ecosystem services by green spaces (Dobbs et al., 2017). Urban forests (trees and associated vegetation) can help mitigate and adapt to extreme weather events (e.g., climate change, increasing temperatures, extreme flooding) (Gill et al., 2007; Lafortezza et al., 2017; Orlandini et al., 2017; Sjöman et al., 2018), and can contribute to biodiversity conservation (Alvey, 2006; Roy et al., 2012; Liveseley et al., 2016). It is agreed that for these to be successful, urban forestry strategies at local and regional scales, need to be aligned with green infrastructure planning at the landscape scale, where goals are defined by spatial patterns and the connectivity of urban green spaces that provide multiple benefits (Escobedo et al., 2019) while minimizing potential disservices.

Urban green spaces in the form of residential yards can provide a significant amount of green infrastructure in cities and attention has been placed on their role for ecosystem services production. Residential green areas comprise a significant portion of green infrastructure within cities (Gómez Sal et al., 2006; Loram et al., 2008; Muñoz-Erickson et al., 2014; Ramos-González, 2014; Haase et al., 2019) and private residential yards often contain the majority of trees of urban forests, consequently, residents are key

in decision-making and management (Konijnendijk van den Bosch et al., 2006; Nowak & Greenfield, 2012; Almas & Conway, 2018). Residential yards are privately managed spaces, best understood by using an integrated multi-scalar, social-ecological approach to develop understanding of these systems (Pickett et al., 2001; Goddard et al., 2010; Cook et al., 2012; Muñoz-Erickson et al., 2014; Chowdhury et al., 2016; Meléndez-Ackerman, Nytch, et al., 2016) and their potential to deliver sustainable ecosystem services.

1.2. General overview of theoretical frameworks

Social-ecological and environmental psychology theories are helpful approaches to understand the factors that will influence the diversity, abundance, structure and composition of residential yards, here viewed as a social-ecological system where social (e.g., values, attitudes toward vegetation, household socio-economic factors) and ecological factors (biotic and abiotic environmental elements) and their interactions and feedbacks will influence yard vegetation in space and time (Cook et al., 2012; Freeman et al., 2012; Casado-Arzuaga et al., 2013; Urgenson et al., 2013; Van Heezik et al., 2014). Residential yard spaces are privately own, and their management is influenced through different motivations and drivers. Exploring the drivers of these systems can help define management goals and tailor strategies that tend to local needs more effectively.

There is consensus that urban trees produce socio-cultural and ecological functions that result in important ecosystem services for human well-being and promote resilience (Roy et al., 2012; Livesley et al., 2016; Ordóñez-Barona et al., 2017; Nowak & Greenfield, 2018; Steenberg, Duinker, et al., 2019). However, trees can also cause perceived or realized disservices (e.g., structural damage to private or public

infrastructure, sources of pollen allergens) if their placement is not planned adequately (Gómez-Baggethun et al., 2013). In fact, recent reviews point to the need of considering disservices as potentially limiting driver of urban vegetation (Lyytimäki & Sipilä, 2009; Escobedo et al., 2011; Dobbs et al., 2014; Lyytimäki, 2014; Lyytimäki, 2014; von Döhren & Haase, 2015; Wang et al., 2015; Shackleton et al., 2016). The interplay between the perceived services and disservices could also represent a barrier for the acceptance of green infrastructure strategies and could also potentially influence the management of green spaces (Lyytimäki et al., 2008; Escobedo et al., 2011; Shackleton et al., 2016). Thus, urban planning for sustainable outcomes, needs to address human interactions with green spaces and how they influence their planning, design and management (Cook et al., 2012; Freeman et al., 2012; Casado-Arzuaga et al., 2013; Urgenson et al., 2013).

Increasing our comprehension of the feedbacks between human activities and the state of urban green areas can be used to improve the provision of multiple ecosystem services and promote urban resilience as a result (Folke, 2006; Mchale et al., 2015; McPhearson et al., 2016; Pickett et al., 2011). From a social perspective, environmental psychology provides theories that can help us understand what is behind people's motivations and potential behaviors. For example, the Cognitive Hierarchy Theory helps us contextualize human-environment relationships by emphasizing on causal nature between individual values, collective values, attitudes (negative or positive evaluations of specific objects or issues), behavioral intentions and human behavior (Rokeach, 1973; Baur et al., 2016). These cognitive factors build on one another going from more abstract and strongly held values to more specific but variable behaviors (Fulton et al., 1996;

Vaske & Donnelly, 1999; Manfredo & Dayer, 2004; Whittaker et al., 2006). Building on this, the value-attitude-behavior hierarchical model provides a framework to study attitudes as intermediary indicators between values and behavior towards objects (e.g., tree) which is useful in understanding yard management practices (Homer & Kahle, 1988; Milfont et al., 2010). This dissertation builds on these theories to characterize residents' expressed values and attitudes toward residential vegetation components, their associated services and disservices, and how these relate to potential yard management behavioral outcomes.

1.3. The research

This research evaluated residential households of San Juan, Puerto Rico, as social-ecological systems and their potential to contribute to urban sustainability through the provision of ecosystem services. This dissertation addressed knowledge gaps in how social and ecological factors may influence vegetation structure and the provision of ecosystem services in residential yards of the Río Piedras Watershed (RPWS) in the Metropolitan Area of San Juan. This is a tropical urban watershed (the most urbanized in the island of Puerto Rico) where the green infrastructure of residential yards has been continuously studied using social-ecological approaches (Garcia-Montiel et al., 2014; Meléndez-Ackerman et al., 2014; Meléndez-Ackerman et al., 2016; Muñoz-Erickson et al., 2014; Vila-Ruiz et al., 2014). This study built upon prior work which shows how multi-scalar social, ecological and physical factors may influence yard vegetation (Muñoz-Erickson et al., 2014; Meléndez-Ackerman, Nytch, et al., 2016) by addressing the following overarching question: Which social-ecological factors could be influencing

the vegetation structure and composition of RPWS residential yards as well as their associated of ecosystem services and disservices across the Río Piedras Watershed? To address this question the study had three general objectives which were to (1) gather information about resident's attitudes towards urban residential trees, whether they recognize trees services and disservices, and how these might relate to yard structure, (2) explore the extent by which resident's views and attitudes towards vegetation in terms of their origin (native versus non-native) may explain the observed low frequency of native plants in residential yards and, (3) evaluate the role of extreme events (i.e., hurricane events) disturbances on residential yards vegetation structure, composition and the provision of ecosystem services. These objectives were analyzed and written as independent chapters in this dissertation and below, I summarize the work for each and main findings for each. The dissertation finishes with a final chapter offering general recommendations that can be used to guide local forestry efforts based on my results.

In **Chapter 2** "*Attitudes towards residential trees and awareness of trees services and disservices in a tropical city*", the objectives were to understand (1) residents' attitudes towards residential trees and their association with household socio-demographic factors, (2) residents' awareness of tree's ecosystem services and disservices in relation to the trees' proximity to the resident (home versus neighborhood), and (3) if residents' attitudes towards trees could influence yard management using tree abundance as a yard management proxy. In 2011, 397 household surveys were conducted using a semi-structured questionnaire with open-ended and multiple-choice socio-demographic, attitude and behavioral questions related to residential green areas, which was

complemented with yard vegetation surveys data. Analyses yielded three main findings. First, most residents self-reported positive attitudes towards residential trees in general. Secondly, the awareness of ecosystem services and disservices differed across geographical scales and can also be influenced by the spatial proximity of tree location relative to the resident (home versus neighborhood), suggesting that differences in the spatial context of trees and residents may contribute to the variation in residents' awareness of tree ecosystem services or disservices. Respondents recognized more tree services (emphasizing shade, lower temperature, food and ornamental/aesthetics) and fewer disservices (emphasizing maintenance hardship, property damage and power line obstruction). Food provision and reduced structural integrity (damages to property) were more frequently recognized for home trees, while aesthetic or ornamental value and power line obstruction were most frequently acknowledged for neighborhood trees. Third, results suggested that expressed positive attitudes toward trees might be more influential in yard tree abundance than expressed attitudes when the study was conducted. Variation in positive attitudes partially explained current variation in yard tree abundance, along with residents' age, housing tenure, yard size and watershed location. A notable finding was that awareness of services generated by trees is rich but important benefits for climate change adaptation are being overlooked. Results emphasize the need to understanding these feedbacks, addressing awareness gaps in ecosystem services where they manifest, mitigating disservices that trees may generate, and the need to further research to identify the underlying values that residents hold toward trees as a way to facilitating current and future urban forestry strategies and green infrastructure planning that integrate ecosystem services frameworks.

In **Chapter 3** "*Social motivations and limitations to the cultivation of native plants in urban residential areas of San Juan, Puerto Rico*", resident's social internal factors (values, attitudes and behavioral intentions) that could be related to yard management decisions and influence yard vegetation structure and functions were evaluated. To that effect, 423 semi-structured household surveys were administered in 2013 to (1) evaluate attitudes toward native plants and whether socio-demographic factors and value orientations might influence these attitudes, (2) evaluate residents' willingness to exchange non-native plants for natives was associated with socio-demographic characteristics and preferred plant habits or ecosystem services, (3) identify preferred growth stage for gifted plants, and (4) explore the level of importance of wildlife habitat provision relative to other services of importance to residents. Most residents self-reported a strong preference for native plants which was mainly driven by "sense of place". Residents minimally recognized common conservation arguments (e.g., invasiveness avoidance, native plant adaptability) and these were not significantly associated to positive attitudes toward native plants. Residents also expressed a willingness to change their yard composition by replacement of a non-native with a native plant, an intention associated with preferences for ornamental shrubs and small food trees. Air purification and food provision were recognized as the most important ecosystem services and wildlife habitat provision the least important. The dominance of non-native species in urban residential spaces may point to discrepancies between investments of efforts by local forestry and conservation initiatives and their

effectiveness. Results reflect on the need to consider these prevailing local attitudes and values to guide effective conservation and urban forestry strategies.

In **Chapter 4**. "*Implications of hurricane-driven changes in vegetation structure and ecosystem services provision in residential yards of San Juan, Puerto Rico*" I took advantage of ongoing studies of residential backyards in San Juan to evaluate vegetation and ecosystem services changes following the effects of the 2017 hurricane season. I revisited 69 residential households where complete pre-hurricane i-Tree Eco inventories were available before hurricanes Irma and María to (1) evaluate hurricane effects on structure, composition and condition of woody species and large herbs, (2) estimate changes in functions and loss of ecosystem services, and (3) evaluate species-specific differences in hurricane-driven ecosystem services changes and mortality. Findings revealed hurricane-driven changes in the structure, composition and ecosystem services provision of inventoried stems. Yard vegetation experienced a 27% reduction of plant stems, high overall mortality (31%) and a 9.5% loss of species richness (9 out of 95). The yard vegetation remained highly dominated by non-native plants before (91.6%) and after (89.3%) the hurricanes. Stem reduction, tree cover loss and structural changes in yard vegetation translated into a reduction of ecosystem services of approximately 19.6% for air pollution reduction services, 19.9% of avoided runoff and 13.1% cooling energy savings for ecosystem services that depended on tree volume and structure. Food provision services exhibited higher losses than ornamental services based on the number of stem losses. The reduction in ecosystem services provided by yards following these extreme events could have implications for urban resilience by compromising the ability

to sustain locally desired functions when faced with disturbances. More research is needed to develop the urban forests using residential areas in ways that not only promote sustainability (by providing multiple ecosystem services) but also resilience to extreme storm events.

Chapter 2. Attitudes toward residential trees and awareness of tree services and disservices in a tropical city

1. Abstract

Attitudes toward urban residential trees and awareness of their ecosystem services and disservices may play an important role in management decisions of private residential green spaces with important consequences to urban sustainability. In 2011, 397 household surveys were conducted in six locations of the Río Piedras Watershed (San Juan, Puerto Rico) to evaluate residents' attitudes toward residential and neighborhood trees and their association with household socio-demographic factors, how awareness of services and disservices relate to the spatial proximity of trees (home versus neighborhood), and whether attitudes are associated with yard management (tree abundance). Most residents self-reported positive attitudes toward trees in general and these appeared to be more frequent than self-reported negative attitudes. Respondents recognized more tree services (emphasizing shade, lower temperature, food, and ornamental/aesthetics) and fewer disservices (emphasizing maintenance hardship, property damage, and power line obstruction). Not all tree services and disservices were equally recognized, and differences in the spatial context of trees and residents may contribute to the variation in residents' awareness of tree ecosystem services or disservices. Variation in positive attitudes partially explained the current variation in yard tree abundance, along with residents' age, housing tenure, yard size, and watershed location. Results have direct implications for urban forest planning and management in residential contexts.

2. Introduction

2.1 Background

Green infrastructure, a key component of urban social–ecological systems, is becoming more important in urban sustainability and planning discussions (Ahern, 2011; Gómez-Baggethun et al., 2013; Lovell & Taylor, 2013; Wu, 2014). Within cities, the term green infrastructure refers to a network of natural and constructed multifunctional green spaces that should be planned, developed, and maintained (Benedict & McMahon, 2006; Ahern, 2007). Elements of urban green infrastructure include many features of the city landscape, such as green roofs, parks, green corridors, isolated green patches, abandoned land, residential yards, churchyards, school grounds, and even cemeteries (Tzoulas et al., 2007). Urban forests, defined here as all the trees and associated vegetation present in a city (Ferrini et al., 2017), are a dominant feature of green infrastructure and urban forestry management strategies are incorporated into green infrastructure planning due to the commonality of the two approaches (Lafortezza et al., 2017). It is generally accepted that these green networks, and trees specifically, can provide a variety of ecosystem services that directly or indirectly influence human well-being in urban socio-ecological systems (Millennium Ecosystem Assessment, 2005; Escobedo et al., 2019). On the other hand, green spaces may inadvertently cause perceived or realized disservices that may affect human well-being (i.e., structural damage to private or public infrastructure, sources of pollen allergens) (Gómez-Baggethun & Barton, 2013) if not planned adequately. Managing for ecosystem disservices may facilitate the optimization of outcomes for well-being rather than just managing ecosystem services alone (Shackleton et al., 2016). One possibility is that the

interplay between perceived services and disservices of green infrastructure within urban spaces could also be influencing management decisions about these spaces and changing them over time (Lyytimäki et al., 2008; Escobedo et al., 2011; Shackleton et al., 2016). If so, urban planning with sustainability as a goal must address the links between human attitudes and behaviors toward green spaces and how these factors may influence the planning, development, and management of urban green infrastructure (Cook et al., 2012; Freeman et al., 2012; Casado-Arzuaga et al., 2013; Urgenson et al., 2013).

Residential green infrastructure has earned visibility as an important component of urban landscapes for its potential to provide multiple ecosystem services. Green infrastructure in the form of residential yards comprises a significant portion of urban space (González-García & Gómez Sal, 2008; Loram et al., 2008) and can also contribute to the general provision of ecosystem services (Gaston et al., 2005; Cook et al., 2012; Calvet-Mir et al., 2012). In addition, private yards can improve human health and well-being by providing physical and psychological benefits, facilitating connections with other people and with nature (Freeman et al., 2012), and by providing services that go beyond their utilitarian value (e.g., emotional, physiological, spiritual) (Dunnett & Qasim, 2000; Soga et al., 2017; Torres-Camacho et al., 2017; Wolf, 2017). Private yards are managed green spaces, and as such, their condition and dynamics will be greatly influenced by both social and ecological drivers, warranting an integrated multi-scalar, social-ecological approach to develop understanding of these systems (Pickett et al., 2001; Goddard et al., 2010; Cook et al., 2012; Muñoz-Erickson et al., 2014; Chowdhury et al., 2016; Meléndez-Ackerman, Nytch, et al., 2016). In that regard, many studies emphasize that anthropogenic factors could dominate over non-anthropogenic ones in

determining the characteristics of green infrastructure elements at the residential scale (Goddard et al., 2010; Cook et al., 2012; Meléndez-Ackerman et al., 2014; Wang et al., 2015). For example, in residential areas, socio-economic and social-psychological characteristics (e.g., household income, formal education, values, attitudes) are considered bottom-up drivers of biodiversity through their influence on garden management, which complement (or dominate over) top-down drivers (e.g., landscaping policies, local regulations, city-level strategies and management decisions) (Kinzig et al., 2005; Goddard et al., 2010; Meléndez-Ackerman et al., 2014). Ultimately, urban yards are likely to reflect (at least in part) individuals' choices that may be motivated by social drivers displayed as a variety of psycho-cultural and socio-economic traits (Cook et al., 2012; Kendal et al., 2012; Kiriscioglu et al., 2013; Van Heezik et al., 2014)**,** meriting an exploration of anthropogenic and non-anthropogenic interactions as possible drivers of household decisions.

Environmental psychology theories can help conceptualize relationships between humans and the environment and how these relationships might shape human actions. For example, theories like the Ajzen's Theory of Planned Behavior and Stern and colleagues' Value-Belief-Norm Theory link cognitive factors such as values and attitudes to human behavior, which can be useful to identify specific cognitive factors that can be linked to behavioral intention (Ajzen, 1991; Stern et al., 1999). Others, like Rokeach's Cognitive Hierarchy Theory, provide a hierarchical relationship between these concepts to help understand how values can indirectly influence behavior through attitudes (Milfont et al., 2010; Baur et al., 2016). Building from this model, Homer and Kale proposed a value-attitude-behavior hierarchy model, where they emphasize that attitudes play a mediating

role between individual values and behaviors, which has since been applied to environmental issues (Homer & Kahle, 1988; Milfont et al., 2010). Attitudes are then defined as negative or positive evaluations of a specific object (i.e., preference, liking, or disliking) (Ives & Kendal, 2014) and can be used a measure to evaluate behavior toward objects, such as a tree (Kaltenborn & Bjerke, 2002; Balram & Dragićević, 2005; Tyrväinen et al., 2005; R. E. Jones et al., 2013). It has been documented that people hold strong positive attitudes toward urban trees (Hull, 1992; Lohr et al., 2004; Schroeder et al., 2006; Zhang et al., 2007; R. E. Jones et al., 2013; Camacho-Cervantes et al., 2014; Avolio et al., 2015; Oliveira Fernandes et al., 2019; Gwedla & Shackleton, 2019), which, based on the value–attitude–behavior hierarchy model, can be used to explore potential behavioral outcomes in urban spaces, such as residential yards. For example, specific attitudes of individuals toward a set of objects may influence explicit decisions (Heberlein, 2012) and flora found in residential yards may likewise be a reflection of people's preference toward certain plant characteristics, such as large flowers or green foliage (Kendal et al., 2012). Residents' attitudes toward certain tree attributes (e.g., large canopy, tree height, low maintenance), ecosystem services (e.g., shade, beauty), and even a tree genus (e.g., *Acer*), can also be positively correlated with urban yard composition (number of trees containing the attributes) (Avolio et al., 2018). Thus, human attitudes could be an important psychological driver influencing the composition of urban residential yards.

Human attitudes are an important component of urban social–ecological systems that relates to the actual interactions of people with urban green space. Studies documenting human attitudes toward trees are more frequently used in the literature to

understand the relationships between residents and urban trees (Flannigan, 2005; Schroeder et al., 2006; Dilley & Wolf, 2013; Shakeel & Conway, 2014; Conway & Yip, 2016; Almas & Conway, 2018). Research suggests that these people-environment interactions consist of complex and dynamic exchanges that enrich and shape one's knowledge of the environment, while shaping the environment as well (Stokols et al., 2013). These interactions may include experiencing ecosystem disservices that lead to detrimental human-environment relationships (Lyytimäki et al., 2008). Experiencing disservices can influence negative attitudes, some examples include allergies produced by city trees (Lohr et al., 2004), street trees blocking visibility (Gorman, 2004), damaging sidewalks (Avolio et al., 2015), producing debris (Flannigan, 2005)**,** or roots of yard trees damaging house walls (Gwedla & Shackleton, 2019). Studies that look at tree planting and removal motivations in residential private spaces are scarce (Kirkpatrick et al., 2012; Conway, 2016), but some studies in western cities have also documented perceived risks to property and management hardships (e.g., processing of vegetation debris) as stated reasons behind tree removal decisions (Summit & McPherson, 1998; Head & Muir, 2005; Kirkpatrick et al., 2012; Conway, 2016; Avolio et al., 2018; Guo et al., 2019). The characterization of ecosystem disservices and how they vary along with socio-economic factors is an understudied area of research when using ecosystem services approaches (Lyytimäki et al., 2008; Lyytimäki & Sipilä, 2009; von Döhren & Haase, 2015; Wang et al., 2015). Scientists argue that studies evaluating green infrastructure within an ecosystem services framework need to incorporate ecosystem disservices, as these may very well influence human attitudes and behavior toward green infrastructure (von Döhren & Haase, 2015; Shackleton et al., 2016; Conway & Yip, 2016) and may pose

challenges toward the implementation of green infrastructure policies in cities (Zuniga-Teran et al., 2019).

Attitudes are context dependent (Abasolo et al., 2008; Warren et al., 2011) and have been understudied in Latin American cities relative to cities in other regions of the world within the context of green infrastructure planning (Barnhill & Smardon, 2012; V. Turner & Jarden, 2016; Russo & Cirella, 2018; Speak et al., 2018). Moreover, studies about attitudes toward green infrastructure in Latin American cities have focused on public or common green spaces and less so on private spaces (Medellín, Colombia: Anguelovski et al., 2019; San Juan, Puerto Rico: López-Marrero et al., 2011; Santiago et al., 2015). For this study, I present a case study that builds upon prior work on the social-ecological processes driving residential vegetation dynamics in the Río Piedras Watershed (RPWS) located in the city of San Juan, in the Caribbean island of Puerto Rico. The objectives are to evaluate: (1) how household demographics and watershed location (site) drive positive and negative attitudes toward urban trees located in two referenced green spaces (residential yards and neighborhoods); (2) whether awareness of ecosystem services and disservices differs according to spatial proximity of the tree (home versus neighborhood); and (3) whether attitudes may drive yard management outcomes (tree abundance) within the Río Piedras Watershed in the San Juan Metropolitan Area (SJMA). I hypothesize that attitudes toward trees would vary between different demographic groups (defined by residents' age and housing tenure status) and watershed location based on prior work showing a positive relationship between these variables and tree abundance (Meléndez-Ackerman et al., 2014). Likewise, I hypothesize that residents may recognize tree services and disservices differently when trees are

located inside their yards relative to when they occur in their neighborhood (i.e., small-scale context-dependent attitudes) and that there is a relationship between attitudes and yard tree abundance. Residents reporting positive attitudes (preference, trees as beneficial) would have more trees and residents reporting negative attitudes (trees as problematic) would have fewer trees. Results contribute to the growing work on the use of social-ecological research to support the application of ecosystem service approaches in Latin American cities and the methods can also serve as model of inquiry for other insular cities in the Caribbean region, where the nature-based solutions may be seen as a strategy for climate change adaptation.

2.2 Social drivers of green infrastructure in San Juan, Puerto Rico

The SJMA is the largest urban area on the island of Puerto Rico. In the 1900s, it experienced rapid urbanization that led to the loss of forest cover (Lugo et al., 2011)**,** a loss that has been linked to socio-economic, governance, and physical-spatial disparities in some neighborhoods (Ramos-Santiago et al., 2014). In the municipality of San Juan, a 42% green cover was estimated for 2002, with the highest amount located on the northern part as isolated small patches of forest, most of which occur in private yards (Ramos-González, 2014). Within the Río Piedras Watershed (RPWS), residential yards vary in the amount of green and tree cover (Vila-Ruiz et al., 2014)**,** found to be partly associated with variation in yard size, with some differences moderately explained by housing tenure, age, and geographic location (Meléndez-Ackerman et al., 2014). Considering these factors have not been able to explain all the variation in yard vegetation cover, one hypothesis is that these associations are also partly driven by differential attitudes toward

residential vegetation and their services across social groups and sites within the watershed (Ramos-González, 2014; Meléndez-Ackerman, Nytch, et al., 2016). At least two studies conducted in the San Juan Metropolitan Area (SJMA) of Puerto Rico have shown that while urban dwellers are able to identify many of the ecosystem services provided by urban green infrastructure, green spaces may also generate disservices (López-Marrero et al., 2011; Santiago et al., 2015). The combined results of these studies also indicate that perceptions of services and disservices may differ across short geographic ranges within a tropical city like San Juan (López-Marrero et al., 2011) and shape the preferences of users for particular green infrastructure designs (Santiago et al., 2015). For example, visual aesthetics (or scenic value) was the ecosystem service most frequently mentioned by visitors of Plaza de la Convalecencia, a small town square in San Juan (Santiago et al., 2015)**,** but it was only the eighth most commonly recognized service for residents of the Río Piedras Watershed (RPWS) when asked about urban forests patches (López-Marrero et al., 2011). Thus, it is possible that values and perceptions of urban vegetation are influenced by the degree of interaction between residents and green spaces, or their sense of ownership of these spaces (Peckham et al., 2013; Martini et al., 2014; Ordóñez-Barona & Duinker, 2014a; Ordóñez-Barona, 2017).

In San Juan, the protection of trees is being recognized by state and city local institutions in various ways. The Puerto Rico Forest Action Plan of 2016 includes an urban forest section, which describes the benefits of urban forests for well-being and define urban forests as priority landscapes, although it does not define specific guidelines to these effects (Department of Natural and Environmental Resources, 2016). A goal of the plan is to enhance the public benefits associated with trees and forests and some of

the priorities outlined in the plan include the need for information related to ecosystem services and other related services of public and private lands. Article 9 of the Puerto Rico Forest Act (Law No. 133 of 1975) prohibits cutting and damaging public or private property trees that have characteristics indispensable or necessary for forestry use, that are at risk of extinction, located in plazas and parks, or that are indispensable for some purpose of essential use (not explicitly defined). A permit is required by the Puerto Rico Department of Natural and Environmental Resources (DNER) for cutting or grooming said trees (Puerto Rico Planning Board Regulation No. 25). The Municipality of San Juan does not have a specific urban forestry or management plan in place but there are several independent tree giveaway campaigns driven by local public and private entities. A collaboration from the Municipality's Office of Planning and Territorial Management, the Puerto Rico Department of Natural and Environmental Resources (DNER), and the San Juan ULTRA (Urban Long-term Research Areas) Network led to public seminars on the importance of green spaces in flooding risk management in the Río Piedras river. This collaboration was followed by municipal resolution OPOT-2015-1, which called for the creation of The Río Piedras Alliance (*Alianza del Río Piedras*) as a unifying network for stakeholders of the Río Piedras Watershed and the development of a green infrastructure plan for the Municipality of San Juan in coordination with corresponding offices and agencies, although the plan is pending preparation. The San Juan Bay Estuary Program (SJBE; http://web.estuario.org), which is the only tropical estuary in the U.S. National Estuary Program, a non-regulatory program of the U.S. Environmental Protection Agency, is also developing a green infrastructure plan for the SJBE basin. Residential areas comprise a considerable proportion of the landscape in the San Juan Metropolitan

Area (Brandeis et al., 2014). Therefore, yard spaces could contribute to the overall green infrastructure of the city. Beyond what exists at the state level, there are no specific laws that regulate tree management in private households, but a few gated communities do regulate their planting practices.

3. Methods

3.1 Study site

This research was conducted in the Río Piedras Watershed (RPWS) located in the San Juan Metropolitan Area, the largest urban area on the island of Puerto Rico (Figure 1). This urban watershed covers an area of 67,000 m^2 and the Río Piedras River has been extensively modified due to anthropogenic activities (Lugo et al., 2011). Based on Holdridge's life zone system it is classified as a subtropical moist forest zone (Ewel & Whitmore, 1973) and presents a mean annual precipitation that ranges from 1509 mm to 1755 mm and a mean annual temperature of 25.7 °C, with a rainfall season that coincides with the hurricane season from June to December and a dry season from January to April (Lugo et al., 2011). The RPWS is part of the San Juan Bay Estuary for which residential land use has been estimated at 36% as the dominating land use (Brandeis et al., 2014). The RPWS and the island were affected by two intense hurricane events during the 2017 season, with considerable effects to grey and green infrastructure (Hu & Smith, 2018; Uriarte et al., 2019). An assessment of tree cover loses for the San Juan Municipality has been estimated to be as high as 24.8% of the total tree cover (Meléndez-Ackerman et al., 2018).

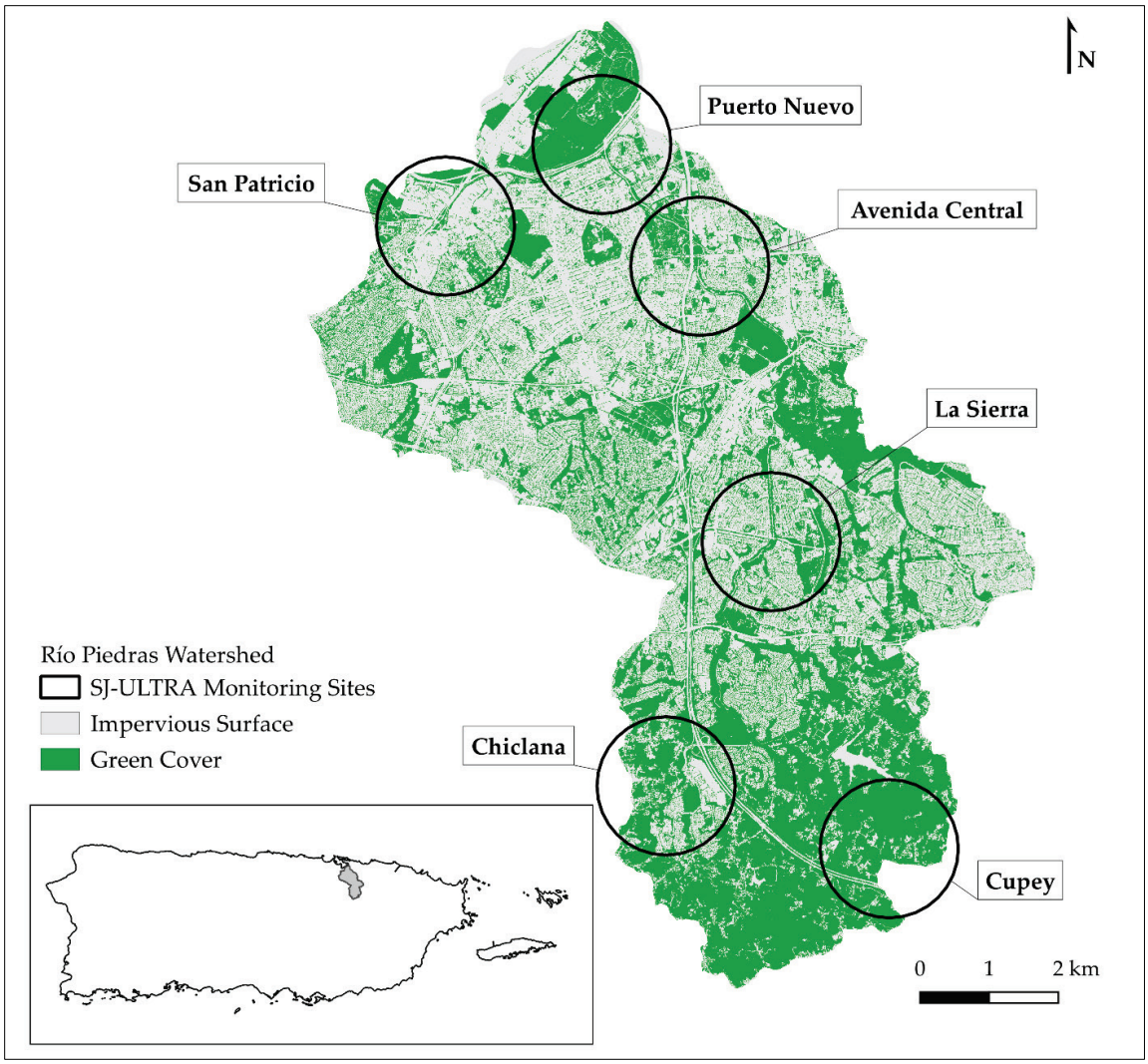

Figure 1. Location of the Río Piedras Watershed with its six monitoring sites and green area cover. Source: Martinuzzi et al., 2018.

3.2. Sampling design

Data was collected through household and yard surveys via a convenient-based recruitment at six permanent monitoring locations within the Río Piedras Watershed (RPWS) based on the San Juan Urban Long-Term Research Area (ULTRA) Collaborative Network stratified sampling scheme (Garcia-Montiel et al., 2014; Meléndez-Ackerman et al., 2014; Vila-Ruiz et al., 2014; Meléndez-Ackerman, Nytch, et al., 2016). The six sites have been studied to address a variety of social-ecological

questions at residential scales since 2011 (Garcia-Montiel et al., 2014; Muñoz-Erickson et al., 2014; Santiago et al., 2014, 2015; Vila-Ruiz et al., 2014; Meléndez-Ackerman, Nytch, et al., 2016; Torres-Camacho et al., 2017) and lie across a rural–urban and elevation gradient of grey coverage (Table 1). At each site a circular buffer zone of 1 km radius was overlaid on aerial photos of San Juan and a street vector file of access roads using ArcGIS v. 9.3 software (Esri, 2008). Roads were selected by generating a random sample in Microsoft Excel with road codes, and surveys were administered depending on willingness to participate and availability of residents by visiting each household on a selected road. The survey was carried on different days of the week during daylight hours, the majority during weekends from 9:15 am to 14:30 pm. Surveys were conducted face-to-face by Spanish speaking students and faculty from the University of Puerto Rico Río Piedras Campus. Surveyors received training on how to conduct surveys and obtained certificates upon successful completion of the National Institute of Health web-based training course "Protecting Human Research Participants". All research protocols with human subjects were designed and implemented in accordance with University of Puerto Rico Institutional Review Board (IRB) requirements (IRB #1011-013). A consent form was provided to all participants explaining the voluntary nature of the survey, a privacy and confidentiality clause, and contact information for principal investigators and the respective IRB office. The minimum sampling size was six households per street and a maximum number of 80 households were sampled per buffer zone.

Table 1. Site descriptions and number of single-family residential units for each site included in the study (N=397).

Site	N	Urban Cover	Housing Density	Watershed Location	Elevation (m.a.s.l.)
San Patricio	43	high	high	low	7
Puerto Nuevo	63	high	high	low	0
Avenida Central	78	high	high	mid	3
La Sierra	72	intermediate	intermediate	mid	20
Chiclana	64	low	low	upper	59
Cupey	77	low	low	upper	157

3.3. Social and vegetation surveys

Field surveys were conducted from January 2011 to July 2011, with an additional survey in October 2011. The survey consisted of a combination of open-ended and multiple-choice questions about perceptions and behavior toward green areas, as well as socio-demographic characteristics. The questionnaire was pre-tested to ensure an appropriate question format, wording, and order. To assess attitudes, residents were asked if they preferred to have trees in their properties (or not) and the reasons why, if trees were perceived as beneficial and/or problematic, followed by open-ended questions about the perceived benefits or problems trees generated (see Appendix A, Table A1). From the survey the following information was gathered on households' socio-demographic and economic variables: the respondent's age, gender, marital status, years of formal education, combined annual household average income (US dollar), and housing tenure. Deductive coding was used for each open-ended question and benefits and problems responses were classified into ecosystem services or disservices using categories from available literature (see Appendix A, Tables A2). The coding scheme was agreed upon through discussion meetings with two lead surveyors (ecologist and social scientist) and two coders. Ecosystem services were classified according to the Millennium Ecosystem

Assessment (2005) framework (Millennium Ecosystem Assessment, 2005) into Regulation (R), Support (S), Provision (P), and Cultural (C) services. Likewise, responses to the question about tree problems were classified following von Döhren and Haase's (2015) classification for type of ecosystem disservices developed from a comprehensive review of available publications according to economic, ecological, health, and psychological impacts. The social survey was complemented with available data on yard vegetation surveys that were used to extract the following variables: number of tree stems and yard area. All woody plants of over 2 cm diameter were included in the survey, habit and species were determined, and photographs of each individual were taken to confirm identification in laboratory. Yard area was estimated from aerial imagery using Google Earth Pro v. 7.0.

3.4 Statistical analyses

3.4.1 Preferences and attitudes versus household socio-economic variables

I ran binomial logistic regressions to evaluate the relationships between household socio-economic variables and the likelihood that residents would (1) prefer trees on their property, (2) express positive attitudes toward trees on their property or in their neighborhood (trees as beneficial), or (3) express negative attitudes toward trees on their property or neighborhood (trees as problematic). Household socio-economic variables included: gender, housing tenure, civil status, age, years of formal education, annual family income, and household size. Only observations that did not have any missing values were considered. This resulted in a smaller sample for each model. Qualitative variables were coded as binary variables as follows: gender (female = 1, male = 0),

housing tenure (owner = 1, renter = 0), civil status (married or living with partner = 1, not married or divorced = 0). Response variables related to the preference for trees, trees as beneficial or problematic, were also coded as binary variables (yes = 1, no = 0). The variable site (categorical) was also included as an explanatory variable in logistic regression analyses with site = 'Cupey' as the reference category. I performed a McNemar's Test with continuity correction to test if the proportion of responses of whether home trees are beneficial or problematic significantly increased or decreased when asked about neighborhood trees. For samples of fewer than 25 records of discordant cells (cells that reflect difference in scale responses, "yes" to one scale and "no" to the other, and vice versa) a binomial distribution was used.

3.4.2 Services and disservices awareness versus scale

I evaluated the frequencies of responses of specific ecosystem services and disservices to determine those most commonly identified by residents. New variables that represented whether a resident mentioned a specific service or disservice when referring to home trees and neighborhood trees were created for each of the six most frequently indicated services (shade provision, food provision, lower temperature, oxygen production, aesthetic value, air purification) and the four most common disservices (maintenance hardship, reduced structural integrity, power lines obstruction, induces pests). McNemar's Tests were used to test whether the proportion of affirmative responses of each specific service or disservice by home trees changed when asked about home versus neighborhood trees. Tests were performed by site and using the pooled data for all sites. All the above analyses were run using SPSS v.25 (IBM Corp. Released, 2017).

3.4.3 Tree attitudes versus yard tree abundance

I used spatial regression analysis to test for the effects of tree preferences (yes = 1, no = 0), trees as beneficial (yes = 1, no = 0), trees as problematic (yes = 1, no = 0), total number of services and disservices at the household scale, on the number of yard tree stems (a yard structure variable). Household socio-economic variables (age and housing tenure) and yard size were also included, building on previous work on the role of these variables on yard tree abundance (Meléndez-Ackerman et al., 2014). For this analysis, I eliminated all cases that contained missing values in any of the variables, as well as a case whose coordinates were inconsistent between the social survey and the vegetation survey, yielding a total of 359 observations. Following Anselin et al. (2006), ordinary least squares regression models (OLS) were generated to test relationships with each of the dependent variables. I used Moran's I statistics and Lagrange Multiplier test statistics to detect spatial autocorrelation. Akaike information criterion was used to select the best fit model for the prediction of the number of yard trees. Spatial analysis was run using GeoDa v.1.12.

4. Results

4.1 Household socio-demographic profiles

The majority of respondents were females (60.6%) and the average respondent's age was 56.6 years. More residents surveyed were married or living with a partner (56.5%) rather than single or divorced (43.5%), and the large majority owned their properties (Table 2). Respondents had an average of 14.7 years of formal education, indicating that on average residents had at least completed a high school diploma and had

spent at least two years pursuing a university degree. The average household size was 2.9 persons, with an annual household mean income of $33,110. Although there is a natural skewedness in the San Juan population toward females and older residents, when compared to official U.S. Census data for the year 2010 these variables were overrepresented in my sample. The sample was overrepresented by single household owners living within the boundaries of the RPWS and considerations need to be made when interpreting the data, since representativeness was not assessed.

Table 2. Descriptive statistics for the seven socio-economic characteristics of 397 households at the RPWS.

A. Categorical	**Variable**	**Class**	**Frequency**
1	gender	female	238
		male	155
2	civil status	single or divorced	171
		married or living with partner	222
3	ownership	owned	341
		rented or other	56
B. Continuous	**Variable**	**Descriptive Statistics**	**Value**
4	age (years)	mean ± se	56.64 ± 0.956
		max	96
		min	18
5	household income	mean ± se	$33,110.80 ± $1,390.50
		max	$80,000
		min	$5,000
6	years of formal education	mean ± se	14.73 ± 0.185
		max	23
		min	6
7	household size	mean ± se	2.95 ± 0.081
	(persons per household)	max	15
		min	1

4.2. Socio-demographic profiles and attitudes toward trees

Most respondents within the watershed (85.3% of 395 responses) preferred having trees on their property. Likewise, many residents responded that their home trees

provide benefits (89.2% of 397 responses); the same pattern was observed when asked about neighborhood trees (88.7% of 397 responses). At the same time, 35.4% (out of 395) of respondents indicated that home trees caused problems, and a similar percentage (34.5% out of 397) indicated that neighborhood trees did. An exact McNemar's test using pooled watershed data found no significant differences across scales (home versus neighborhoods) in the proportion of residents that identified trees as beneficial, nor the proportion of residents that identified trees as problematic (Figure 2). When differences in the proportion of residents that identified trees as beneficial or problematic were also evaluated at each watershed location (site), none of the tests reflected differences across sites (all $X^2 < 3.2$, $p > 0.06$).

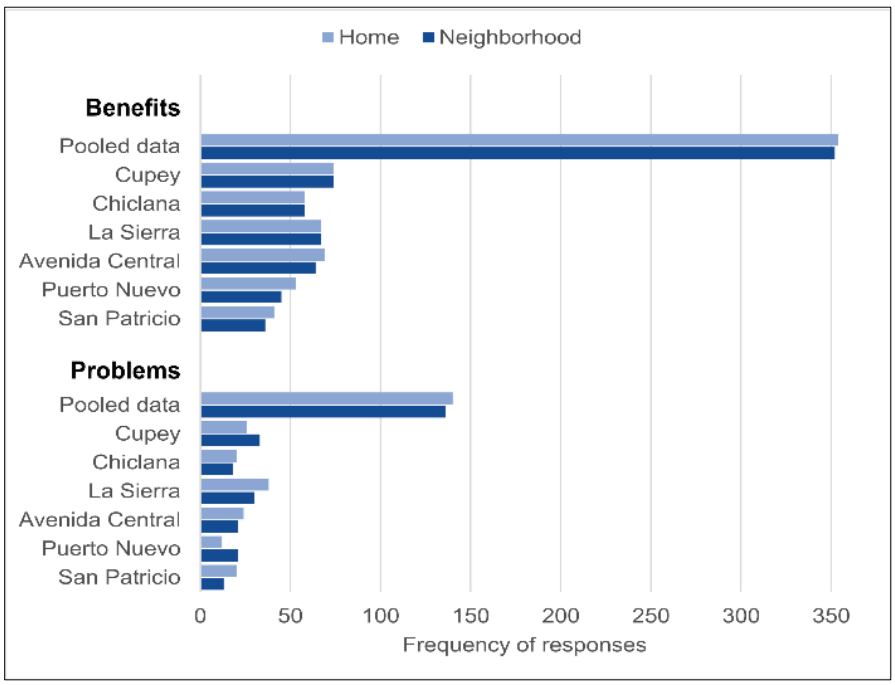

Figure 2. Frequency of resident affirmative responses (yes) to the question of whether trees provide benefits or problems for total responses and per site, a comparison between home and neighborhood trees. No significant scale differences were found for exact McNemar's tests in neither the pooled data nor the site data (all p's > 0.05).

Logistic regression yielded no significant associations between household-level socio-economic variables and the likelihood of preferring home trees, and recognizing home trees as beneficial or problematic. However, the likelihood as to whether home trees were identified as beneficial (or problematic) was somewhat related to location (site) within the watershed (Table 3). In the Puerto Nuevo and Avenida Central sites (lower watershed sites), residents were less likely to perceive household trees as beneficial when compared to residents in Cupey (upper watershed). Also, the odds of finding home trees problematic was found to be 2.45 times higher for La Sierra (mid-watershed) residents versus Cupey (upper watershed). Models did show that residents' age and gender were factors associated with the likelihood of identifying neighborhood trees as beneficial (Table 3). Males and older residents were less likely to identify benefits derived from neighborhood trees than females and younger residents. In addition, residents from San Patricio, Puerto Nuevo, Avenida Central, and Chiclana were less likely than those from Cupey to acknowledge neighborhood trees as beneficial. None of the socio-economic or site variables were related to the likelihood of recognizing trees as problematic at the neighborhood scale. Overall, model variation in positive and negative attitudes toward home and neighborhood trees considering socio-economic and site factors was small and always below 19% of the total explained variation.

Table 3. Regression coefficients from binomial logistic multiple regression analyses testing the relationship between household socio-economic variables and the likelihood of positive or negative responses by residents of the RPWS on whether they prefer trees in their property, and perceive benefits and problems from trees at the property and their neighborhoods. Significant values in bold.

Independent Variables	Prefer Trees on Property	Property Trees as Beneficial	Property Trees as Problematic	Neighborhood Trees as Beneficial	Neighborhood Trees as Problematic
gender (1)	−0.139	−0.234	0.039	**−0.739** †	−0.244
age	−0.009	−0.002	0.005	**−0.025** *	−0.001
civil status (1)	0.264	−0.034	0.410	−0.375	−0.238
years of formal education	0.116	0.021	−0.010	0.065	0.03
annual average income	0	0	0	0	0
ownership (1)	0.972	−0.448	−0.351	0.032	0.266
household size	0.163	0.018	−0.003	0.107	−0.071
site					
(1) San Patricio	−1.482	−0.031	0.576	**−2.061** *	−0.808
(2) Puerto Nuevo	−1.051	**−2.563** **	−0.781	**−1.851** *	−0.146
(3) Avenida Central	−1.198	**−1.616** †	−0.153	**−2.433** **	−0.668
(4) La Sierra	−0.957	−1.306	**0.895** *	−1.392	−0.106
(5) Chiclana	−0.546	−1.307	−0.052	−1.432 *	−0.583
X^2	19.339	25.556	24.304	33.586	10.241
df	12	12	12	12	12
Nagelkerke R^2	0.098	0.141	0.093	0.183	0.04
N	345	347	345	347	347
p	0.081	**0.012** *	**0.018** *	**0.001** **	0.595

† $p = 0.05$, * $p < 0.05$, ** $p < 0.01$

4.3. Awareness of ecosystem services

Residents mentioned a total of 43 different ecosystem services when we pooled responses about why residents preferred trees and why they offered benefits (see Appendix A, Table A3). Services were distributed by type as follows: 16 cultural, 11 support, 8 regulation, and 7 provision. Shade (22.43%), temperature reduction (18.45%), food provision (15.44%), aesthetic value (12.38%), oxygen production (11.66%), and air purification (6.58%) were the most commonly indicated ecosystem services. Less than two percent of the residents mentioned services that included habitat for flora and fauna,

several regulation services (natural hazard moderation, erosion control, carbon sequestration, noise reduction), and cultural services (privacy, relaxation, spiritual).

Statistically significant differences across scales (home versus neighborhood trees) were evident for at least four services (food provision, air purification, aesthetic services, and shade provision, Figure 3). However, these scale differences in the identification of certain services was more evident for food provision, where residents were twice as likely to indicate food provision as a service from home trees than from neighborhood trees (Figure 3). This trend was significant using pooled responses, and in all but two sites located in the lower watershed (Puerto Nuevo and San Patricio). The proportion of residents who mentioned shade as an ecosystem service was higher for home trees than for neighborhood trees, but this was significant when using the pooled data and only for the Cupey site when each site was analyzed independently (Figure 3). In contrast to food provision and shade, air purification and aesthetic value were recognized more often for neighborhood trees (Figure 3). These tendencies were only significant when evaluated using the pooled data, but not when evaluated for individual sites. Neither oxygen production nor temperature reduction were associated with significant differences in perception for home or neighborhood trees when using the pooled data (Figure 3). However, residents of San Patricio and La Sierra mentioned temperature reduction more frequently as a service from neighborhood trees than from home trees (Figure 3).

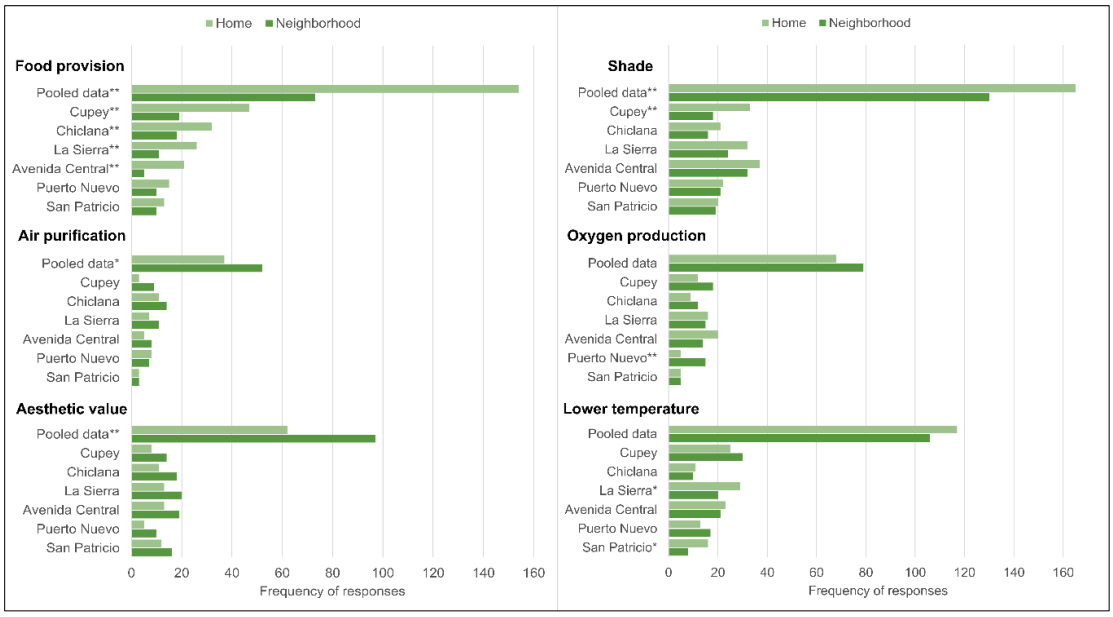

Figure 3. Frequency distribution of responses of the six most common services between home versus neighborhood trees (per site and aggregate). Symbols indicate significant differences using McNemar's tests (*$p < 0.05$, **$p < 0.01$). X^2 values for significant McNemar's tests ranged from 4.083 to 52.893.

3.4. Awareness of ecosystem disservices

Residents identified a total of 18 ecosystem disservices (see Appendix A, Table A4) including 8 economic, 2 health, 6 psychological, and 4 related to ecological impacts. The most frequently mentioned disservices were those related to economic impacts: maintenance hardship (37.11%), reduced structural integrity (21.91%), and power lines obstruction (10.82%). They were followed by an ecological disservice, inducing pests (7.73%). Together, these disservices represent over three-quarters of all pooled responses. Other disservices mentioned were: leading to neighbor disputes (4.64%), increased risk to personal injury (3.09%), and potential for property damage due to natural hazards (4.38%). In terms of differences in the proportion of responses considering disservices between home and neighborhood trees, reduced structural integrity was mentioned significantly more often at the home scale when using pooled resident data, but

differences were not significant for any given site (Figure 4). Likewise, powerline obstruction was mentioned more often as a disservice for neighborhood trees than for home trees in Cupey; results were not replicated in pooled watershed data (Figure 4). Neither maintenance hardship nor induced pests showed differences in the rate of responses across scales (home versus neighborhood trees).

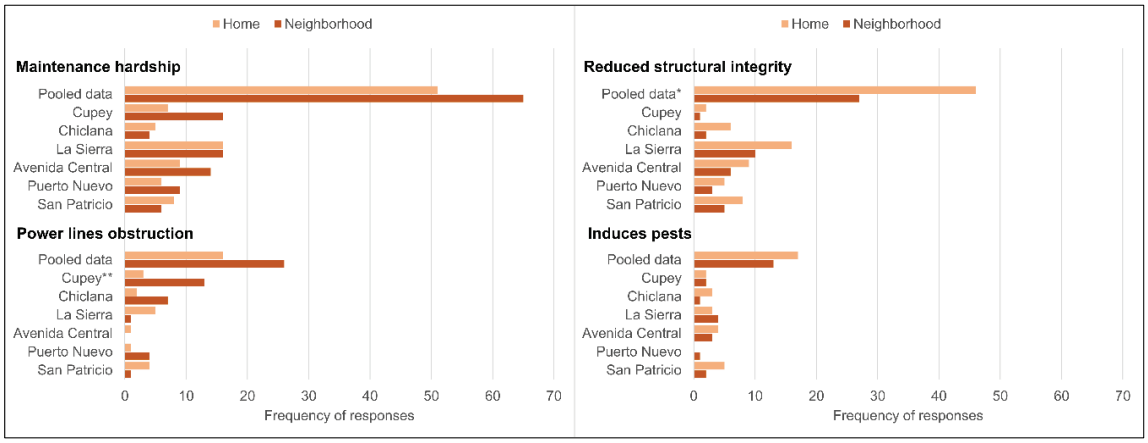

Figure 4. Frequency distribution of responses of four more common disservices from home versus neighborhood trees (per site and aggregate). Symbols represent significant values for McNemar's test (*$p <$ 0.05, **$p < 0.01$). X^2 values for significant McNemar's tests were both 6.75.

4.5 Relationships between overall respondent profile and yard vegetation

Multiple regression analysis (Table 4) showed that the number of tree stems was positively associated with some of the seven variables in the model ($p < 0.001$). The model explained 46% of the total variation and the strongest regression coefficients were yard area, tree preference, and ownership (housing tenure). Recognition of home trees as beneficial showed a lower regression coefficient, which approached acceptable levels of significance, while trees viewed as problematic was not a factor contributing in a significant way to the variation in the number of trees per yard in this system.

Table 4. Regression coefficients from ordinary least squares (OLS) multiple regression analyses testing the association between household social variables and total number of tree stems. Coding for ownership as follows: owner = 1, renter and other = 0. Both number of tree stems and yard area were transformed to Log10 (N+1) and cases with missing values were excluded from the analysis for a total of N = 359. Significant values in bold.

Independent Variables	Tree Stems
ownership	**0.217** **
age	**0.005** **
site	**0.059** **
preference trees	**0.230** **
home trees as beneficial	**0.177** *
home trees as problematic	−0.025
yard area	**0.604** **
Whole Model Statistics	
F_{OLS}	44.974
df	351
AIC	375.516
R^2	0.473
Adjusted R^2	0.462

** Significant values $p < 0.001$; * Significant value $p < 0.05$.

5. Discussion

In this work, the goal was to understand residents' attitudes toward residential and neighborhood trees and their association with household socio-demographic factors, residents' awareness of trees' ecosystem services, and disservices in relation to the trees' proximity to the resident (home versus neighborhood), and if residents' attitudes toward trees could influence yard management using tree abundance as a yard management proxy. The study had three main findings. First, most residents self-reported positive attitudes toward trees in general and these appeared to be more frequent than the self-reporting of negative attitudes. Second, not all tree services and disservices were equally recognized by residents, but variation in their awareness of tree ecosystem services or

disservices may be influenced by the spatial proximity of trees relative to the resident (home versus neighborhood). Finally, results suggest that positive attitudes toward trees might be more influential than negative ones in yard planting decisions when the study was conducted. Below, I discuss the significance of these findings, considering other studies and the implications for the use of an ecosystem services framework for urban forestry and green infrastructure planning with an emphasis on green residential spaces.

4.1. Drivers of attitudes toward trees

The fact that most residents had positive opinions toward trees rather than negative ones is consistent with the literature (Hull, 1992; Lohr et al., 2004; Schroeder et al., 2006; Zhang et al., 2007; Jones et al., 2013; Camacho-Cervantes et al., 2014; Avolio et al., 2015; Oliveira Fernandes et al., 2019; Gwedla & Shackleton, 2019), but most of these studies evaluated attitudes toward trees in public spaces (but see Jones et al. 2013, Avolio et al. 2015, Gwedla & Shackleton 2019)**.** My results are consistent with these findings but add to the idea that there is complexity as to what may drive attitudes toward trees. For example, this study showed that proximity of trees to a resident (home versus neighborhood) is not likely to influence the frequency of self-reported positive attitudes. On the other hand, results also suggested that self-reported negative attitudes (i.e., viewing trees as problematic) among residents may indeed be influenced by their proximity to trees as well as the socio-economic profile of the resident. Specifically, men and older residents were more likely to view trees as problematic, but this result was only evident for neighborhood trees, not home trees. Other studies have alluded to the complexity of factors that may drive attitudes toward trees, but with different results. In

Lohr et al. (2004), most respondents strongly agreed on the importance of city trees, but men and lower income younger residents with less formal education were less likely to strongly agree. Avolio et al. (2015) also found that a higher income level ($150,000 or more) was strongly correlated with the importance of yard trees, but not public trees. While I found attitudes toward trees by gender to be significant, the relationship was weak (i.e., $R^2 < 18\%$ for models including gender) relative to those found in other studies, reinforcing the need to avoid generalization of the way social groups may self-report their attitudes toward trees and the importance of local information to understand the social constructions on their environment.

Due to the face-to-face and self-reporting nature of the recruitment strategy, positive attitudes might be overrepresented due to the enhancement of social desirability bias. Face-to-face self-report surveys can be affected by what respondents consider to be socially acceptable, to be viewed favorably by others (Fisher, 1993). A reason for caution in assuming most people have a positive attitude toward trees is the argument that negative attitudes might be stigmatized by prevailing normative discourses of trees as being "universally good" and "should be loved by everyone" (Braverman, 2008; Kirkpatrick et al., 2012). This research strived to minimize social desirability bias by including open-ended questions on tree benefits and problems and by explaining the voluntary and confidential nature associated with participation. Future studies might address social desirability bias using more in-depth qualitative analysis (e.g., semi-structured interviews) or mixed methods approaches that give subjects more space to discuss their attitudes and perceptions toward trees in more detail.

5.2 Awareness of ecosystem services and disservices

The most frequently mentioned services were shade, lower temperature, food provision, and ornamental/aesthetics. The most frequently mentioned disservices were maintenance hardship, property damage, and power line obstruction. The recognition of shade and temperature regulation is not surprising given the tropical environment of San Juan, which has been shown to exhibit a strong urban heat island effect (Murphy et al., 2011) and temperatures and extreme heat episodes have been on the rise for the past 40 years as a result of climate change (Méndez-Lázaro et al., 2016; Méndez-Lázaro et al., 2018). Awareness of shade is consistent with previous temperate city studies of residents ranking shade along with aesthetic value (i.e., ornamental, beauty) as the most important services of urban trees (Gorman, 2004; Lohr et al., 2004; Flannigan, 2005; Schroeder et al., 2006; Avolio et al., 2015, 2018), in addition to oxygen supply in Morelia Mexico, (Camacho-Cervantes et al., 2014) and fruit provision in South Africa (Gwedla & Shackleton, 2019). My findings are also consistent with previous studies where residents ranked damages to structures (e.g., sidewalks) and maintenance problems (e.g., fallen debris) higher than other disservices (Gorman, 2004; Flannigan, 2005; Schroeder et al., 2006; Camacho-Cervantes et al., 2014; Avolio et al., 2015; Gwedla & Shackleton, 2019). However, the extent of awareness of all ecosystem services and disservices was not necessarily uniform across cities in the cited studies. It is particularly notable that food provision and power line obstruction were frequently mentioned by respondents in the Río Piedras Watershed, but these were not prevalent in resident responses in most studies of attitudes toward residential trees. While evaluating the effect of those differences is beyond the scope of this study, these deserve future consideration when designing studies

seeking to evaluate ecosystem service awareness. First, there were methodological differences on the exploration of attitudes between this study and other studies. Most studies provided a list of specific services and disservices to be ranked by residents, where food provision (a benefit) or power lines (a problem) were seldom included. Studies also differed in the metrics employed to evaluate services and disservices (e.g., scales, ranking values, attitude statements, visual scenarios) or the spatial and numerical context of the reference trees (e.g., one tree or many, yard street or street tree, private or public tree), factors which may influence attitude responses.

Variation in attitudes toward residential trees and their related services and disservices have been found to differ according to the location of the tree relative to the resident's property (Gorman, 2004; Schroeder et al., 2006) or whether they are located in private or public property (Avolio et al., 2015). They have also been linked to variation in resident experiences across spatial scales and social–ecological contexts (Lyytimäki et al., 2008; Escobedo et al., 2011; Shackleton et al., 2016). My results suggest these relationships as well for resident awareness of tree services and disservices. Even when positive attitudes toward trees were not dependent on the proximity of trees, the results supported that the awareness of tree services and disservices did vary when asked about home trees versus neighborhood trees and that awareness of all services and disservices was not uniform across sites. Ecosystem services related to provision (mostly food) were more often acknowledged for home trees (trees in a private space). Ecosystem services related to cultural services, mainly aesthetic or ornamental values, were more often acknowledged for neighborhood trees (trees in private and public spaces). Likewise, reduced structural integrity, which is related to damages to residents' properties, was

more frequently mentioned for home trees, while power line obstruction was more frequently acknowledged for neighborhood trees, particularly at the upper watershed (Cupey). At this site, public and private places tend to be more heavily forested than other sites within the watershed (Ramos-González, 2014). In Cupey, where heat vulnerability indexes are low (Méndez-Lázaro et al., 2018), shade (and not temperature reduction) was more commonly perceived as a home service, while in San Patricio (lower watershed), where heat vulnerability is high and there is more grey infrastructure, temperature reduction was highly perceived as a home tree service (more than a neighborhood service). This suggests that variation in the climatic conditions experienced by residents may influence the awareness of services by residents, which is consistent with other studies (Schroeder et al., 2006). In Los Angeles, residents experiencing hotter climates were more likely to value shade trees than those in cooler ones (Avolio et al., 2015). In Curitiba, Brazil people associated feelings of thermal comfort directly with street trees (Martini et al., 2014). In Hong Kong, residents placed a high value in the heat stress functions of trees, particularly if they anticipated an increasing trend of occurrence of adverse weather events, like rising temperatures (Lo et al., 2017). The combined results of these studies suggest that residents may visualize different types of spaces differently when it comes to the provision of services, and vice versa, making a stronger case for the importance of considering perceptual differences in urban green planning strategies.

5.3. Tree attitudes and yard tree abundance

A previous study considered the potential role of household socio-economic variables but not variation in attitudes and preferences for trees as factors that could influence yard tree abundance (Meléndez-Ackerman et al., 2014). My findings suggest that positive attitudes and expressed preference toward trees might be more influential in tree abundance than expressed negative attitudes. As hypothesized, variation in positive attitudes and expressed preference toward trees (but not expressed negative attitudes) partially explained the current variation in yard tree abundance, along with resident's age, housing tenure, and yard size. This relationship between positive attitudes toward trees and the number of trees in residential yards was also found by Shakeel and Conway (Shakeel & Conway, 2014), but in that study available planting space, a property characteristic, also influenced yard tree abundance. In this study, yard size, a property level variable, was the most important factor associated with tree abundance, which is something that was already stated by the previous study. When adding attitudes and preferences, my current model captured a larger percentage of the variation in yard tree abundance. Nevertheless, more than half of its variation was still unaccounted for in the new model. Other factors that have been suggested as contributing to the presence and composition of yard trees in San Juan in separate studies include plant gifts through social networks (i.e., family members, neighbors), historical process of urban development, homeowner association regulations, length of house tenancy, and natural dispersion (Summit & McPherson, 1998; Cook et al., 2012; Ramos-Santiago et al., 2014; Vila-Ruiz et al., 2014; Torres-Camacho et al., 2017). Studies elsewhere have found that the acknowledgement of tree services and disservices could motivate tree planting and

removals (Summit & McPherson, 1998; Head & Muir, 2005; Kirkpatrick et al., 2012; Conway, 2016; Avolio et al., 2018). In some studies, visual aesthetics, shade provision, wildlife habitat provision, and privacy enhancement are the main reasons for tree planting decisions, while in others, removal decisions were related to tree condition (diseased, advanced age, poor health, dead or dying), maintenance problems, and damages to infrastructure by roots or potential hazards (fallen limbs, danger) (Summit & McPherson, 1998; Head & Muir, 2005; Kirkpatrick et al., 2012; Conway, 2016; Avolio et al., 2018). This study did not evaluate the relationship between the awareness of specific services or disservices and tree management in an explicit way, but the fact that positive attitudes explained (at least in part) yard abundance suggests that evaluating the relationship between awareness and preferences for specific tree ecosystem services and plant abundance could help us understand yard planting decisions in San Juan if they reflect the residents' preferred plant ecosystem services (Avolio et al., 2018). Further research could address tree planting motivations specifically and utilize value research approaches that allocate more space or time for respondents to clearly articulate thoughts about trees, leading to a deeper understanding of their motivations.

5.4. Potential implications for green infrastructure planning

Tree planting is marketed as a go-to solution to increase ecosystem services provision (Dobbs et al., 2017), mitigate and adapt to extreme events (climate change, increasing temperatures, extreme flooding) (Gill et al., 2007; Lafortezza et al., 2017; Orlandini et al., 2017; Sjöman et al., 2018), and biodiversity conservation (Alvey, 2006; Roy et al., 2012; Liveseley et al., 2016). In this study, residents were aware of many

services, but did not always align with most of these goals. For example, benefits related to thermal and air regulation (i.e., shade, temperature reduction, oxygen production, and air purification) were more frequently mentioned relative to other regulating services of importance in the region (natural hazard moderation, flood control, erosion control, carbon sequestration, noise reduction). Residents' awareness of shade provision and temperature regulation services may be an asset in the incorporation of green infrastructure planning prescriptions, but the authors note that other services just as important for climate change adaptation, such as flooding and soil erosion control are not as recognized.

This study suggests that in RPWS, different services may be prioritized differently in different spaces. Planners may incorporate such information to develop green infrastructure plans that consider these differences. For example, food provision and shade services by trees are more often recognized at the household scale than at the neighborhood scale, while air purification and aesthetic services are more often recognized at the neighborhood scales. Urban forestry practitioners could adjust tree management and species selection to local needs (Schroeder et al., 2006; Doody et al., 2010), provide information to residents for species selection and management needs (Summit & McPherson, 1998), and the benefits or cost effectiveness of planting trees (Summit & McPherson, 1998), local nurseries, and tree distribution programs could embrace providing tree species that provide frequently mentioned services, such as fruit and low maintenance trees (Nguyen et al., 2017; Avolio et al., 2018). Urban forestry strategies can be developed to not only maximize ecosystem services of interest, but also to minimize potential disservices (Avolio et al., 2018). In the context of urban vegetation

management, widely acknowledged disservices, such as damage to property (e.g., house, sidewalks, pipes), obstructing power lines, or maintenance hardships, could theoretically be addressed by adequate site and species selection and appropriate management (Lugo et al., 2011; Brandeis et al., 2014). The combined results of this and previous studies suggest that not all services and disservices may be equally important among residents, and the extent that these may be motivators for planting and removal should be considered when developing management strategies at the residential scale.

Positive attitudes toward street trees have been attributed to the perception of services they provide, often expressed by residents despite their awareness of their disservices (Mullaney et al., 2015), and the diminished recognition of tree disservices relative to services is consistent across spatial/geographic scales within this watershed. While it would appear that the majority of RPWS residents see trees as beneficial and not problematic and that disservices do not influence yard tree abundance, I argue that these relationships can be dynamic and need to be monitored. For example, residents seldom mentioned natural hazard moderation as a service and property damage due to natural hazards as a disservice. The surveys presented here, however, were performed before Puerto Rico suffered a devastating hurricane season in 2017, causing severe damage to urban green and grey infrastructure and a significant vegetation loss that has been estimated to be up to 31% in Puerto Rico and the U.S. Virgin Islands combined (Van Beusekom et al., 2018). Experiences after the hurricanes, including the complexities of managing accumulated vegetation debris, the damage to power lines as a contributing factor to the complete collapse of the electric system, the damage of fallen trees and branches to private and public infrastructure, potentially influenced perceptions of tree

disservices that could play a role in attitudes or management decisions (Conway & Yip, 2016). One could hypothesize that the recognition of trees as problematic could increase due to negative experiences, and that perceptions of tree disservices (versus services) may have changed as a result. While ecosystem services frameworks may be useful in understanding urban green infrastructure dynamics, people's attitudes can be influenced by contextual changes and, as such, can be dynamic (Stern & Dietz, 1994; Ordóñez-Barona et al., 2017). Since this study was conducted, the social–ecological system of the island of Puerto Rico has been subjected to important social (e.g., Puerto Rico's debt crisis) and ecological events (e.g., prolonged drought, catastrophic hurricane) that have been accompanied by profound demographic changes and may have, in turn, changed the worldviews and values of island residents.

6. Conclusions

These explorations of the dynamics of urban tropical residential space within the Río Piedras Watershed continue to shed light on the complexities that characterize these systems. I emphasize the role of scale in understanding social-ecological interactions of dynamic urban residential infrastructure, the prevailing land use in urban landscapes. The awareness of services generated by trees is rich but important benefits for climate change adaptation are being overlooked. Understanding this feedback, addressing awareness gaps in ecosystem services where they manifest, mitigating disservices that trees may generate, and further research to identify the underlying values that residents hold toward trees could go a long way in facilitating current and future urban forestry strategies and green infrastructure planning.

Chapter 3. Social motivations and limitations to the cultivation of native plants in urban residential areas of Puerto Rico

1. Abstract

Residential yards spaces of the San Juan Metropolitan area of Puerto Rico, are characterized by complex dynamics and a dominance of non-native ornamental shrubs. However, local urban forestry and conservation initiatives prioritize the use of native trees. In 2013, 423 household surveys were conducted in six locations across the Río Piedras Watershed to evaluate resident's values, attitudes and behavioral intentions regarding native plants. The specific goals were to (1) evaluate attitudes toward native plants and their association with socio-demographic factors and value orientations, (2) evaluate residents' willingness to exchange non-native plants for native ones, (3) identify preferred growth stage for gifted plants, and (4) explore the level of importance of different ecosystem services to residents. Most residents self-reported a strong positive attitude toward native plants (73.4%) which was mainly driven by *sense of place* (57.2%). *Ecosystem services* responses were variable across the level of agreement towards preference for native plants. Only 1.7% of respondents reported strong negative attitudes toward native pants, that were mainly driven by *biospheric values*. Although residents recognize common conservation arguments (e.g., invasiveness avoidance, native plant adaptability), these were not significantly associated to positive attitudes toward native plants. The majority of residents (65.2%) also expressed a willingness to change their yard composition by replacing a non-native plant with a native one and the intention was associated with a preference for small food trees and ornamental shrubs. Air purification and food provision were recognized as the most important *ecosystem*

services and wildlife habitat provision the least important. Results emphasize the need to consider these prevailing attitudes and value orientations in guiding effective conservation and forestry strategies in urban areas.

2. Introduction

The promotion of urban biodiversity conservation and the optimization of ecosystem services to improve well-being are becoming important goals of urban green infrastructure and urban forestry management (Ordóñez-Barona & Duinker, 2013; Almas & Conway, 2016, 2018; Nilon et al., 2017). Prioritizing native species over non-natives species is often encouraged by urban reforestation programs around the world (Almas & Conway, 2016). On the other hand, while urban environments often have species-rich vegetation communities, they are frequently dominated by non-native species (Brandeis et al. 2014; McKinney, 2008; Smith et al., 2006). This may point to discrepancies between goals of local forestry strategies and conservation initiatives. It has been argued that even when using urban areas as a venue for biodiversity conservation is important, its proponents often fail to effectively articulate the motivations for species conservation and they need to improve how particular conservation goals are decided in urban areas (Dearborn & Kark, 2010). Indeed, many researchers highlight the usefulness of incorporating social-ecological system approaches to define urban forestry strategies and the need to integrate local socio-cultural and ecological values (Moffatt & Kohler, 2008; Ordóñez-Barona & Duinker, 2012; Steenberg, Duinker, et al., 2019; Conway et al., 2019). The hypothesis is that these approaches may help reconcile species conservation goals with those of urban forest management.

Urban forests are a distinctive natural feature of green infrastructure (Ferrini et al., 2017) and for the purpose of this study are defined as all public and private trees and associated vegetation located in a city (Konijnendijk van den Bosch et al., 2006; Escobedo et al., 2011; Roy et al., 2012; Baur et al., 2016; Ferrini et al., 2017; Lafortezza et al., 2017; Roman et al., 2018). Urban forestry management activities provide opportunities to understand, design and manage urban forests structure (species composition, diversity, health, age classes) and functions (ecological and societal values) at local scales that in turn can be incorporated in green infrastructure planning (Ordóñez-Barona & Duinker, 2010; Lafortezza et al., 2017; Steenberg et al., 2017). At the same time, urban forestry has evolved beyond focusing on biophysical components to include people, in part because of the growing recognition of their influence in urban forests structure and function (Steenberg, Duinker, et al., 2019). Social-ecological theory has been applied as a guiding approach to evaluate urban forests in general due to the complexity and dynamism of the interactions that occur between humans with the different elements of the urban landscape (Johnson et al., 2019; Steenberg, Millward, et al., 2019). This approach has been particularly useful in evaluating the drivers of diversity of green spaces in urban residential landscapes (Cook et al., 2012; Chowdhury et al., 2016) particularly when applied to households were spaces are privately owned, managed and partially dependent on resident's actions (Meléndez-Ackerman, Nytch, et al., 2016).

Residential private yards contribute to a significant amount of green infrastructure in cities (Cook et al., 2012; Brandeis et al., 2014) and often contain the majority of trees of the urban forest (Konijnendijk van den Bosch et al., 2006; Nowak & Greenfield, 2012;

Almas, 2017). Consequently, urban forestry practices have shifted from a traditional focus in public spaces by "branching out" to private residential spaces in order to meet strategic goals (Miller et al., 2015; Nguyen et al., 2017). Urban residential yards are now also recognized by their potential to substantially contribute to biodiversity protection, ecosystem services provision, and human well-being (Lubbe et al., 2010; Goddard et al., 2010; Cook et al., 2012; Freeman et al., 2012; Garcia-Montiel et al., 2014; Vila-Ruiz et al., 2014; Camps-Calvet et al., 2016). With regards to biodiversity conservation, it has been suggested that yards could offset the threats of urbanization to native biodiversity when managed in consideration of native plant-wildlife interactions (Lerman & Warren, 2011) and their role at improving connectivity for species that have limited habitat (Doody et al., 2010). Nevertheless, while often rich in plant species, these spaces tend to have high percentages of non-native species (Acar et al., 2007; Akinnifesi et al., 2010; Bigirimana et al., 2012; Van Heezik et al., 2014; Vila-Ruiz et al., 2014; Meléndez-Ackerman, Nytch, et al., 2016) emphasizing a lack of consistency between current and past urban vegetation management with conservation initiatives that promote the use of native species.

The structure, composition and function of urban residential yards is determined by complex interactions between humans and the environment occurring at different temporal and spatial scales (Cook et al., 2012; Müller et al., 2013; Vila-Ruiz et al., 2014; Padullés Cubino et al., 2019). At the household scale, property characteristics (e.g., housing age), personal characteristics (e.g., income, age) and cognitive components (e.g., values, attitudes) influence yard management decisions and landscaping practices (Peterson et al., 2012; Dearborn & Kark, 2010; Cook et al., 2012; Conway, 2016; Almas

& Conway, 2017). As a result, residential yards are reflection of valued-based priorities, personal identity, as well as personal and community held utilitarian and non-utilitarian values. In that regard, important contributions from environmental psychology on the role of psychological drivers of behavior and behavioral intentions (Stern & Dietz, 1994; Guagnano et al., 1995; Schultz & Lynnette, 1999; Stern et al., 1999; Milfont & Duckitt, 2010; Wolf, 2017) can provide useful frameworks for understanding how environmental values may limit or facilitate environmental behaviors.

The study of human-environmental interactions is an increasingly common discourse in urban forest management to help guide practices that meet the needs of people and that increase civil involvement (Krajter Ostoić & Konijnendijk van den Bosch, 2015). The Cognitive Hierarchy Theory (Homer & Kahle, 1988) is a commonly applied framework that helps to contextualize human-environment relationships, where individual views of the environment can be organized in a cognitive hierarchy where each element builds on each other (Vaske & Donnelly, 1999). This theory emphasizes the hierarchical and causal nature between individual values (more abstract and resistant to change), collective value orientations, specific attitudes toward objects or actions, behavioral intention and actual human behavior (more specific and variable) (Whittaker et al., 2006; Baur et al., 2016; Jacobs et al., 2019). The key concepts are that values are more strongly held and less variable than attitudes, that some values are commonly held by groups (i.e., value orientations, beliefs patterns), and that attitudes can work as a mediator between people' values and their behavior (Homer & Kahle, 1988; Ives & Kendal, 2014; Ordóñez-Barona et al., 2017). Values are judgements of what individuals consider important (Rokeach, 1973; Ordóñez-Barona et al., 2017) while attitudes refer to

positive or negative evaluations that people hold towards a specific object, place or issue (Lichtenstein & Slovic, 2006; Milfont et al., 2010; Ives & Kendal, 2014; Maio et al., 2019). Value orientations (i.e., the patterns among basic beliefs) help us link stable but abstract values (referring to a specific belief) with more specific cognitions (attitudes, behavioral intentions and behaviors) (Fulton et al., 1996; Vaske & Donnelly, 1999; Manfredo & Dayer, 2004; Whittaker et al., 2006). Consideration of public values and attitudes is increasingly viewed as important in the development of effective urban forest management strategies and goals that are socially accepted, have public support, and more likely to succeed (Ives & Kendal, 2014; Baur et al., 2016; Larson et al., 2016; Jones et al., 2016; Wolf, 2017). Information on stakeholders' social attitudes and motivations toward the native flora at a residential scale, can aid in the clarification of these seemly conflicting motivations toward native versus non-native vegetation and help capture a range of local value associations.

Few studies have evaluated the household drivers related of native plants in residential spaces or whether residents support common municipal goals related to native plants (Almas & Conway, 2018). In New Zealand, a study of residential yard structure, found that presence of native plants can be influenced by previous owners gardening practices (Van Heezik et al., 2014). In China, native plant species richness in urban residential yards (as well as total, herb and tree richness) was found to be positively related to distance to the Beijing urban center (Wang et al., 2015). The lack of native plant availability in local retail nurseries has also been suggested to influence yard composition in the United Sates (Avolio et al., 2018) and Puerto Rico (Torres-Camacho et al., 2017) were inventories showed a high prevalence of non-native species. In relation

to personal social drivers, studies have showed conservation attitudes and preferences for native plants to be related to their presence in residential gardens (Head & Muir, 2006; Kurz & Baudains, 2012; Kendal et al., 2012) and specific planting decisions (Kirkpatrick et al., 2012). Attitudes that people have toward native species can be shaped by species-specific attributes (Selge et al., 2011; Kendal et al., 2012; Jenerette et al., 2016) or by the benefits that species provide (ecosystem services) (Belaire et al., 2015). Humans may also hold cultural value judgements associated with native species, with individual motivations favoring native species over "alien" ones (Gröning & Wolschke-Bulmahn, 2003; Lafortezza et al., 2009). For example, people can associate their own national identity with a species "nativeness" (Selge et al., 2011) and native species can promote cultural identity and provide a sense of place (Moro et al., 2014) and thus provide important cultural service (Hausmann et al., 2016). Understanding residents values in relation to native species is therefore important in defining goals for urban biodiversity conservation (Dearborn & Kark, 2010) and urban forest management (Ordóñez-Barona & Duinker, 2012). Furthermore, identifying value orientations can help understand and predict attitudes and behavior toward particular environmental issues (Fulton et al., 1996; Whittaker et al., 2006). Environmental values can influence attitudes toward urban forests and how they are managed (Baur et al., 2019) and could result in support or opposition for specific tree management strategies and policies (Wolf, 2017). Overall, people hold different values and there are multiple pathways that could drive one particular behavior (Ives & Kendal, 2014). Therefore, by focusing on one particular attributed value of the urban forest (e.g., the provision of habitat for wildlife for native species conservation), we could be very well be overlooking a range of values that influence attitudes and

behaviors when trying to achieve favorable outcomes. Ultimately, urban management initiatives with a goal to increase native plant populations in urban environments may be better served from knowledge of how local value orientations, preferences and attitudes may limit or facilitate such initiatives before implementation which can be complemented by ecological knowledge on whether ecological processes favor or not the intentions to move towards specific groups of species.

This study is part of a larger interdisciplinary research effort designed to explore top-down and bottom-up factors that may facilitate or limit the cultivation of native plants in yards of the Río Piedras Watershed (RPWS) in San Juan, Puerto Rico, using a social-ecological systems approach (Meléndez-Ackerman, Olivero-Lora, et al., 2016; Torres-Camacho et al., 2017). Although the goals of local conservation initiatives (e.g., tree planting initiatives and local tree giveaways) and urban forestry plans prioritize the use of native species (Torres-Camacho et al., 2017), RPWS yards host a high percentage of non-native species dominated by ornamental shrubs (Vila-Ruiz et al., 2014). The study particularly explores household level social-psychological factors (residents' values, attitudes and behavioral intentions) regarding native plants, and how these might be influenced by social and demographic variables at the household level. Here, I applied Cognitive Hierarchy Theory to help contextualize the human cognitive drivers that may influence residential yards landscape practices at the household scale. The specific goals were to (1) evaluate attitudes toward native plants and whether socio-demographic factors and value orientations might influence these attitudes, (2) evaluate if willingness to exchange non-native plants for natives was associated with socio-demographic characteristics and preferred plant habits or ecosystem services, (3) identify if residents

had a preferred plant propagation stage when gifted plants, and (4) explore the level of importance of wildlife habitat provision relative to other services of importance to residents. I hypothesized that positive attitudes toward native plants would be associated with socio-demographic characteristics and specific value orientations. I also predicted that residents may express vegetation preferences based on plant origin (i.e., native or non-native), structural traits (e.g., plant habit, growth stage, size) and their ecosystem services (e.g., ornamental, food). Results are followed with a discussion on how to integrate the information generated to develop urban forest management strategies that are more effective at integrating conservation goals, the gaps in understanding of socio-psychological drivers related to the planting of native plants, and how this information contributes to the integrated comprehension of the Río Piedras Watershed social-ecological dynamics.

3. Methods

3.1 Study site

This research was conducted in the Río Piedras Watershed (RPWS) located in the San Juan Metropolitan Area (SJMA), which is the largest and most populated urban area on the island of Puerto Rico. The area lies within what is classified as subtropical moist forest zone Based on Holdridge's life zone system (Ewel & Whitmore, 1973) with mean annual precipitation that ranges from 1509 mm to1755 mm and a mean annual temperature of 25.7 ºC (Lugo et al., 2011). The Río Piedras River itself has been subject to severe anthropogenic pressures and modifications (Lugo et al., 2011) and further channelization is projected for upcoming years by the U.S. Army Corps of Engineers

(U.S. Army Corps of Engineers, 2019). The RPWS covers about 67,000 m^2 of which 42% had been categorized as green cover in 2014, with residential yards adding a high fraction of the city overall green infrastructure (Ramos-González, 2014). Vegetation in San Juan consists of novel combined assemblages of native and non-native species as a result of natural and anthropogenic processes (Lugo & Helmer, 2004a; Brandeis et al., 2009; Muñoz-Erickson et al., 2014). The RPWS is within the San Juan Bay Estuary (SJBE) boundaries, were residential areas are the dominant land use with 36% cover (Brandeis et al., 2014). The latest inventory found 69% of woody species in RPWS residential yards to be non-natives with ornamental shrubs being the most abundant (10 species all non-native), trees being the most common food plants (7 species, only one native) with the exception of the food producing herbs *Musa acuminata* and *Musa x paradisiaca*, and the palm *Cocos nucifera* (Vila-Ruiz et al., 2014). Some of the drivers that have been found to partially explain variation in yard vegetation composition in the Río Piedras Watershed (RPWS) are resident's age, housing tenure, total yard area, and watershed location (Meléndez-Ackerman et al., 2014), prevailing uses (ecosystem services) such as food and ornamental plants (Vila-Ruiz et al., 2014), plant gifts, natural dispersion, and historical plantings (Torres-Camacho et al., 2017). Also, self-reported positive attitudes toward home trees have been found to partially explain a higher abundance of trees in yards (Chapter 2, Olivero-Lora et al., 2019).

3.1.1 Local conservation and urban forestry initiatives

There was not an official forestry plan in place specific for the municipality of San Juan at the time this study was conducted. However, Puerto Rico Department of Natural and Environmental Resources (DNER) held from 2008-2012 the *Puerto Rico*

Verde (‘Green Puerto Rico’) campaign with the goal of planting a million of four to six feet tall native trees in urban and coastal areas. The Puerto Rico Forest Action Plan of 2016 (Department of Natural and Environmental Resources, 2016) currently promotes native species (propagation and use) to manage the threat of invasive plants and as a conservation strategy to increase services provided by urban forests. Also, the Urban and Community Forestry Program (UCF), funded by the U.S. Department of Agriculture’s Forest Service (Federal) and managed by Department of Natural and Environmental Resources of Puerto Rico (State), sponsors different interventions and management alternatives for public and private green infrastructure, also embraces initiatives favoring native plants. In the SJMA, several local initiatives work independently to promote the use of native trees. A well-known initiative in this island comes from a local non-governmental organization, Para La Naturaleza*,* that advocates native and endemic species reforestation through the program *Árboles más Árboles* (‘Trees, more Trees’) focused on producing, distributing and planting. Their five nurseries produce over 60,000 trees annually available for sale and the organization holds a public annual fair in San Juan that gives over 4,000 native trees to assistants at no cost. The San Juan Bay Estuary Program supported by the U.S. Environmental Protection Agency and associated with the National Estuary Program (NEP) also sponsors native tree plantings on managed green spaces (e.g., sidewalks) within the San Juan Bay Estuary covering five cities, including San Juan. In addition, a private initiative by a local bank under their program *Uno con el Ambiente* (‘one with the environment’) has given away thousands of fruit and native trees each year (over 10,000 over the last 9 years) at their 14 branches across the island, five of which are located in the San Juan Municipality. Indeed, this initiative, has increased the

number of fruit trees offered due to the increasing demand of participants. Planting and tree giveaway campaigns have resurfaced after the 2017 devastating hurricane season when an estimated 23-31 million trees were lost (Feng et al., 2018) in the island and is estimated the San Juan Municipality lost up to 24.8% of tree cover (Meléndez-Ackerman et al., 2018). The DNER created the *Sembrando Futuro* ('Planting a Future') program with a goal of planting 500,000 trees across the island in five years, species selection is determined by professionals according to site. Para La Naturaleza established the *Hábitat* program, with a goal of planting 750,000 native trees across the island of which nine native species are the most planted. To reach this ambitious goal, they have created a parallel program *Ciudadanos Botánicos* ('Botanical Citizens') to train citizens in species identification, seed collection, education, volunteer recruiting and data entry. To complement, eight new nurseries have been created at public and private schools to be used as food gardens for sales and consumption.

3.2 Study design

Study design followed the original research design developed for the San Juan Urban Long-Term Research Area (ULTRA) Collaborative Network where variety of social-ecological questions at residential scales have been studied (Garcia-Montiel et al., 2014; Meléndez-Ackerman et al., 2014; Ramos-Santiago et al., 2014; Vila-Ruiz et al., 2014; Meléndez-Ackerman, Nytch, et al., 2016; Torres-Camacho et al., 2017; Olivero-Lora et al., 2019). Research was conducted using a stratified sampling design at the six ULTRA social-ecological monitoring areas. Sites were randomly selected and are located across a rural-urban and elevation grey-green coverage gradient (Figure 1 of Chapter 2).

At each site a circular buffer zone of 1 km radius was overlaid on aerial photos of San Juan and a street vector file of access roads using ArcGIS software (v. 9.3). Using Microsoft Excel, a random sample of roads codes where selected and each road was visited. The team visited each road were 423 total surveys were administered via convenient-based recruitment (depending on willingness to participate and availability of residents) from January to October 2013 ranging from 68 to 79 per site as follows: San Patricio (N=69), Puerto Nuevo (N=68), Avenida Central (N=72), La Sierra (N=73), Chiclana (N=69), and Cupey (N=72). The majority of the surveys were carried out during weekends from 9:00 am to 5:00 pm. Surveys were conducted by students and faculty members of the University of Puerto Rico Río Piedras Campus after successfully completing online training course "Protecting Human Research Participants" from the National Institute of Health (NIH). The design and implementation of all human research protocols followed the specifications of the university of Puerto Rico Institutional Review Board (IRB #1011-013) including providing each participant a consent form which contained contact information of principal researchers and the local IRB office, included a privacy and confidentiality clause and explained the voluntary nature of the survey.

3.3 Household social surveys

The surveys were elaborated by an interdisciplinary team (faculty professors, graduate, and undergraduate students) thru open discussion (Meléndez-Ackerman, Olivero-Lora, et al., 2016) and consisted of a combination of closed and open-ended questions. The aim of this instrument was to identify socio-psychological factors that

may be related to the cultivation of native plant species on residential yards (see survey questions used in this study Appendix B, Table B1) as part of a broader interdisciplinary project called *Agents of change* (Meléndez-Ackerman, Olivero-Lora, et al., 2016). The questionnaire was pre-tested to ensure appropriate question format, wording, duration and order. Attitudes were addressed using a Likert-type question and participants were asked if they though preference should be given to Puerto Rican plants over plant from other places, this was followed by an open-ended question on the reasons why to determine the values associated with self-reported attitudes. To explore antecedent of behavioral intention, we included a question assessing participant's willingness to exchange non-native plants for native plants, and for those who responded affirmatively, we followed up asking what kind of plant would they accept based on their (1) habit (small tree, shrub, small herb, big tree, palm or tree fern, more than one, anything) and their (2) use (ornamental, food, medicine, shade, other). To help inform local initiatives, we also asked the preferred propagation method (seed, cutting, young plant, small plant, adult plant, more than one option) if gifted a plant at that particular moment. Finally, to compare the importance of wildlife habitat provision (a common native species conservations argument) in relation to other services documented as important to local residents (Olivero-Lora et al., 2019), surveyed residents were asked to rank plant benefits (aesthetics/ornamental, food, shade, air purification, habitat/space for wildlife) from the highest to lowest rank of importance. For each respondent, the survey also gathered the following individual or household-level demographic and socio-economic information: the respondent's age, gender, marital status, years of formal education, annual average income household size and household tenure.

3.4 Analyses

In order to carry out analyses, only variables that did not have any missing values were considered, resulting in a smaller sample for each model. Household socio-economic variables were either qualitative (gender, household tenure, civil status) or quantitative (age, years of formal education, annual family income, household size). Qualitative variables were coded as binary variables as follows: gender (female = 1, male = 0), household tenure (owner = 1, renter = 0), civil status (married or living with partner = 1, single or divorced = 0). Only one respondent did not attend school whatsoever and the case was collapsed with elementary school responses. Descriptive statistics were generated for all socio-demographic household characteristics. The remaining variables generated from the surveys and their specific analyses are described below. All the statistical analyses were run using SPSS v.25 (IBM Corp. Released, 2017).

3.4.1 Drivers of preferences (attitudes) toward native plants

To assess attitudes, residents were asked if they thought that priority should be given to the cultivation of native plants over non-native ones, with the question framed as follows: "*Should preference be given to plants from Puerto Rico over plants from other places?*". This was a closed-ended Likert-type question and responses were given on a 5-point level of agreement range from *strongly disagree* (coded = 1) to *strongly agree* (coded =5). This was called *stated preference*, and the majority of responded expressed strong agreement. Therefore, a new binary variable called *strong preference* was created by dividing responses into those that strongly agreed (code = 1) and those who did not (code = 0). To evaluate if *stated preference* responses were consistent across *watershed locations,* Fisher's exact test based on Monte Carlo method (10000 permutations) was

performed. We ran binomial logistic regression to evaluate if *age*, *gender*, *income*, *education*, *marital status*, *household size*, *housing tenure* and *watershed location* were associated to the likelihood of residents having a strong positive attitude (*strong preference*) toward native plants. Assumption of linearity of continuous variables (age, income, years of formal education, household size) were assed via the Box-Tidwell (1962) procedure (all continuous variables $p > 0.23$) with Bonferroni correction statistical significance of $p < 0.00625$.

To assess related value orientations, follow-up explanative responses to the open-ended question about priority to native plants were transcribed and translated into English. Thematic analysis was conducted following suggested procedure (Maguire, M., & Delahunt, 2017; Nowell et al., 2017) with the inductive coding approach and definition of common value themes based on frameworks and definitions drawn from published work from the literature. Coding and theme extraction followed similar approaches used to define the value systems of people in relation to urban forests taking into account the frequency a value was mentioned to create sub-themes and categories (Peckham et al., 2013; Sinclair et al., 2014; Ordóñez-Barona & Duinker, 2014a; Ordóñez-Barona et al., 2016, 2017). Themes were collapsed into five well-defined categories or *value orientations* (see Results, Table 7) with 10 responses classified as "other" because these were not specific enough. Fisher's exact test based on the Monte Carlo method was carried out to test for differences in the distribution of *value orientations* across *watershed location*. We also, ran a Fisher's exact test to evaluate the association between the different explanatory categories of *value orientations* with the level of agreement *stated preference* toward natives. A Chi-square test of independence between the stated

strong preference categories (strongly agree versus non-strongly agree) and *value orientation*s categories was also performed considering standardized residuals (z-scores) greater than two for post-hoc analyses of cell comparisons (Agresti, 2019). For the three tests the category "other" was excluded from the analyses. A binomial logistic regression was performed for each of the three dominant *value orientations* (i.e., *sense of place*, *ecosystem services*, *conservation*) to test if certain demographic groups had a higher probability of stating each of them.

3.4.2 Willingness to trade non-native plants for natives

Residents responses on their willingness to trade a non-native plant for a native one were treated as a binary response (yes = 1, no = 0). A Chi-square tests of homogeneity (2 x 6) was performed to evaluate whether the proportions (or binomial distributions) of positive and negative responses were similar across the six *watershed locations*. This was followed by a binomial logistic regression to test if the probability of expressing *willingness to exchange* non-native plants for native ones was associated with household-level socio-demographic groups. Box-Tidwell (1962) procedure and Bonferroni correction was applied to test assumption of linearity for all four continuous variables.

3.4.3 Role of preferred plant habit and ecosystem services in willingness to trade

Those residents that were willing to trade a non-native with a native plant were asked to indicate their preferred *plant habit* (small herb, shrub, tree fern and palm, big tree, more than one, anything) and preferred *ecosystem service* (food, medicine, ornamental, shade and other) in the plant exchange. Due to low number of responses, "shade" and "other" were collapsed into a new category "shade or other". Chi-square

tests of homogeneity were used to test for differences among *watershed locations* in the frequency distribution of preferred *plant habit* or *ecosystem services* in the plant exchange. Due to low expected counts, Fisher exact tests (Monte Carlo procedure) was performed to test differences between preferred *plant habit* differed by *watershed location* as well as for preferred *ecosystem services* by *watershed location*. We chose the most common categories for preferred *plant habit* (small tree, shrub, big tree) and *preferred services* (food, medicine, ornamental) and ran a Chi-square test of independence to evaluate the association between these variables. Standardized residuals greater than two (z-scores) were considered to make cell comparisons.

3.4.4 Preferred propagation method for plant gifts

Residents were asked about how they would prefer a plant gift in terms of six pre-defined *propagation methods*: "seed", "cutting", "small plant", "young plant", "adult plant" and "more than one". A Chi-square test of independence was conducted to evaluate if the frequency distribution of *propagation method* categories was associated with *watershed location*. This test was run excluding the "cutting" category due to a low frequency of responses at each cell. Standardized residuals greater than 2 were considered to make post-hoc cell comparisons and to determine the strongest evidence against the null hypothesis of no association between preferred *propagation method* categories for their plant gift and *watershed location*.

3.4.5 Ranking of importance of ecosystem services

We asked residents to rank five *ecosystem services* (ornamental, food, shade, air purification, wildlife habitat) that are known to be of significance to RPWS residents based on empirical data (Olivero-Lora et al., 2019). We converted each service into a

ranked variable and carried a non-parametric Friedman test to analyze differences among *ecosystem services* in their ranking scores as stated by respondents. As the null hypothesis was rejected (see Results), post hoc pairwise comparisons were carried out to evaluate the differences in the rankings between pairs of services. A Kruskal-Wallis test was also conducted to determine if the ranking score of each *ecosystem service* differed among *watershed location* followed with Dunn-Bonferroni post hoc method analysis to evaluate contrasts among *watershed locations*.

4. Results

4.1 Household socio-demographic characteristics

The majority of respondents were females (55.6%) (Table 5). Out of 417 respondents, 58.8% were married or living with a partner as opposed to single or divorced. A significant number of residents were single family homeowners (81.3%) instead of renters or having other types of living arrangements. Residents were 60 years of age on average with an average of 14.7 years of formal education. Households had an average annual family income below but close to $33,000 and the average number of residents per household was less than three occupants.

Table 5. Descriptive statistics for the seven social-economic characteristics of 423 RPWS households.

A	Variable	Class	Frequency
1	Gender	Female	233
	(n=419)	Male	186
2	Civil status	Single or divorced	172
	(n=417)	Married or living with partner	245
3	Ownership	Owned	340
	(n=418)	Rented or other	78
B	**Variable**	**Descriptive Statistics**	**Value**
4	Age (yrs.)	Mean ± SE	59.91 ± 0.9
	(n=410)	Max	97
		Min	18
5	Household income	Mean ± SE	$32,553 ± $1,414
	(n=327)	Max	$80,000
		Min	$5,000
6	Years of formal education	Mean ± SE	14.7 ± 0.2
	(n=412)	Max	22
		Min	6
7	Household size	Mean ± SE	2.8 ± 0.7
	(persons per household)	Max	9
	(n=418)	Min	1

4.2 Resident's stated preferences (attitudes) toward native plants

When asked about whether preference should be given to plants from Puerto Rico over plants from other places the majority of the 421 respondents (85.4%) agreed with the statement with 73.4% responding that they *strongly agreed* and 15.0% responding that they *somewhat agreed*. A small minority *did not agree nor disagree* (7.4%), *somewhat disagreed* (2.6%) or else *strongly disagreed* (1.7%). The distribution of responses to these questions was statistically similar across watershed locations (Figure 5).

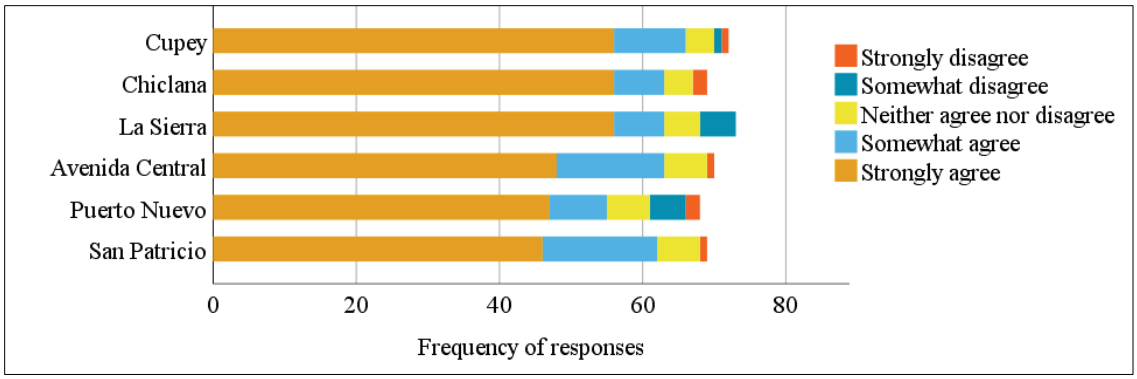

Figure 5. Frequency of responses among watershed locations as to whether preference should be given to Puerto Rican plants over plants from other places (Fisher's exact test, $X^2 = 26.211$, $p = 0.078$).

A binomial logistic regression model showed that some socio-demographic factors were statistically associated with the likelihood of strongly agreeing with regards to the question about preference for native plants (Table 6). Specifically, older residents and residents from larger households were more likely to *strongly agree* with giving a preferential treatment to native plants. Nonetheless, this model explained only 14.1% (Nagelkerke R^2) of the variation in responses, showed a poor fit (Hosmer and Lemeshow Test: $p = 0.04$) and the area under the receiver operating characteristic (ROC) curve was 0.701 (95% CI, 0.634 to 0.768) indicating a poor to moderate level of discrimination by the model.

Table 6. Logistic regression predicting likelihood of strong stated preference (coded = 1) for native plants based on socio-demographics. Reference categories for qualitative variables were defined as follows: gender (males compared to females), civil status (single or divorced compared to married or living with partner), ownership (renter or other compared to owners), and site (each site compared with Cupey).

Variable	B	SE	Wald	*df*	*p*	Odds Ratio	95% CI for Odds Ratio	
							Lower	Upper
gender (1)	0.003	0.29	0	1	0.991	1.00	0.56	1.78
age	0.003	0.01	9.86	1	**0.002**	1.03	1.01	1.05
civil status (1)	-0.012	0.31	0.00	1	0.969	0.99	0.54	1.81
years of formal education	-0.101	0.06	3.23	1	0.072	0.90	0.81	1.01
annual average income	0	0	0.22	1	0.64	1	1	1
ownership (1)	0.269	0.41	0.43	1	0.51	1.31	0.59	2.92
household size	0.332	0.13	6.27	1	**0.012**	1.39	1.07	1.81
site			10.04	5	0.074			
(1) San Patricio	-0.501	0.49	1.06	1	0.304	0.61	0.23	1.57
(2) Puerto Nuevo	-0.695	0.45	2.37	1	0.124	0.50	0.21	1.21
(3) Avenida Central	0.337	0.52	0.41	1	0.521	1.40	0.50	3.92
(4) La Sierra	0.549	0.53	1.09	1	0.297	1.73	0.62	4.862
(5) Chiclana	0.139	0.54	0.07	1	0.797	1.15	0.40	3.318
constant	0.16	1.20	0.02	1	0.894	1.17		

4.2.1 Value orientations related to stated preferences (attitudes) toward native plants

There were five *value orientations* related to *stated preference* that emerged from the overall explanatory responses. These included *sense of place*, *ecosystem services*, *biospheric values*, *local adaptation* and *conservation* (Table 7). Ten responses (2.4% of total responses) were classified as *other* as they did not clearly follow any of the emerging concepts. *Sense of place* represented the majority (47.5%) of all responses, followed by *ecosystem services* (25.2%), *conservation* (12.6%), *local adaptation* (6.5%) and *biospheric values* (5.8%). Out of the 104 responses related to *ecosystem services*, 14.0% referred to *provision services*, 1.5% to *cultural services*, 1.0% to *supporting services* and only one (0.2%) related to a *regulating service*. Several respondents (8.5%) did not specify a particular type of *ecosystem service*. There were no significant

differences in the relative distribution of responses related to the five emerging *value orientations* across *watershed location* ($X^2 = 15.378$, $p = 0.750$).

Table 7. Overarching categories, examples of prevailing themes and verbatim responses of value orientations that emerged from coded responses as to why preference should be given (or not) to plants from Puerto Rico over plants from other places.

Categories	Prevailing themes	Examples of verbatim responses
Sense of place	place attachment place identity Place dependence	"Natives are more beautiful" "Because it is traditional" "Because we are native" "Because it is our patrimony" "Because my country is first" "What is ours goes first than what is foreign"
Ecosystem services	ecosystem services cultural services regulating services provision services supporting services	"Other plants are also beneficial" "Preference should be given to what is beneficial" "Use our flowers" "Because we need to plant to eat. Everything comes from outside" "In a crisis they are beneficial, accessible" "They give us oxygen" "Because the land is productive"
Biospheric values	equality equal importance equal rights equal priority to tall	"Everything has its importance" "Preference needs to be given to all plants" "Everything needs to be treated the same" "Natives have a right"
Local Adaptation	climate adaptation better performance less management	"They are more resistant because they are from here" "Exotic plants are brought, and they adapt well to our climate"
Conservation	avoid extinction species conservation avoid invasives avoid ecosystem disruption species coexistence species competition	"That way we avoid their disappearance" "They are becoming extinct" "Non-native trees compete with ours" "Exotic plants affect the natural balance" "Both types of plants can coexist"

There was a significant difference among of *value orientation* categories in the frequency distribution of *stated preference* level categories (Figure 6). Those who strongly agreed (*strong preference*) with native plants cited more frequently justifications related to the provision of a *sense of place*, 57.2% compared to 22.0% who cited *ecosystem services.* The proportion of respondents that expressed *biospheric value* as a

reason for a *strong preference* was significantly lower than any other category (1.3% of strongly agree responses), conversely, *biospheric value* accounted for the majority of responses associated with a *strong disagreement* (66.7%). Also, a positive significant association was found for "neutral" responses (neither agree nor disagree) and *biospheric values* (40%) and a negative one was found for "neutral" responses and s*ense of place* (8.0%).

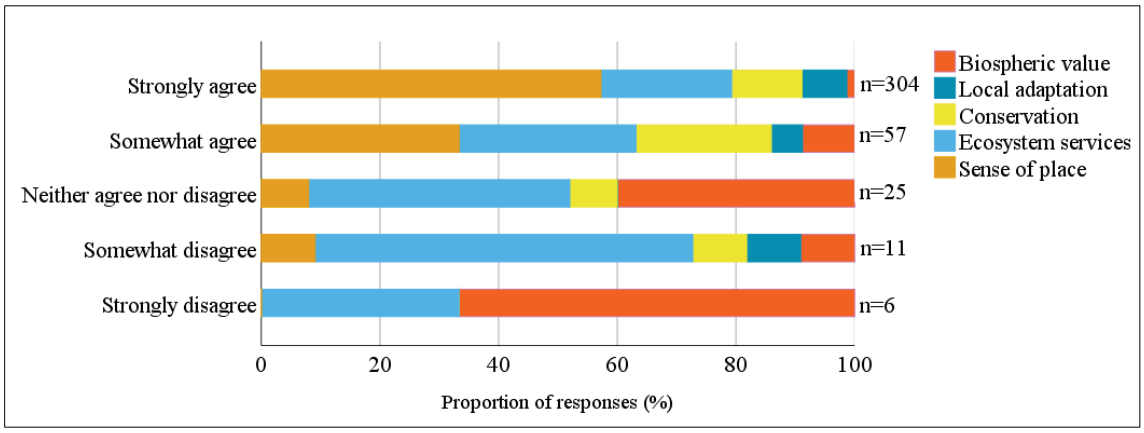

Figure 6. Proportion of responses of each level of agreement with stated preference for natives by value orientation (Fisher's exact test: $X^2 = 96.51$, $p < 0.001$, N = 403).

Chi-square test of independence yielded a strong (Cramer's V = 0.45) positive association between *strong preference* (strongly agreed =1, did not strongly agree = 0) and *sense of place* (z = 6.1) and a negative association with *biospheric values* and *ecosystem services* (z = -6.9 and z = -3.0 respectively; $X^2(12) = 72.824$, $p < 0.001$, N = 403. The likelihood of stating *sense of place*, e*cosystem services,* or *conservation* as a reason to strongly prefer native over non-native plants was not significantly related to socio-demographic variables (site, gender, civil status, household tenure, age, income, education and household size) at the household scale (binomial logistic regression: s*ense*

of place $X^2(12) = 14.808$, $p = 0.252$, N = 307; *ecosystem services* $X^2(12) = 15.075$, $p = 0.237$, N = 306, *conservation* $X^2(12) = 15.694$, $p = 0.206$, N = 306).

4.3 Willingness to trade non-native plants for natives

A majority of respondents (65.2%) expressed a *willingness to exchange* non-native plants with native ones at the time of the study and the frequency of positive and negative responses were not significantly different across *watershed locations* ($X^2(5) = 5.685$, $p = 0.338$, N = 417). The likelihood of expressing *willingness to exchange* non-native plants with native ones was significantly influenced by the residents' *age* using a binomial logistic regression model ($X^2(12) = 37.015$, $p < 0.001$, N = 309). The odds of a person being willing to exchange a native for a non-native decreased 0.974 times per year of age increase (B = -0.030, $p = 0.001$). This model explained 16% (Nagelkerke R^2) of the variance in *willingness to exchange* and area under ROC curve was 0.711 (95% CI, 0.649 to 0.773) ranging from a poor to moderate level of discrimination. None of the other socio-economic predictor variables included in the model was statistically significant.

4.3.1 Role of plant habit and ecosystem services in willingness to exchange

When willingness was expressed, not all *plant habits* or *ecosystem services* were equally preferred by residents. Global responses on what *plant habits* were preferred were distributed as follows: *small tree* (32.8%), *shrub* (25.1%), *more than one* (16.6%), *small herb* (8.5%), *big tree* (7.3%), *palm or tree fern* (6.2%), anything (3.5%). Reponses on what *ecosystem services* were preferred were distributed as follows: *food* (38.6%), *ornamental* (34.8%), *medicine* (16.1%), *other* (10.5%) and *shade* (4.8%). Fisher exact

tests yielded significant differences in the frequency distribution of preferred *plant habit* and *ecosystem services* categories across *watershed location* (Figure 7). For some sites (Puerto Nuevo, Chiclana and Cupey), trees were the most preferred option relative to shrubs, but not in others (San Patricio and La Sierra). In one site (Avenida Central), the most frequent response alluded to preferring more than one option, followed by small trees and shrubs.

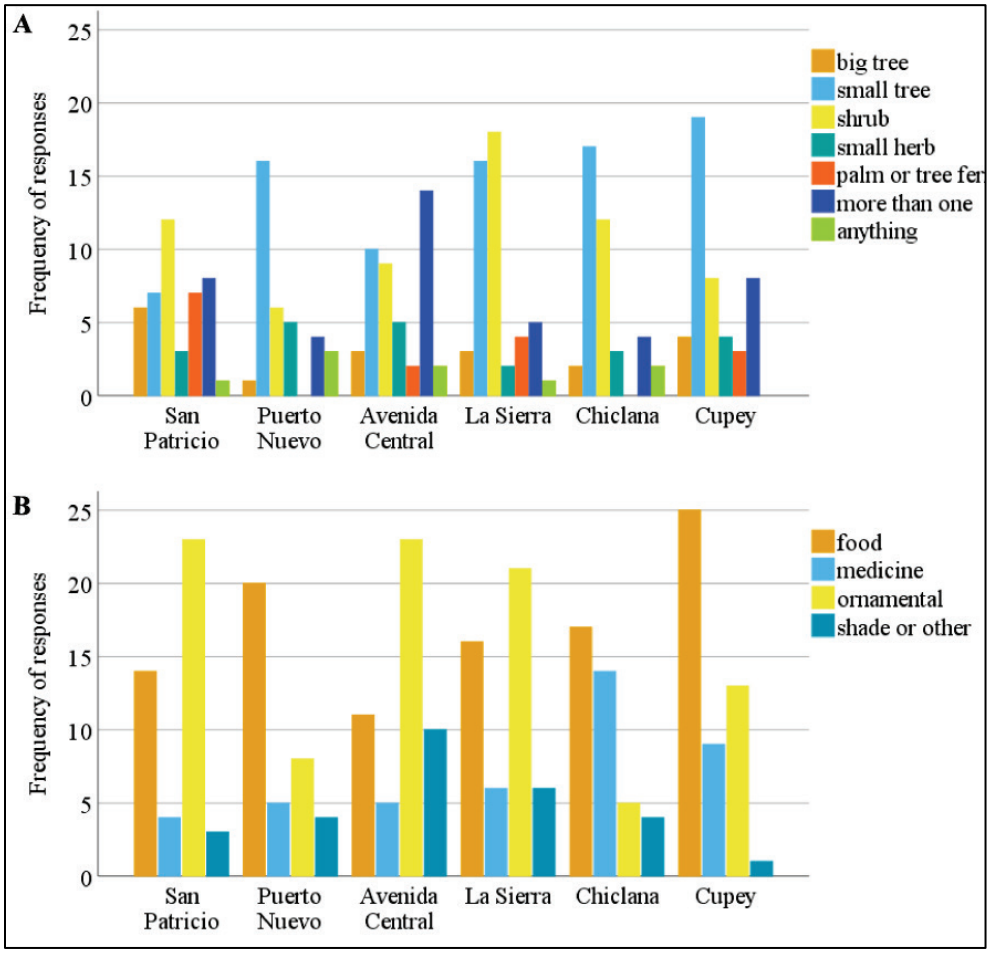

Figure 7. Frequency of responses on preferred (A) plant habit (Fisher's exact test, $X^2(30) = 47.393$, $p = 0.023$, N = 259) and (B) preferred ecosystem service (Fisher's exact test, $X^2(15) = 44.857$, $p < 0.001$, N = 267) for residents willing to exchange a non-native plant for a native plant.

We tested whether or not there was an association between *plant habit and ecosystem services* responses using a Chi-square test of independence for the four most mentioned *plant habits* (shrub, small herb, small tree, big tree) and the three most mentioned *ecosystem services* (food, medicine, ornamental). The association was moderate (Cramer's V= 0.224) but statistically significant ($X^2(6) = 17.089$, $p = 0.009$), with only two cells with expected frequency less than 5 representing 16.7% of all cells (Figure 8). Preference for *ornamental* plants was positively associated with preference for *shrubs* (z = 2.3), and negatively associated to *small trees* (z = -3.5). Preference for *food* plants was positively associated with *small trees* (z = 2.9) and negatively associated to *shrubs* (z = -2.4).

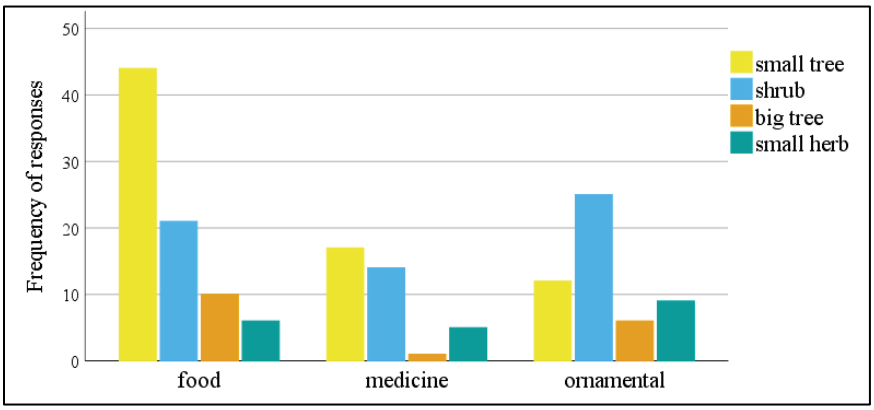

Figure 8. Frequency of responses (N=153) of each preferred plant habit for non-native plant exchange in relation to preferred ecosystem service.

4.4 Preferred propagation method for plant gifts

When asked about the preferred *propagation method* for gifted plants, a simple majority (35.3% of respondents) preferred receiving *young plants,* followed by those who preferred *small plants* (21.8%), *seeds* (18.5%), *adult plants* (11.8%) and a 10.6% were

amenable to *more than one* type if not all of them. Only 9 responded preferring *cuttings* representing 2.2% of total responses. There was a statistical relationship between the preferred plant *propagation method* and *watershed location* of respondent, although it was weak (Figure 9). Chiclana residents preferred receiving *seeds* (33.8% of responses) more often compared to residents from other *watershed locations* and were less likely to respond preferring *young plants* as gifts (7.5% of responses) relative to the other *watershed locations*.

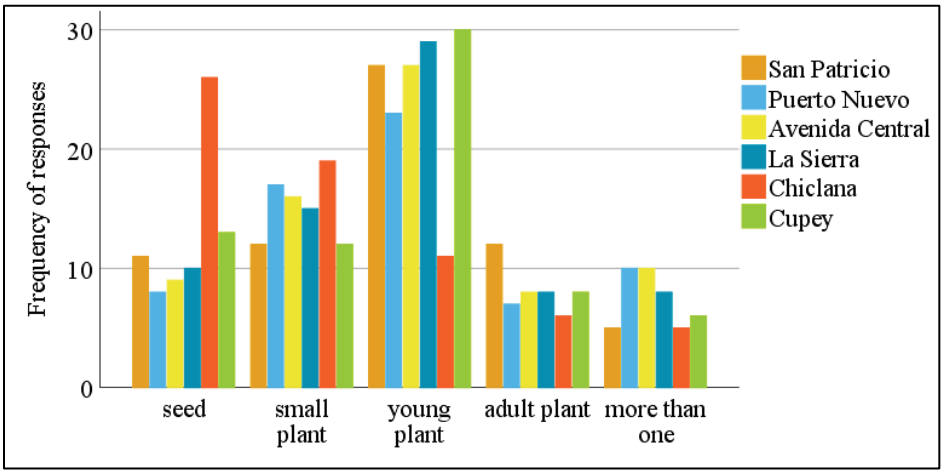

Figure 9. Frequency of responses for preferred propagation method for plant gift by watershed location. Chi-square: $X^2(20) = 36.417$, p = 0.014, N = 408; Cramer's V = 0.149 and strongest influence (z = 4.6) was for Chiclana.

4.5 Ranking of importance of ecosystem services

Friedman's test for differences in ranking of importance (5 = most important, 1 = least important) was statistically significant between *ecosystem services* (Table 8). Pairwise comparisons on mean rank differences showed that rankings were significantly different between all pairs of *ecosystem services* (all p's < 0.005) except between *food provision* and *air purification* (p = 1.0). *Food* provision and *air purification* had the

highest rankings followed by *shade*, *ornamental* and *wildlife habitat* which had the least (Table 8). Nevertheless, the proportion of residents that ranked *ornamental* services as most important relative to those that rated shade as most important was twice as high (15.5% vs 6.9% respectively). Only 3.8 % of respondents rated *wildlife habitat* as the most important service.

Table 8. Median, mean rank and rank sum for ranking of importance of ecosystem services (5 = most important, 1 = less important). Friedman's test $X^2(4) = 385.563$, $p < 0.001$, N = 417.

Ecosystem service	N	Median	Mean rank	Rank sum
air purification	418	4	3.76	1575
food	419	4	3.75	1577
shade	418	3	2.95	1237
ornamental	419	2	2.47	1042
wildlife habitat	417	2	2.07	869

When analyzing differences in the frequency of responses across *watershed locations* for each ranking category (Figure 10), these were only significant for *ornamental* services, *food provision* and *wildlife habitat* (*ornamental*: $X^2(5) = 16.018$, $p = 0.007$, N = 419; *food provision*: $X^2(5) = 20.848$, $p = 0.01$, N = 419; *wildlife habitat*: $X^2(5) = 13.691$, $p = 0.018$, N = 417). Tests showed no significant differences between rankings for *shade* ($X^2(5) = 5.540$, $p = 0.354$, N = 418) or *air purification* ($X^2(5) = 9.223$, $p = 0.1$, N = 418) across *watershed location*. For the *ornamental* service, post hoc analyses revealed significant differences only between *Puerto Nuevo* (mean rank = 171.44) and *San Patricio* (mean rank = 235.46; $p = 0.022$), and between *Puerto Nuevo* and *Avenida Central* (mean rank = 239.49; $p = 0.009$). For *food provision* significant differences in mean ranks were found only between San Patricio (mean rank = 168.05) and *Chiclana* (mean rank = 253.40; $p < 0.000$) and no other group combination. In terms of *wildlife*

habitat analyses showed significant differences only between *Chiclana* (mean rank = 199.10) and *La Sierra* (227.63; $p = 0.052$) as well as between *Chiclana* and *San Patricio* (mean rank = 185.63; $p = 0.054$).

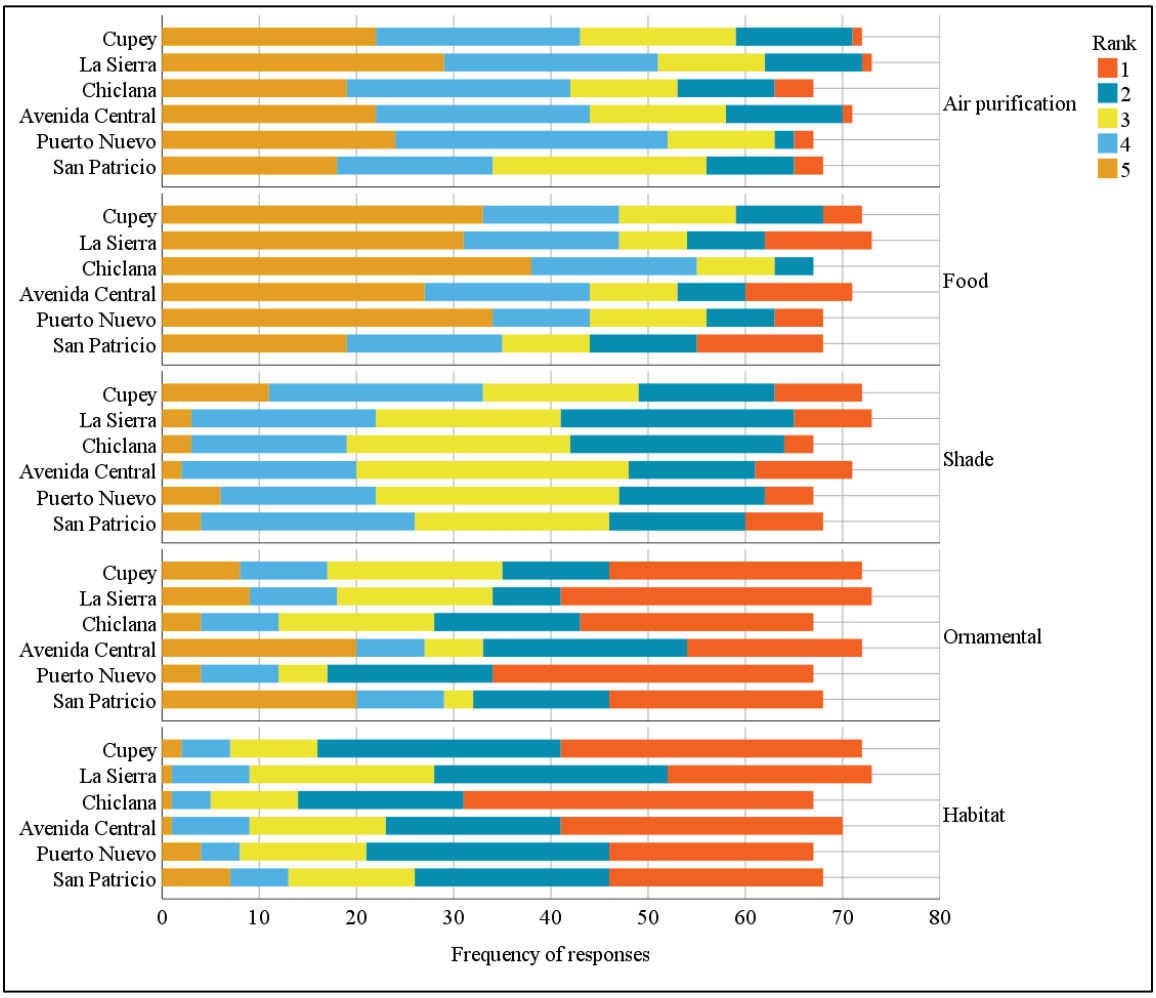

Figure 10. Frequency of responses for ranked ecosystem services by watershed location. For each of the ecosystem services 5 = more important and 1 = less important.

5. Discussion

This study found opportunities for integrating residential landscapes of the San Juan Metropolitan area as places for native species management by exploring socio-

psychological factors that may influence yard management decisions and the acceptance of local strategies. Overall, most residents in the Río Piedras Watershed expressed a preferential attitude toward native plants (or plants from Puerto Rico) and this sentiment seemed widespread across watershed locations and socioeconomic groups. Value orientations and socio-demographics help explain the strong favorable attitude toward native species that was self-reported by residents. Households with bigger families, older people and residents with value orientations related to *sense of place* were more likely to express a strong positive attitude toward native plants. Most residents also expressed willingness to replace non-natives plants for native ones, driven by a preference for small food trees and ornamental shrubs. As people get older the likelihood of expressing strong favorable attitudes toward native plants increases, but their willingness to modify their yards by exchanging non-natives for native plants decreases. Food provision and air purification were recognized as the most important ecosystem services. Below I discuss how these findings help identify research gasps as well as opportunities and limitations for conservation initiatives associated with urban trees.

5.1 Attitudes toward native plants

Studies exploring resident's attitudes toward native vegetation specifically have been rarely explored. Nonetheless, results from this study are partially consistent with findings from Almas & Conway (2018) in Ontario were residents self-reported a positive attitude toward native vegetation, which was positively related to years of formal education of respondents. Results from this study revealed a relationship between a positive attitude toward native vegetation with the residents age' but not with the years of

formal education, reinforcing the context-specific nature of these dynamics in residential social-ecological systems (Cook et al., 2012). In relation to planting activities, positive attitudes toward native plants in residential yards have also been reported in New Zealand, were residents expressed a preference for planting native plants, more so than planting non-natives or a mix of both (van Heezik et al., 2012). In contrast, the study in Ontario found that positive attitudes do not necessarily translate into native plant selection for planting, particularly if they were more expensive or created potential hazards (Almas & Conway, 2018). Also to consider is that a common limitation of self-report studies like this one is the role of social desirability bias in the answers of residents, where the respondents are inclined to answer favorably in order to present themselves in a positive way (Maio et al., 2019). There is a strong normative discourse that prevails in Puerto Rico on the importance of native plants that could potentially have influenced residents' responses. To prevent social desirability bias in responses, surveyors in this study avoided expressing themselves in favor in native species planting and purposely avoided using the term "native" plants and referred to them as "plants from Puerto Rico" due to the potential differences of how residents might define "native" when assessing attitudes and related values. We believe that combining ranking attitudinal statements with value-elicitation open-ended question helped to elucidate attitudes toward native plant species and helped mitigate the common sources of bias associated with self-report measures. Overall, while positive attitudes toward native plants do present an opportunity for conservation initiatives in my sites, other social factors suggested by this study may limit their effectiveness if not considered (see sections 5.2 and 5.3).

5.2 Value orientations associated with native plants

Value orientations helped explain a strong favorable attitude toward native species with *sense of place* dominating residents' responses and that was consistent across watershed localities and socio-economic groups. *Sense of place* theory addresses how interconnected social and biophysical components are in alignment with social-ecological theory (Masterson et al., 2017). Although these are complex, context dependent and changing constructs (Wolf, 2017), the way residents value, attach to and feel about a place such as their yard and its vegetation, could potentially relate to the way they modify those spaces (Chapin & Knapp, 2015). Out of the three protruding place constructs in environmental psychology research (place identity, place attachment, place dependence) (Jorgensen & Stedman, 2001), responses behind *sense of place* in this study were mainly motivated by residents' place identity, highlighting personal cultural and national identity. Research in Colombian cities has also found that people associate urban forests with a sense of identity and tradition, and that trees specifically provide a *sense of place* (Ordóñez-Barona & Duinker, 2014a). The value of native plants to provide a *sense of place* to residents of the RPWS seems to be of importance in the local context, particularly due to the lack of substitutes of built or grey infrastructure to replace them (Keeler et al., 2019). Nevertheless, other studies have found that non-native species can provide *sense of place* due to their cultural importance (Kirkpatrick et al., 2012). Although we did not evaluate this relationship on this study, there is a possibility that residents attribute *sense of place* to species that they believe to be native, but are not (i.e., because they are naturalized). This research group is currently evaluating if this is the case using data collected from the present survey.

Value responses classified as *conservation* included themes such as: avoid extinction, species conservation, avoid invasives, diversity, avoid ecosystem disruption, species coexistence, reduced ecological impact, and species competition. These responses seem to align with arguments in favor of native species conservation such as habitat modification (Vitousek et al., 1997)**,** loss of species (Chapin et al., 2000; Lowe et al., 2000), competition for resources (Reichard et al., 2001; Van Ham et al., 2013), effect on food webs (David et al., 2017), or pests (Manchester & Bullock, 2000). There was also a number of responses that referred to *local adaptations*, expressing that native plants "*adapt to the environment and do not generate imbalance*" in concurrence with the protagonist arguments behind most native species campaigns alluding to the higher fitness of species populations to their native environment (Lascoux et al., 2016). Not all residents agreed with giving preference to native plants reflecting their *biospheric* values more frequently than other factors as a reason to explain their strong negative attitudes toward native plants. These responses reflected residents concern over plants for their own sake beyond personal gain (Milfont & Duckitt, 2010; Scott et al., 2016; Steg & de Groot, 2019). Overall the results provided a variety of value themes (see also section 5.4) that underlie their attitudes responses which seem to be minimally incorporated into current strategies. A reason that might explain these differences could be differences in values and attitudes between practitioners and lay people when it comes to biodiversity conservation (Selge et al., 2011; Fischer et al., 2014). Another potential influence for response variation is the presence of different ethnic groups currently living in San Juan, which might have influenced responses in a way that was not captured by the survey. Ethnicity has been found to be linked to attitudes and values related to urban forests and

was not incorporated in this study (Ordóñez-Barona, 2017). Future research can explore the potential contribution of ethnicity in attitudes and values toward native vegetation. All said, the local adaptation value of native species which dominates the local management and conservation initiatives in Puerto Rico, might not be equally held by RPWS residents. This is an important issue to consider in guiding local conservation initiatives focused on residential private spaces.

5.3 Willingness to trade non-natives for natives

The majority of people expressed willingness to change their non-native plants for natives and that was related to preferences for plant habit and plant benefits (i.e., ecosystem service). Residents did indeed indicate that in order to exchange non-native plants for native ones they would prefer species that were small trees and shrubs and their preferred services food and ornamental. These findings coincide with the actual composition of San Juan yards which are dominated by shrubs, ornamental and food species (Vila-Ruiz et al., 2014) and the high awareness of food provision services of yard trees of RPWS residents (Chapter 2, Olivero-Lora et al., 2019). On the other hand, the willingness to modify the spaces by replacing non-natives with natives decreases as people get older, which is consistent with previous findings where the likelihood of RPWS residents making green infrastructure improvements to their yards within the next five years decreased with age (Meléndez-Ackerman, Nytch, et al., 2016). Population rates in San Juan are in decline and the median age of the island's population is increasing (Flores & Krogstad, 2019). Thus, demographic changes in San Juan may present a barrier to long-term implementation of native planting strategies that involve plant exchanges. It

is important to note that previous studies have documented that many residents were willing to replace a non-native with a native, but some residents were willing to do the opposite (Kirkpatrick et al., 2012). However, we did not explore their willingness to conduct the opposite behavior (replace native for non-native) which may be a one-sided view of the issue and their expressed behavioral intentions.

5.3 Ecosystem services

Proposed conceptual frameworks in residential social-ecological systems include *sense of place* as an ecosystem service and argue that the identities shared by residents could influence their landscape preferences and that promoting them could have long lasting effects (Cook et al., 2012). Although we classified *sense of place* as a separate value domain because of its recurrence in resident's responses, it is worth mentioning that these findings can be incorporated into agendas that use an ecosystem services framework approach. *Sense of place* has been classified as a cultural ecosystem service in the Millennium Ecosystem Assessment as cultural heritage ("*association of an individual of natural or semi-natural features with identity, community or society*"; Daniel et al., 2012). In this sense, native species conservation and well-being derived objectives can be aligned because actions also "*help maintain sense of place and culture identity related to native biodiversity*" (Hausmann et al., 2016). Identifying those species that bring a local *sense of place* should be considered in the implementation of ecosystem-services approaches to yard green-infrastructure planning. The combined results also support the importance of benefits related to non-cultural services, namely, food provision, ornamental and air purification services to RPWS residents over other services when it

comes to potential yard plantings but only food provision and ornamental services would seem to have immediate influence on the willingness to exchange non-native plants for native ones.

My findings on the values attributed to food provision relative to other ecosystem services (including wildlife habitat provision) complement previous studies with the same population that found residents' high awareness of food provision by yard trees (Chapter 2, Olivero-Lora et al., 2019), the use of yards as a complementary source of food acquisition (Garcia-Montiel et al., 2014), the high frequency of residential yards with food plants (Vila-Ruiz et al., 2014), and the most frequently mentioned ecosystem services provided by their yards (Meléndez-Ackerman, Nytch, et al., 2016). Residential yards can also play an important role in food security (e.g., access to fresh produce) while facilitating community and social cohesion through exchanges (Gray et al., 2014). Ornamental cultural services, as in this study, are commonly valued in urban areas (Camacho-Cervantes et al., 2014; Baur et al., 2016; Ordóñez-Barona et al., 2017; Avolio et al., 2018; Gwedla & Shackleton, 2019). One argument is that aligning multiple ecosystem services goals for this site that include at least ornamental and food services with local conservation goals may encourage native vegetation plantings. However, this may be difficult to achieve. An important limitation to the planting of natives in residential yards in San Juan is the lack of available native plants in local stores and nurseries (Torres-Camacho et al., 2017). Residents from RPWS self-reported that they buy plants mainly from departmental stores and private nurseries, but inventories performed at those local commercial establishments found only 8.3% of all plants were native (Torres-Camacho et al., 2017). An obvious strategy would be to promote the

increase of available native plant material in commercial establishments whenever possible. Developing a local inventory that includes native species that provide these locally valued services should be a priority.

5.4 Management implications

Research findings present some areas of opportunity and some potential barriers that should be considered for the development of these and other green management actions directed towards residential yards within RPWS. For example, the expressed preference for young and small plants as gift is in alignment with most local initiatives, although certain differences might occur depending on yard watershed location (e.g., seed preference for Chiclana residents) which might need to be considered for future campaigns. A common reason why conservation messages often fail is the way messages to stakeholders are framed, in a way that only appeal to a subset of the intended audience (Ives & Kendal, 2014). Since people value native plants in different ways, pitching messages of local campaigns in different ways (beyond biodiversity conservation arguments) could have a better reception with residents.

This study supports the notion that "nativeness" can be a plant species trait that contributes to an important cultural service (*sense of place*). Studies indicate that *sense of place* can raise support for management, promote stewardship and volunteering, and even help with fundraising (Ryan, 2005). Baur et al. (2016) suggest that "*expanding the ecosystem narrative by including intangible benefits, such as sense of place or aesthetic value, can help connect managers with the general public*". Frameworks that incorporate *sense of place* into biodiversity conservation decision-making like the one defined by

Hausmann et al. (2016), provide an alternative to identify actions such as the conservation of native biodiversity traits as a way to "*maintain cultural identity and sense of place that is related to native biodiversity*". Nonetheless, *sense of place* as cultural *ecosystem service* is overlooked and there is a lack of information on how to incorporate it as a cultural service into decision-making and management (Hausmann et al., 2016; Larson et al., 2016). This study was limited by a lack of understanding on how to operationalize and incorporate *sense of place* into ecosystem-based management approaches beyond outreach efforts and this is an issue that deserves more attention.

Efforts focused in providing plant material centered on specific ecosystem services (whether by plant gift campaigns or modifying local nursery offerings), particularly focused in the provision of food trees and ornamental shrubs, may have a strong potential of acceptance in RPWS residences. Results support the idea that local forestry efforts that promote native plantings could benefit from incorporating ecosystem services goals that relate to food production and ornamental services when possible. Results indicate that a considerable fraction of residents that prefer native plants (22%) do relate their preference to their socio-ecological functionality (i.e., provision of *ecosystem services*). This suggest that for at least a sector of the population integrating an ecosystem service approach might strengthen support for native tree species conservation.

Some studies argue that valuation of species should not be focused on their origin but on their functions (Davis et al., 2011; Lugo & Brandeis, 2005). A useful approach to adaptive urban forest management in a regional context has been proposed by Dwyer et al. (2003) that integrates (1) needs and attitudes of the community, (2) what urban forest

structure is necessary to best address those needs, (3) periodical re-assessment of those needs and urban forest structure to ensure management plans remain appropriate. Also, research on the links between residents preference for traits, nursery offering and yard species composition, advocate for increasing taxonomic and functional diversity of nursery offering that reflect services (and minimize disservices) valued by residents (Avolio et al., 2018). Thus, one offset of the *ecosystem services* approach to species conservation in urban context is that in some cases non-native species might be providing the same services or even more services than native species. Current focus on managing for ecosystems services objectives will benefit from incorporating the values and preferences of local residents, identifying differences in performance from native and non-native species, and choosing proposed actions accordingly (Dobbs et al., 2017). Indeed, gathering information about the added services provided by native compared to non-native species in San Juan yards would be important to some residents based on my findings.

6. Conclusion

The use of socio-psychological approaches and value elicitation of native species framed in cognitive hierarchy theory, provides interesting insights and new areas of opportunity to consolidate residential values with prevailing institutional goals. Results suggest that local initiatives for native plantings that do not necessarily include ornamental shrubs or small food trees would be disconnected from known factors that would influence the residents' willingness to modify their green spaces. While, more research is needed to clarify the disparities on whether native species really do better in

urban environments, there are social opportunities taking place in San Juan. Alternative native planting campaigns such as re-branding campaigns to highlight native origin (i.e., eco labeling) while incorporating preferred ecosystem services (i.e., food provision, air purification) and creating markets for these preferred plants traits are some examples of possibilities to be considered. Notwithstanding, this study calls for the need to reflect on the differences and similarities between the needs and values of city dwellers and on defining the goals of conservation (species-biodiversity, ecosystem services) to better articulate conservation initiatives in residential areas in San Juan. Urban forestry and conservation approaches would benefit from redefining goals in urban areas, aligning objectives and strategies within different entities (green infrastructure plans, national and local urban forestry plans, etc.) and considering multiple solutions beyond native tree planting and giveaway programs.

Chapter 4. Implications of hurricane-driven changes in vegetation structure and ecosystem services provision in residential yards of San Juan, Puerto Rico

1. Abstract

Residential urban forest resources and the ecosystem services they provide are likely vulnerable to the effects of extreme weather events, such as tropical cyclones. Understanding these effects is an important step to minimize potential risks and to increase resilience of these systems. A total of 69 single-family housing units were surveyed before and after the passage of hurricanes Irma and Maria (2017) in San Juan, Puerto Rico, using i-Tree Eco inventories to study the changes in ecosystem services provided by yard vegetation. The specific objectives of the study were to (1) evaluate hurricane effects on structure, composition and condition of woody species and large herbs, (2) estimate changes in functions and loss of ecosystem services, and (3) evaluate species-specific differences in hurricane-driven ecosystem services changes and mortality. Results showed that the combined yard community structure went from having a marked dominance structure (dominated by *Musa x paradisaca*) to one where such dominance had disappeared. Yard vegetation remained highly dominated by non-native species before and after the hurricanes, but lost approximate 10.5% of the species, experienced a 27% reduction of standing stems (N = 136) and an overall mortality of 31% (N = 152). Stem reduction, tree cover loss and structural changes in yard vegetation translated into a reduction of ecosystem services of approximately 19.6% for air pollution reduction services, 19.9% of avoided runoff and 13.1% cooling energy savings. Food provision services (provided by large proportion of yards before the hurricane) exhibited

much higher losses (41.9%) than ornamental services (15.6%) based on the number of stem losses. There was a significant decrease in the median condition of all inventoried plants from fair to critical. We found no statistically significant relationship between woody species and mortality, but woody plants with small stem diameter were more likely to die. Individual species differed in their contribution and losses of ecosystem services. The reduction in ecosystem services provided by yards following this extreme event could have important implications for urban resilience in residential social-ecological systems. Locally, more research is needed to develop the urban forests using residential areas in ways that not only promote sustainability (by providing multiple ecosystem services) but also resilience to extreme storm events.

2. Introduction

Current urban forestry and green infrastructure discourse advocates the use of an ecosystem services framework to advance sustainable management of city landscapes (Dwyer et al., 2003; Tratalos et al., 2007; Dobbs et al., 2011; Thomas & Geller, 2013; Krajter Ostoić & Konijnendijk van den Bosch, 2015; McPhearson et al., 2015; Miller et al., 2015). A general assumption of this framework is that ecosystem services maximization can be achieved by adequate allocation of trees and other forms of vegetation that yield desired functions, that in turn provide ecosystem services that can be valuated and linked to urban sustainability goals (Carreiro et al., 2008; Dobbs et al., 2017). Privately managed residential spaces are increasingly in the spotlight as important contributors of ecosystem services, because they contain the majority of the urban forests resource and provide functional connectivity of the green infrastructure of cities (Gaston

et al., 2005; Cook et al., 2012; Calvet-Mir et al., 2012; Nowak & Greenfield, 2018). Despite this, the urban forests resources in both public and private forms, can be vulnerable to the unwanted effects of extreme weather events (i.e., hurricanes, tornadoes, snow storms, heat waves, droughts) (Burley et al., 2008; Ordóñez-Barona & Duinker, 2014b; Foran et al., 2015; Yan & Yang, 2018; Steenberg, Millward, et al., 2019) and increasing trends in extreme weather events occurrence could potentially be making these resources increasingly vulnerable (Méndez-Lázaro et al., 2016; Yin et al., 2018). Therefore, understanding the responses of urban forests components to these events is critical to the planning of sustainable and resilient urban green infrastructure systems.

Residents of urban areas are more vulnerable to natural disasters and experience more losses compared to natural areas due to the high concentration of people, infrastructure and services (Dickson et al., 2012; McPhillips et al., 2018; Elmqvist et al., 2019). Air pollution is higher in cities due to the high demand for fossil fuels (Fenger, 1999). The urban heat island phenomenon associated to cities makes them more vulnerable to extreme heat-waves (Méndez-Lázaro et al., 2015, 2016). Urban areas are also more vulnerable to flooding caused by the increase in impervious areas with consequences to human life and property (Güneralp et al., 2015). Many cities are located in coastal areas and are vulnerable to the effects of sea-level rise and coastal erosion (Grimm et al., 2008; Nicholls & Cazenave, 2010; Hallegatte et al., 2013). At the same time, expected global changes in climate (increases in temperature, and changes in precipitation patterns) would also be accompanied with increases in extreme weather events (storms, droughts, heat waves, etc.) all of which will affect cities disproportionately. Green infrastructure has gained popularity as a strategy to improve

the well-being of urban residents not only by directly providing multiple ecosystem services to human residents but also as a way to mitigating problems associated to the urban condition and climate change (Gill et al., 2007; Matthews et al., 2015; Zölch et al., 2016; Lafortezza et al., 2017). However, to meet these goals green infrastructure needs to be planned and implemented in ways that they can be resilient and sustainable in the face of global change.

2.1 Hurricanes in the Caribbean Region and effects on forest structure, composition and condition.

Hurricanes are a common occurrence in the Caribbean, but their intensity and effects are expected to increase in the region (Gould et al., 2015). There is a vast amount on information on how tree species in the Caribbean respond to hurricanes but most of it is circumscribed to non-urban environments. Native species may be more able to tolerate to hurricane damage than non-native ones (Brokaw, Zimmerman, et al., 2012). Hurricanes effects can result in high instant tree mortality (Lugo & Scatena, 1996; Lugo, 2000; Brokaw, Zimmerman, et al., 2012). Recent estimates of tree mortality by hurricane María on these same natural systems, found that tree mortality doubled relative to other major hurricane disturbances that have passed through the island (Uriarte et al., 2019). Some of the structural variables related to vulnerability of tropical plant species to hurricane events are height, crown dimensions, leaf features, stem diameter, stem or wood density, and their root traits (Zimmerman et al., 1994; Van Bloem et al., 2005; Uriarte & Papaik, 2007; Canham et al., 2010; Brokaw, Crowl, et al., 2012). For example, for the subtropical moist forest of Puerto Rico, faster growing species and larger trees

have been reported to experienced more damage (Ostertag et al., 2005). Indeed, height was found to be the most influential trait related to vulnerability to hurricane wind forces following hurricane María in 2017 (Uriarte et al., 2019). Also, high wood density species have been found to be more resistant to stem damage and mortality in studies analyzing the effects of hurricanes disturbances in tropical forests (Zimmerman et al., 1994; Canham et al., 2010; Uriarte et al., 2019).

While findings have supported this generalization in natural systems, empirical evidence suggesting that the same occurs in urban ecosystems, which is characterized by novel assemblages of species with different abiotic and biotic components, is sparse. One study in the Florida panhandle, found that species with high wood density were more resistant to hurricane damage and mortality following hurricane Ivan (category 4) (Duryea, Kampf, & Littell, 2007). Another in study in the city of San Juan, Puerto Rico following hurricane Georges, a category 3 event confirmed that taller trees were also more prone to hurricane-mortality (Francis, 2000). A comparison between the effects of hurricanes Jeanne and Charlie in Florida and hurricane Georges in Puerto Rico, found higher survival for Florida native species but not for Puerto Rican species (Duryea, Kampf, Littell, et al., 2007). Results on the differences in mortality rates between urban and non-urban areas are inconsistent. Although we lack the empirical evidence to determine the performance of urban native species in the face of hurricane disturbances, there is sufficient information to support that different species respond differently and have different sensitivities to urban and natural stressors (Francis, 2000; Steenberg et al., 2017; Hilbert et al., 2019).

Along with structural traits and species composition, plant condition is an important variable determining the vulnerability to hurricanes in both natural and urban conditions. Tree condition, is a common metric related to the health of forest resources, particularly in urban forestry, and trees in poor conditions experience high mortality rates and are more sensitive to stressors or disturbances (Nowak et al., 2004; Lu et al., 2010; Steenberg et al., 2017; Hilbert et al., 2019). Studies in the subtropical moist forest have found a strong relationship between hurricane structural damages and past disturbance history (Ostertag et al., 2005). Long-term hurricane data from Jamaica's tropical forest found previous historical hurricane damaged plants experience up to eight times higher mortality rates than undamaged after 19 years later (Tanner et al., 2014). Branch and canopy damages caused by hurricane winds also have delayed effects on tree mortality months and years after a the event occurrence (Walker, 1995; Lugo, 2000; Uriarte et al., 2004). Thus, vegetation condition before a hurricane event can be related to mortality and assessing the condition of remaining vegetation could be an indicator of potential latent mortality and of vegetation vulnerability in the face of future storm events.

2.2 Case study

During the 2017 Atlantic hurricane season, the island of Puerto Rico suffered the abatement of two intense hurricane disturbances. Hurricane Irma passed to the north of the island (90 km) of Puerto Rico as a category 4 hurricane on September 7, 2017 leaving perceivable effects on the island's green and grey infrastructure (Uriarte et al., 2019). Two weeks later hurricane María, the strongest hurricane in record to pass the island since 1928 (Pasch et al., 2019), entered the island as a category 4 storm and was declared

a catastrophic event with island-wide effects. Maximum sustained winds of hurricane María were 155 mph and the rainfall intensity for the 24-hour period during hurricane María passage are the highest ever recorded for the island (Ramos-Scharrón & Arima, 2019). Top down approaches using remote sensing estimated that about 23 to 31 million trees were killed or suffered severe damages (Feng et al., 2018), other estimates fluctuate between 20 and 40 million killed or severely damaged trees (Uriarte et al., 2019). The combined effects of hurricanes Irma and María produced an estimated 51% immediate loss of greenness for the Luquillo Experimental Forest in Puerto Rico, and 31% in all U.S. Caribbean region which includes Puerto Rico and the Virgin Islands (Van Beusekom et al., 2018). Hurricane María killed twice as many trees as previously studied hurricanes like Hugo and George, the number of broken trees was three times more and in the case of some species reaching rates of up to 12 times more breakage (Uriarte et al., 2019). Other assessments of effects on vegetation revealed that for urban areas such as San Juan, tree cover losses were as high as 24.8% and losses in health-related ecosystem services provision like pollution removal, diminished in an estimated 30% for the city (Meléndez-Ackerman et al., 2018).

The passage of hurricanes Irma and María presented an opportunity to evaluate the extent of effects following these large storm events. We took advantage of the interruption of ongoing studies of residential backyards in San Juan to evaluate vegetation and ecosystem services changes following the passage of hurricanes Irma and María. We conducted a rapid post-hurricane assessment by revisiting yards that had been inventoried prior to these events using the tool i-Tree Eco (www.itreetools.org). These standardized methodologies have been applied in many cities around the world for the

inventory of their tree resources and the valuation of urban forest ecosystem services (Nowak et al., 2018). In this study, the main objectives were to: (1) evaluate hurricane effects on structure (plant height, stem diameter, leaf area, leaf biomass, basal area, tree condition), composition (number of plants and species) of woody species and large herbs (global change, yard change), (2) estimate changes in functions and loss of ecosystem services (global and by yard), and (3) evaluate species-specific differences in hurricane effects and mortality. The characteristics of the built environment surrounding urban vegetation such as site type, site size and impervious cover are potential determinates of their vulnerability to different stressors (Steenberg, Millward, et al., 2019). Yard size has been found to be positively associated to plant abundance and species richness in residential yards of the Río Piedras Watershed (Vila-Ruiz et al., 2014) but the question remains on whether variation in hurricane-related losses of ecosystem services and biodiversity would be influenced by yard size. Results help inform urban green infrastructure planning and urban forests management strategies to develop urban green infrastructures that are less vulnerable to extreme weather events and that increase the resilience of residential social-ecological systems.

3. Methods

3.1 Study Site

The San Juan Metropolitan Area (SJMA) is the largest urban area on the island of Puerto Rico. During the 1900s, the SJMA experienced rapid urbanization that led to loss of forest cover (Lugo et al., 2011). Out of the six municipalizes that comprise SJMA, San Juan is the most populous of all with an estimated population of 337,288 people (U.S.

Census estimate for 2017). The main watershed in San Juan is the Río Piedras Watershed (RPWS) which is part the San Juan Bay Estuary (SJBE) and is very variable in ecological conditions (Lugo et al., 2011). The Río Piedras River originates on barrio Caimito at 150 meters of elevation and flows for 16,000 m north where it merges with the Puerto Nuevo River and via channelization, unites with Martin Peña Canal and flows into the Bahía de San Juan (Lugo et al., 2011). The river has been extensively modified due to anthropogenic activities and channelization has been ongoing on the Puerto Nuevo River tributary for the past 30 years as mitigation for possible extreme 100-year flooding events, but the watershed still suffers from recurrent urban flooding events (Ramsey et al., 2019). The RPWS total area has been estimated on of 49 km^2 since the first hydrological investigations (Lugo et al., 2011) but considering the latter modifications where now the Puerto Nuevo River joins the Río Piedras River, the total area has been reported up to 67,000 m^2 (Ramírez et al., 2014; Meléndez-Ackerman, Olivero-Lora, et al., 2016). Species combinations in San Juan seems result of both natural processes (self-organization, naturalization, succession) as well as human species selection contribution to novel ecological systems with combined assemblages of native and non-native species (Lugo & Helmer, 2004b; Brandeis et al., 2014; Muñoz-Erickson, 2014). Due to its warm and moist maritime climate, the vegetation of the city of San Juan's grows rapidly (Lugo & Helmer, 2004b; Muñoz-Erickson, 2014). Introduced species result from planting decisions as well and natural regeneration, which is characterized by three species: *Albizia procera*, *Spathodea campanulata*, and *Melaleuca quinquenervia* (Lugo et al., 2011). Overall, the number, size and distribution of trees in the SJBE watershed have been shown to affect the function of benefits. For example, outside the SJBE mangrove

forests, the introduced *Spathodea campanulata* is the most frequently encountered species and the species with highest carbon storage (Brandeis et al., 2014). Based on Holdridge's life zone system it is classified as subtropical moist forest life zone (Lugo et al., 1999) with mean annual precipitation ranging from 1509 mm on the coast to 1755 mm on more elevated areas, and a mean annual air temperature of 25.7ºC (Lugo et al., 2011). It has been estimated that 72% of all rainfall results in high runoff (Osterkamp, 2001) from volcanic rocks and impervious surfaces (Lugo et al., 2011; Ramírez et al., 2014). Rainfall is seasonal and coincides with hurricane season (June - December) and the dry season typically runs from January to April. May is usually a secondary wet period although it was not the case in 2015 when San Juan (and the rest of the Caribbean) experienced an extended drought. This is not the first time that an extreme drought has been reported for San Juan where records are shown for 1920s, 1930s, 1960s, 1970s, and 1990s (Lugo et al., 2011). The RPWS is also exposed to hurricanes from which the strongest recorded is San Felipe in 1928 with winds up to 240 km/h (Lugo et al., 2011) until the 2017 catastrophic hurricane season. Maximum sustained winds for hurricane María were of 250 km/hr. with 381 mm to 507 mm of storm total rainfall estimated at location of the Río Piedras Watershed (Pasch et al., 2019).

3.2 Study design

Yards included in this study were located in the RPWS and initially, yard selection followed a stratified random sampling deigned at San Juan ULTRA six permanent monitoring sites for which a number of social-ecological studies have been conducted to date (Meléndez-Ackerman et al., 2014; Meléndez-Ackerman, Nytch, et al.,

2016; Torres-Camacho et al., 2017; Olivero-Lora et al., 2019). Originally, backyard vegetation surveys were conducted first between June 2016 and May 2017 as part of a larger project that was disrupted by the hurricanes before the inventory was completed. A total of 89 households were revisited during the month of October 2017 immediately after hurricane María crossed over the island of Puerto Rico. Rapid post-hurricane yard and vegetation assessments were conducted for 69 (75% success rate) backyards located in three of the initial six locations: Puerto Nuevo, Avenida Central and La Sierra (Figure 11).

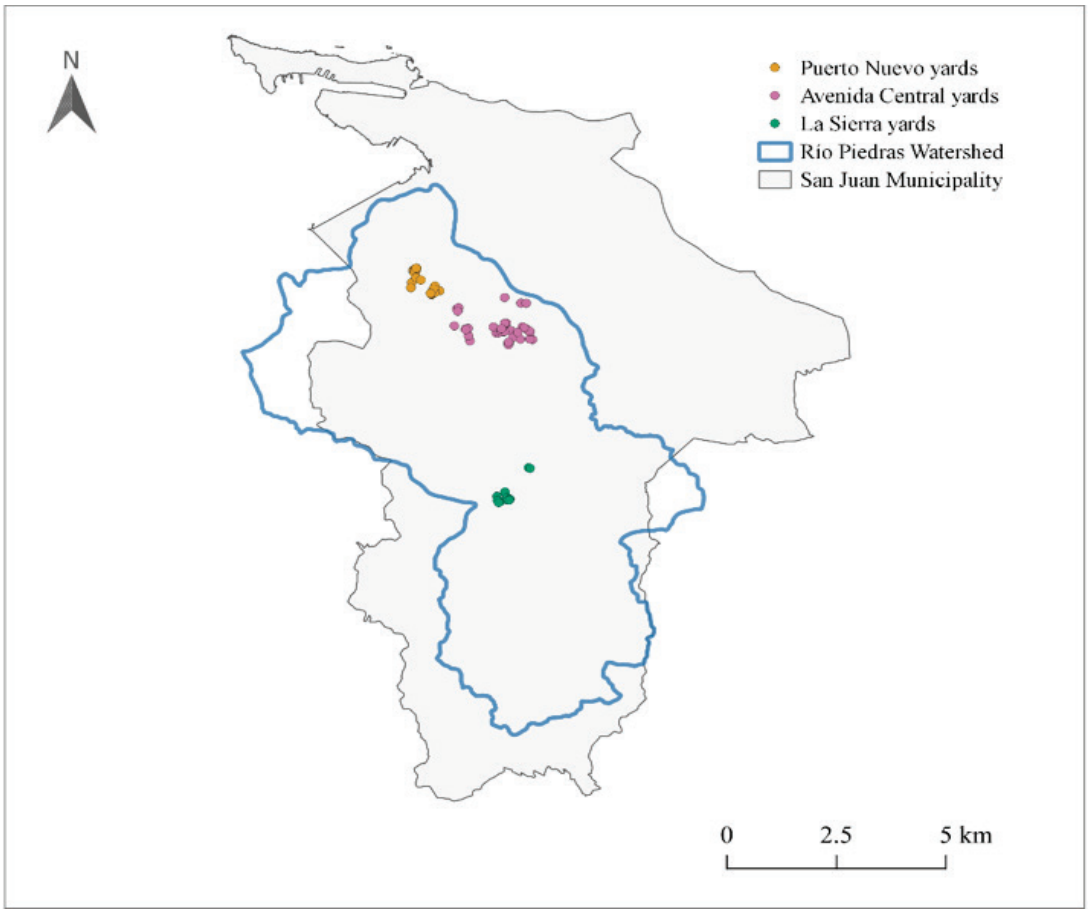

Figure 11. Map of the distribution of surveyed yards in relation to the Río Piedras Watershed and the San Juan Municipality.

3.3 i-Tree Eco model inputs

The i-Tree Eco v.6 modeling tool (former UFORE) was used to generate pre- and post-hurricane estimates of ecosystem services (Table 9). i-Tree Eco is part of a suit of peer-reviewed software tools developed by the U.S. Forest Service to assess urban forests structure and its related ecosystem services and disservices. To generate ecosystem services estimates, air quality data for model inputs [for carbon monoxide (CO ppm/hr.), nitrogen dioxide (NO^2 ppm/hr.), ozone (O^3 ppm/h**)**, sulfur dioxide (SO^3 ppm/hr.**)**, breathable suspended particulate matter (PM^{10} $\boldsymbol{\mu}g/m^3$) and fine particulate matter ($PM^{2.5}$ $\boldsymbol{\mu}g/m^3$)] as well as temperature (°C) and rainfall (cm/hr.) and wind velocity ($m\ s^{-1}$), obtained from the National Weather Service from the San Juan, Puerto Rico-Luis Muñoz Marín International Airport (LMMA, USAF:785263) station for the year 2012 (available in i-Tree Eco suit for San Juan). Project was configured as complete inventory to estimate values of individual plants and obtained data output was exported to excel spreadsheets to conduct further analysis. Values for electricity were modified to 20 cents per kilowatt-hour based on estimates for Puerto Rico in 2017 (U.S. Energy Information Administration, 2018) and heating costs to zero due to the lack of frost days in the tropical environment. Default values were used for the price of carbon ($188 per metric ton) and avoided runoff ($2.36 per m^3).

Table 9. Ecosystem services variables included in this study, their unit of measurement and source.

Ecosystem service	Unit of measurement	Source
carbon storage	kg	i-Tree Eco
gross carbon sequestration	kg/yr.	i-Tree Eco
avoided runoff	m^2/yr.	i-Tree Eco
oxygen produced	kg/yr.	i-Tree Eco
pollution removed	g/yr.	i-Tree Eco
cooling energy savings	Kwh / yr.	i-Tree Eco
food production	yes / no	database & literature review
ornamental value	yes / no	database & literature review

3.4 Field surveys and plant traits

Pre-hurricane field surveys consisted of complete structured vegetation inventories at each yard following i-Tree Eco protocols (i-Tree Eco Field Guide, 2019; i-Tree Eco User's Manual, 2019). At each yard all woody vegetation (trees, shrubs, palms) with stem diameter of 2.5 cm or higher were inventoried, plantains and banana "trees" were included in the vegetation survey due to high abundance in the yards. Structural variables included diameter at breast height (DBH), total tree height, crown dimensions (height, width, % missing crown), crown condition and crown light exposure. Plant location (distance and direction relative to building) were measured for plants over 6.1 meters of height and up to a distance of 18.3 meters from residence building. All individuals were photographed and identified to species level in the laboratory and in consultation with local experts. Most surveyed plant species were on the i-Tree Eco species database, but for those few that were not, the genus was used as a proxy. For each species included in the survey, it was determined if it had a food provision value (present or not), ornamental value (present or not) and its species origin (native versus non-native) following the most recent classification of RPWS yard plants (Vila-Ruiz et al., 2014) and

using the PLANTS database (USDA-NRCS, 2019), the Germplasm Resources Information Network (GRIN) (USDA, 2019) and consultation with related literature (Whistler, 2000; Little et al., 2001; Axelrod, 2011; Joglar & Longo, 2011). Wood density was also documented for each species using the global Wood Density Database (DRYAD) as first resource (Chave et al., 2009), and complemented with the African Wood Density Database (Carsan et al., 2012) and the Food and Agriculture Organization (Brown, 1997).

While pre-hurricane sampling included detailed structural variables (DBH, height, canopy width, canopy height, etc.), the urgency to conduct a rapid post-assessment and resource constraints limited the number of variables that we were able to re-measure. Each yard and individual plant was re-visited and photographed when present. To assess damage, the variable crown condition (defined as 1 - % dieback) was recorded for each individual plant and the occurrence of broken or uprooted trees were documented as well. We also recorded their recovery status as refoliation (new leaves or re-sprouting from trunks or stumps). Yard area (m^2) was estimated using Google Earth Pro v. 7.3. Out of the 69 revisited households, 15 (22.4%) of them did not have any woody vegetation or large herbs that met sample requirements (DBH $\geq$ 2.5 cm) and two of them did not have yard space in the property lowering the sample size to 52. The survey included a combined area of 0.7 hectares (7,129 m^2) made of all yards containing vegetation.

3.5 Statistical analysis

The following categorial variables were coded to conduct analysis (Table 10): food provision value, ornamental value, plant origin, plant habit, mortality, bole damage,

crown condition. Data was aggregated by into a new data matrix by yard and one by species for analysis. New variables were created for quantitative variables to define the magnitude of change (t_1 - t_2) for all individuals, and yard and species aggregated data. Descriptive statistics and frequency tables were generated for the description changes in vegetation structure, composition and ecosystem services provision. A chi-square goodness of fit test was performed to determine if there was a statistical difference in the frequency distribution of individuals across stem diameter (DBH) classes and as well as condition categories after the hurricanes using the observed values from pre-hurricane inventories. A Wilcoxon signed-rank test was also conducted to determine hurricane-driven changes in median condition (depended ordinal) of all individuals (N = 491). We ran Chi-square for independence to evaluate differences in the frequency distribution of stem diameter classes between dead and alive individuals in the post-hurricane assessment considering standardized residuals (z-scores) for post-hoc analyses of cell comparisons (Field, 2013; Agresti, 2019).

Table 10. List of categorical variables, their description and coding.

Variable	Variable description	Coding
plant origin	origin status related to the region	native (1), non-native (0)
plant habit	plant life form	tree (1), shrub (2), herb (3), palm (4), shrub/small tree (5)
crown condition	crown condition (1-%dieback)	dead (1), dying (2), critical (3), poor (4), fair (5), good (6), excellent (7)
presence	post hurricane presence dead or alive	yes (1), no (0)
mortality	overall mortality	dead (1), alive (0)
bole damage	snapped trucks and uprooted plants	none (0), snapped (1), uprooted (2)
refoliation	evidence refoliation or sprouting	yes (1), no (0)
reproduction status	evidence of flowers and/or fruits	none (0), flowers (1), fruits (2), both (3)
food provision value	food production value	yes (1), no (0)
ornamental value	ornamental value	yes (1), no (0)

3.5.1 Hurricane-driven changes in yard vegetation structure, composition and ecosystem services

The following variables were compared between inventories to evaluate hurricane driven changes: the number of plants, number of species, number of native species, number of food plants, number of ornamental plants, sum of canopy cover, leaf area, leaf biomass, basal area, and total provision of each i-Tree Eco estimated service (carbon storage, gross carbon sequestration, avoided runoff, oxygen production, pollution removal, cooling energy savings). Mean values per yard were used for stem diameter, height and leaf area index. Because normality could not be assumed, we ran an exact Sign test to evaluate the changes in yard variables before and after hurricanes. Only yards with vegetation were included in the analysis, 52 out the 69 households re-visited. The percent loss of ecosystem services ($\Delta = - (t_1 - t_2 / t_1) * 100$) was estimated by yard for further comparison of yard ecosystem services losses. To test the hypothesis that bigger yards had bigger losses in structure, composition and ecosystem services, Kendall's tau-b (τ_b) correlation statistic was performed to test the association.

3.5.2 Species-specific difference in hurricane effects

The relative contribution (0 - 100%) of each species to the total amount of each ecosystem service (ES) was calculated (species ES_i contribution / total ES_i contribution * 100). Then proceeded to construct an ecosystem service index (ESI) by adding the relative contribution of all eight services for each species for comparison. The ESI index reflected the relative contribution of multiple ecosystem services (carbon storage, gross carbon sequestration, avoided runoff, oxygen production, pollution removal, cooling energy savings, total food plants, total ornamental plants) by each species in my sample.

To estimate hurricane-driven changes, the percent loss of each service for each species was estimated. Mortality was estimated (% mortality = # dead indvs. Post / # alive indvs. Pre * 100) for each species (see Appendix C, Table C2 for full list of species and frequency data). Binomial logistic regression was performed to evaluate the effects of continuous structural variables (plant height, wood density, diameter at breast height), condition as ordinal, and species as a categorical variable on the likelihood of plants to be found dead in post-hurricane inventory. Only species with more than 10 individuals were included in analysis resulting (N = 194). The species included were: *Annona muricata* (N = 10), *Citrus aurantifolia* (N = 11), *Codiaeum variegatum* (N = 13), *Duranta sp.* (17), *Dypsis lutescens* (N = 22), *Ficus benjamina* (N = 27), *Hibiscus rosa-sinensis* (N = 39), *Mangifera indica* (N = 11), *Psidium guajava* (N = 14) and *Ptychosperma macarthurii* (N = 30). Due to the low frequency counts in crosstabulation between condition and mortality, categories were collapsed into three categories: bad (≥ 26% dieback), fair (11% to 25% dieback) or good (≤ 10% dieback). I estimated a "full model" with all variables included and proceeded to select final built model by adding and removing variables. The best model was defined as the one that lead to an increase in R^2 and a decrease in Akaike Information Criteria (AIC). All analysis was performed using SPSS v.25 (IBM Corp. Released, 2017).

4. Results

4.1 Hurricane-driven changes in vegetation composition and condition

Prior to the hurricanes a total of 491 plants were inventoried distributed across 95 different species, 16 of them classified as native. A total of 450 individuals (91.6%) were

non-native species and 41 (8.4%) were native species. At the time the total canopy cover was estimated as 4,760 square meters and small-stem plants dominated with 82.3% of all vegetation having a diameter of 15.2 cm or less. Following the hurricane sampled yards had lost 136 individuals (27.7%) and 9 species (9.5%). From those individuals standing, 16 were dead, increasing the number of dead individuals to 152 which represented an overall mortality of 31% for large yard vegetation. The proportion of native plants in post-hurricane assessments changed very little and remained low 10.7% with negligible changes in median values of native plants per yard (Table 12).

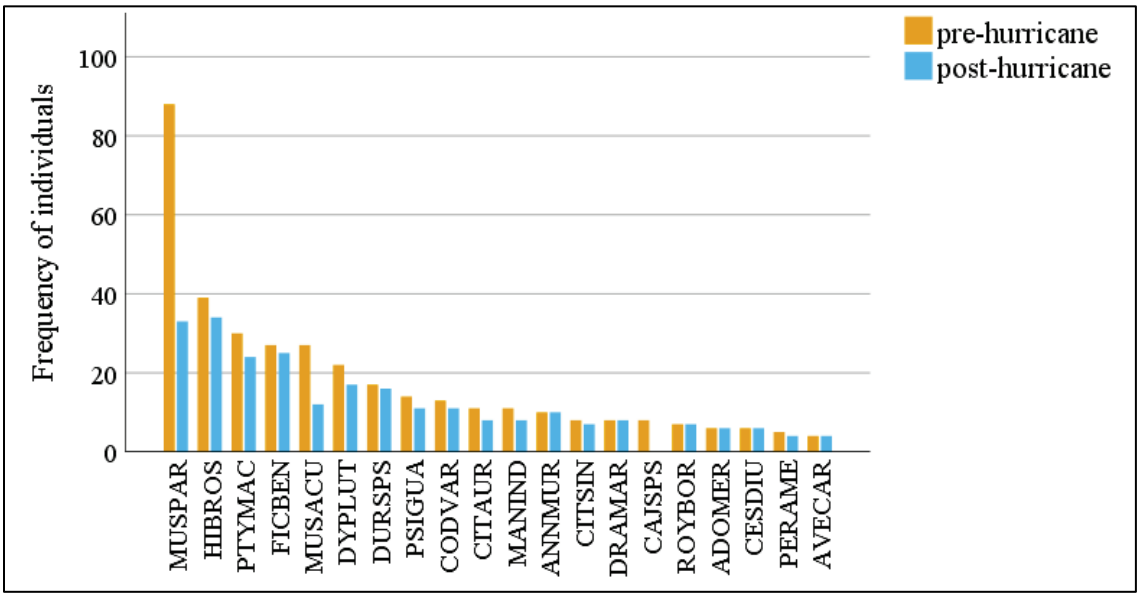

Figure 12. Frequency of individuals for 20 most abundant species pre-hurricanes (orange) and post-hurricane (blue).

The proportion of individuals classified as large herbs decreased from 23.4% (115 indvs.) to 12.7% (45 indvs.) due to the loss of *Musa* species (*Musa x paradisiaca* and *Musa acuminata*). The hurricane also changed the dominance structure of the overall yard vegetation community from one that was highly dominated by *Musa x paradisiaca*

to one that was not (Pre vs Post: 17.9% vs 9.3%) which suggests species-specific mortality differences in this community (see Appendix C, Tables C1 & C2 for species-specific frequency and mortality data). This community also experienced a slight increase in the combined proportion of shrubs and trees (Pre vs Post: 61.3% vs 70.0%) as well as the proportion of palms (Pre vs Post: 15.3% vs 17.2%).

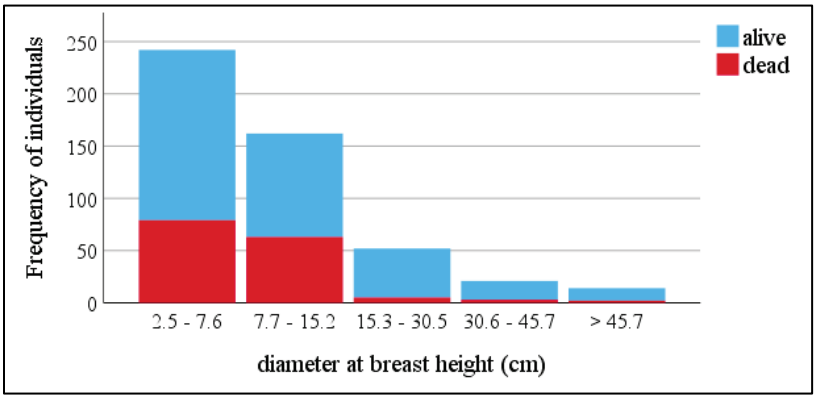

Figure 13. Frequency of live (blue) and dead (red) individuals at each DBH class in post-hurricane inventories.

Distribution of DBH classes did not change from that expected based observation from pre-hurricanes inventories (Chi-square goodness-of-fit test: $X^2(4) = 5.946$, $p = 0.203$, N = 355). However, the relative distribution of DBH categories for stems that were alive and dead in the post-hurricanes inventory was different (Chi-square test: $X^2(4) = 21.864$, $p = 0.000$, N = 491; Figure 13). The relative proportion of dead individuals was higher in smaller DBH categories (< 15.2 cm) relative to bigger DBH categories (> 15.3 cm). More individuals died in the 7.7-15.2 cm DBH category ($z = -2.7$, $p < 0.01$) and less in the 15.3-30.5 cm category ($z = 3.5$, $p < 0.001$). Species-specific changes were related at least in part to variation in DBH (see section *3.3 Species-specific hurricane-driven changes* below). The condition of surveyed individuals exhibited a dramatic decline

following the hurricanes (Chi-square goodness-of-fit test: $X^2(6) = 7028.6$, $p < 0.001$, N = 491; Figure 14) with the yard vegetation community experiencing a decrease in median condition that changed from fair (median within 11-25% dieback) to critical (median within 26-75% dieback; Wilcoxon signed-ranked test: $Z = -17.3$, N = 491, $p < 0.001$). The collective percent of individuals in excellent, good and fair conditions diminished from an 88.6% to a 20.8% (Figure 14). We also observed 16 uprooted plants (4.3%) and 41 snapped (8.4%) stems or trunks but at the time of the survey 88.7% of the remaining standing vegetation showed some of form of refoliation or sprouting.

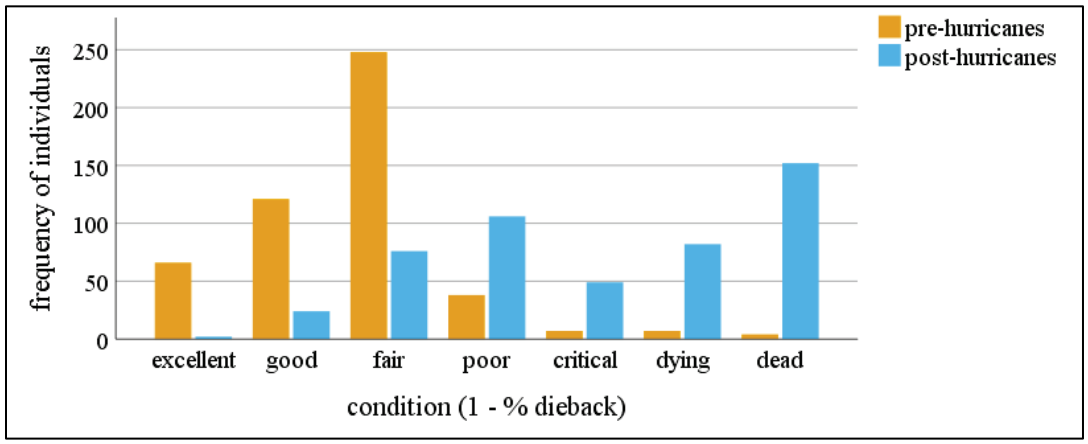

Figure 14. Frequency of individuals at each condition category pre-hurricane (orange) and post-hurricane (blue).

4.2 Hurricane-driven changes in vegetation structure and ecosystem services

When considering the pooled area of yards, the following variables exhibited observable losses: total canopy cover (from 4,758.0 m² to 3,783.4 m² or 20.5% loss), leaf area (15,585.8 m² to 12,523.7 m² or 19.6% loss), leaf biomass (1,823.2 kg to 1,419.2 kg or 22.2% loss), and basal area (from 8.9 m² to 7.8 m² or 12.4% loss). At the yard level, there were significant declines in the median values for most variables associated to

vegetation structure (Table 11A). Namely, the number of plants showed significant declines in 65.4% of yards, the number of species in 40.4% of yards, and estimated canopy cover, leaf area, and leaf biomass in 67.3% of yards. Analyses did not yield changes in the median values for DBH, plant height and basal between surveys (Table 11A).

Table 11. Results of paired sign test of estimated yard changes in structural, composition and services (N = 52). Mean value by yard was used for diameter at breast height, plant height and leaf area index.

	range pre	range post	median pre	median post	negative observs.	tied observs.	z statistic	p - value
A. Structure and composition								
# plants	1 - 42	0 - 33	6.00	5.00	34	18	-5.66	$p < 0.001$
# species	1 - 21	0 - 19	3.50	3.00	21	31	-4.36	$p < 0.001$
# native plants	0 - 11	0 - 9	0.00	0.00	3	49	-1.16	$p = 0.250$
diameter at breast height (cm)	5.2 - 80.7	0 - 80.7	10.63	10.83	13	18	1.20	$p = 0.230$
plant height (m)	1.8 – 19.00	0 – 19.00	3.84	3.87	14	19	0.70	$p = 0.486$
canopy cover (m^2)	1.3 - 546.9	0 - 430.6	59.80	42.85	35	17	-5.75	$p < 0.001$
leaf area (m^2)	4.6 - 1,713.6	0 - 1,441.9	219.45	152.45	35	17	-5.75	$p < 0.001$
leaf biomass (kg)	0.3 - 140.0	0 - 136.1	26.50	20.55	35	17	-5.75	$p < 0.001$
leaf area index	1.2 – 6.9	0 – 5.9	3.18	2.87	23	16	-1.50	$p = 0.134$
basal area (m^2)	0 - 2.9	0 - 2.5	0.00	0.00	5	47	-1.79	$p = 0.062$
B. Ecosystem services								
carbon storage (kg)	4.9 - 8,139.4	0 - 8,130.8	95.95	61.05	35	17	-5.75	$p < 0.001$
carbon sequestration (kg/year)	0.1 - 216.8	0 - 215.4	16.25	10.40	32	20	-5.48	$p < 0.001$
avoided runoff (m^2/year)	0 - 17.5	0 - 14.70	2.10	1.50	31	21	-5.39	$p < 0.001$
oxygen production (kg/year)	0.6 - 578.5	0 - 574.5	43.40	27.60	35	17	-5.75	$p < 0.001$
pollution removal (g/year)	7.4 - 2,758.2	0 - 2,320.6	353.00	245.45	35	17	-5.75	$p < 0.001$
cooling effects (kg/year)	0 - 1,064.7	0 - 1,049.0	34.28	18.23	15	36	-3.25	$p < 0.001$
food production (# plants)	0 - 39	0-27	3.00	2.00	28	24	-5.10	$p < 0.001$
ornamental (# plants)	0 - 34	0 -31	3.00	2.50	17	35	-3.88	$p < 0.001$

Collectively yards experienced losses for all ecosystem services evaluated but losses were not necessarily homogeneous across services (Table 12). For those services evaluated using i-Tree Eco, losses ranged from 9.2% to 19.9% with the largest values shown for avoided runoff and pollution removal. For services evaluated based on the loss of stems (food provision and ornamental services), losses were twice as high for food

provision. At the yard level, there were significant declines in the median values for all ecosystem services measured in this study (Table 11B). Results showed significant reduction in the median values for carbon storage, oxygen production and pollution removal in 67.3% of the yards, for carbon sequestration in 61.5%, for avoided runoff in 59.6%, for cooling energy services in 28.8% and for food production and ornamental value in 53.8% and 32.7% respectively (Table 11B). Patterns for average percent losses in ecosystem services per yard were consistent with patterns for global losses with avoided runoff and air pollution removal showing the largest reduction for services evaluated with i-Tree Eco and food production manifesting proportionally larger losses relative to ornamental services based on the % loss in the number of stems (Figure 15). We found a moderate but positive association between yard area (m^2) and proportion of change in number of species per yard ($\tau_b = 0.281$ $p = 0.008$) but not with the other structural or ecosystem services variables (all τ_b's ≤ 0.157, all p's > 0.144).

Table 12. Estimated overall loss of i-Tree Eco modeled and estimated ecosystem services.

Ecosystem service	**Pre-hurricanes** (N = 491)	**Post-hurricanes** (N = 355)	**Net loss**	**% loss**
carbon storage (kg)	29,914.8	27,165.1	2,749.7	9.2
carbon sequestration (kg/yr.)	1,836.2	1,630.1	206.1	11.2
avoided runoff (m^2/yr.)	154.9	124.1	30.8	19.9
oxygen produced (kg/yr.)	4,902.0	4,348.1	553.9	11.3
pollution removed (g/yr)	25,082.6	20,154.1	4,928.5	19.6
cooling effects (Kwh/yr)	6,081.1	5,285.6	795.5	13.1
food production (# plants)	227	132	95	41.9
ornamental value (# plants)	262	221	41	15.6

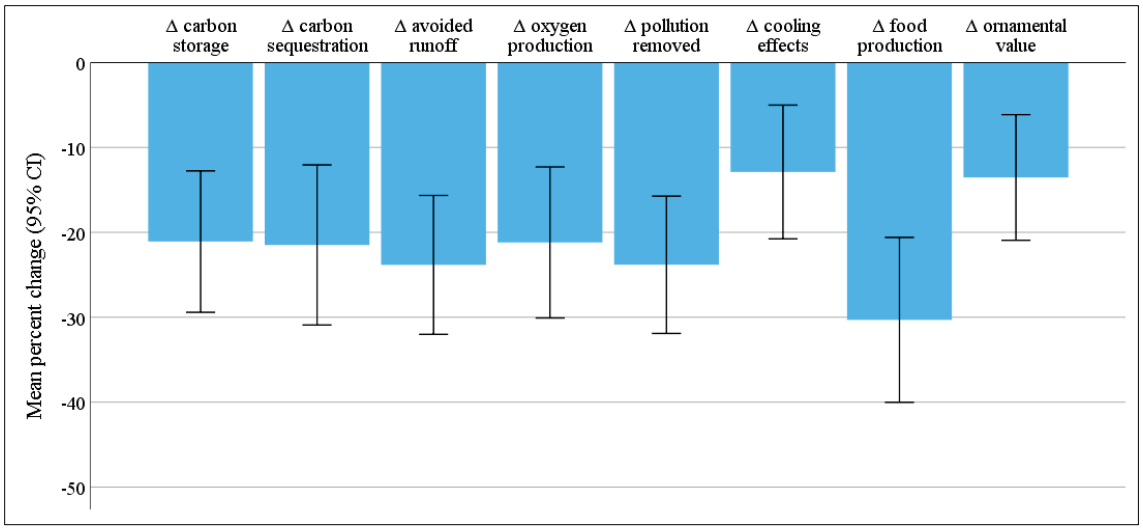

Figure 15. Average percent losses in ecosystem services in 52 yards of the Rio Piedras Watershed.

4.3 Species-specific ecosystem services and hurricane-driven changes

An evaluation of which species provided the most services (higher ESI index) prior to the hurricane events indicates that 13 of those were ranked also among the 20 most dominant species before the hurricane (Figure 16). Eleven of the top 20 service providers were also food species (*Musa x paradisiaca*, *Musa acuminata*, *Psidium guajava*, *Mangifera indica*, *Citrus aurantifolia*, *Annona muricata*, *Citrus sinensis*, *Persea americana*, *Malpighia emarginata*, *Artocarpus altilis*, *Cocos nucifera*). There is considerable variation in the provision of services among species.

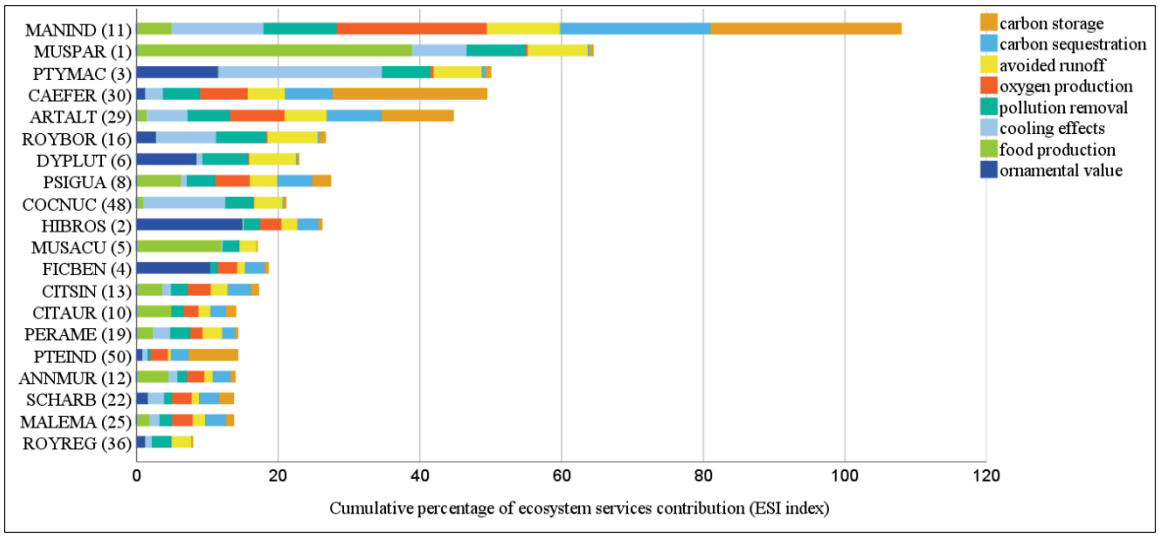

Figure 16. Comparison of the top 20 species with the highest cumulative percent contribution of multiple ecosystem services (ESI index) before the hurricane events. Species frequency ranking values in parenthesis Codes for species names are provided in Table C2).

The binomial logistic regression on the probability of mortality as a function of structural variables, previous condition categories and species yielded a model that was significant when including stem diameter, height, condition and species (Table 13), the model classified 82% of cases correctly and explained 20.2% of variation (Nagelkerke R^2), Hosmer and Lemeshow Tests indicated good fit ($p = 0.766$), the area under ROC curve was .746 (CI 0.667 to 0.825) indicating an acceptable level of discrimination. The likelihood of an individual dying decreased 0.9 times with each increase in stem diameter unit.

Table 13. Binomial logistic regression analysis on plant mortality as a function of structural variables and plant condition (reference condition = good) that generated the best fit model ($X^2(13) = 25.492$, $p = 0.20$, N = 194); The model included species interactions but none of them was significant. Only species with more than 10 individuals were included in this model. The species included were *Annona muricata*, *Citrus aurantifolia*, *Codiaeum variegatum*, *Duranta sp.*, *Dypsis lutescens*, *Ficus benjamina*, *Hibiscus rosa-sinensis*, *Mangifera indica*, *Psidium guajava* and *Ptychosperma macarthurii*. Wood density was excluded because of lack of fit.

Variable (N=194)	B	S.E.	Wald	df	Sig.	Odds Ratio	95% C.I. for Odds ratio	
							lower	upper
DBH (cm)	-0.121	0.058	4.304	1	0.038	0.886	0.79	0.993
Plant height (m)	0.322	0.196	2.715	1	0.099	1.38	0.941	2.025
Condition			3.137	2	0.208			
Condition (bad)	0.507	0.814	0.388	1	0.533	1.661	0.337	8.195
Condition pre (fair)	-0.629	0.463	1.849	1	0.174	0.533	0.215	1.32
Species code			6.425	9	0.697			
Constant	-1.985	1.197	2.751	1	0.097	0.137		

Ecosystem service losses for those species that provided the most services before the hurricane were not homogeneous among species. Overall, both *Musa acuminata* and *Musa x paradisiaca* (both food species) experienced the largest declines with ecosystem services losses of over -50.0% for carbon storage, carbon sequestration, avoided runoff, pollution removal, oxygen production and food production (Figure 17). Combined, they were the most frequent element in the vegetation of yards before the storm events and the first and second ranked species in terms of ecosystem service losses with the remaining top five consisting of species that were not as frequently used in yards (*Pterocarpus indicus, Malipigia emarginata, Roystonea regia)* (Figure 17). *Artocarpus altilis,* not as frequent as the other food species that provided the most services, was the one that proportionally lost the least in terms of number of stems relative to the other food plants. Only six of the 20 species that provided the most services before the hurricanes exhibited observable losses (> 5%) in ornamental value with half of those consisting of palm species (*Roystonea regia, Ptychosperma macarthurii*, *Dypsis lutescens)* ranked among

the top 20 most frequent species prior to the hurricane (Figure 17). In the list of the 20 species highest relative contribution multiple ecosystem services (ESI index), out of the five that presented the least amount of losses (~0%) after the hurricanes, two were among the 20-most frequent species in yards before the hurricane (*Annona muricata, Roystonea borinquena*) and the remaining three (*Artocarpus altilis,* a food species and two ornamentals (*Schefflera arboricola*, *Caesalpinia ferrea)* were not.

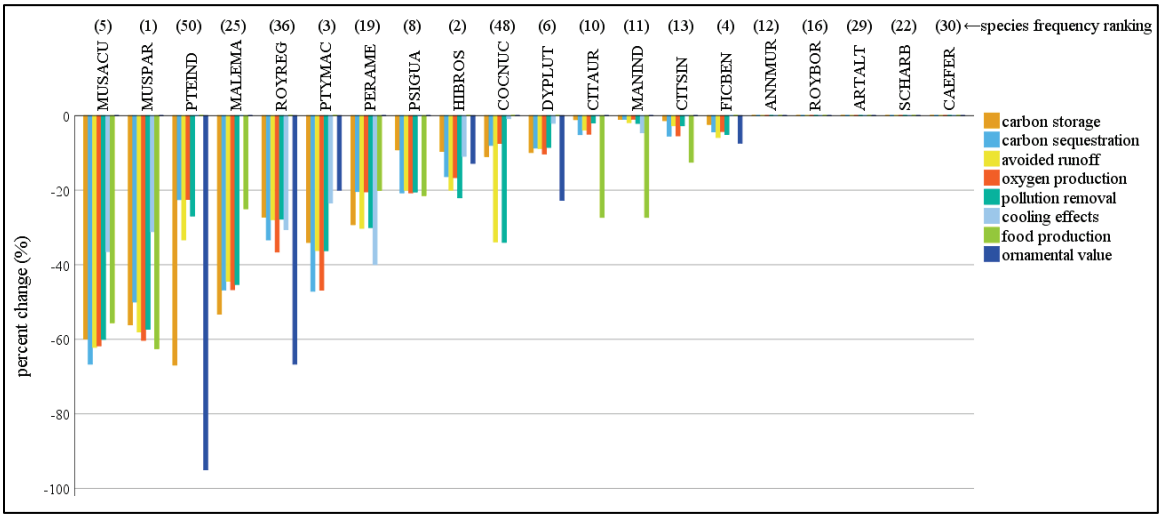

Figure 17. Percent loss of ecosystem services of 20 top contributors. Species frequency ranking values in parenthesis. Codes for species names are provided in Table C2.

5. Discussion

Given the extent of green spaces within residential land uses in urban areas (Gaston et al., 2005; Loram et al., 2008; Cook et al., 2012; Brandeis et al., 2014), yards are seen as sites that can contribute to the provision of ecosystem services in urban systems (e.g., air pollution removal, noise reduction; Calvet-Mir et al., 2012; Cook et al., 2012; Freeman et al., 2012; Goddard et al., 2010; Lubbe et al., 2011) including those that can help mitigate different phenomena associated with global changes in climate (e.g.,

the reduction of extreme climate risks to flooding, heat waves and even food security; Alves et al., 2018; Gray et al., 2014; Mason and Montalto, 2015; Norton et al., 2015). With the expected increases in extreme weather events occurrence (Méndez-Lázaro et al., 2016; Yin et al., 2018), now more than ever it is important to understand how resilient is the urban green infrastructure and associated services in the face of these events to help guide local "nature-based" initiatives, guide urban foresters or planners, and in the context of climate change adaptation and mitigation. The work presented here focused on the island of Puerto Rico, a place where hurricanes are a common natural disturbance and that most likely will experience more damaging and intense hurricanes given current models (Peduzzi et al., 2012; Gould et al., 2018; Ramos-Scharrón & Arima, 2019).

This study evaluated hurricane-driven changes in ecosystem services of the large vegetation elements in residential yards associated to the Río Piedras Watershed, the most urbanized watershed on the island and where private residential yards are known occupy a significant fraction of the total area (Lugo et al., 2011; Ramos-González, 2014). Results evidenced high plant mortality in residential yards, higher than what has been documented for forest areas following hurricanes Irma and María, likely as a result in differences in dominant species composition. Changes in ecosystem services were considerable following the 2017 hurricane events with yards losing up to 40% of their pre-hurricane capacity for some important services, particularly in air pollution removal (~20%), avoided runoff (~20%) and food production (~40%). Results showed that the large vegetation elements of yards at this site are neither equivalent in the production of ecosystem services nor are they in the loss of these services following large storm events. What follows is a discussion on how results contribute to the growing literature to how

trees growing in urban and forest areas may respond to hurricane events, what are the implications of change patters in tree cover, structural changes and ecosystem services to urban function and what are the implications of findings to green infrastructure planning for the development of sustainable and resilient green infrastructure systems within residential spaces in San Juan.

Global immediate mortality rates for yard species was approximately 31% and twice the average found for forest trees after hurricane María (15.40%; Uriarte et al., 2019). Most hurricane-mortality studies include woody species, thus a logical explanation for the elevated overall mortality observed in this study was related to the inclusion of large herb species. Specifically, the high mortality for *Musa spp.* (plantain and banana) in my sample is not surprising given their highly susceptibility to wind damage (Paull & Duerte, 2011). Nevertheless, recalculated mortality without *Musa spp.* was estimated at 21%, remaining considerably high compared to other mortality estimates. Differences in mortality rates have also been observed for hurricane Georges (Cat 3) in Puerto Rico between secondary forests (5.2%; Ostertag et al., 2005) and urban residential front and backyards (13%; Duryea et al., 2007b), but there are significant differences in design and methods between these studies. Although with mixed findings, it has been suggested that native species have higher survival to hurricanes (Duryea et al., 1996; Duryea, Kampf, & Littell, 2007; Brokaw, Crowl, et al., 2012; Duryea & Kampf, 2014), which could be a potential explanation for lower mortality rates reported in native species dominated forests but more research is needed to evaluate this in urban forests of Puerto Rico. Most surveyed yard species were non-native to Puerto Rico corroborating prior work (Meléndez-Ackerman et al., 2014; Vila-Ruiz et al., 2014), but this overdominance of

non-native plants resulted in low sample sizes for native species that prevented us from evaluating functional differences between native vs. non-native species.

Some of the documented relationships between tree species traits and hurricane-driven mortality were reflected in this study but others were not (Table 6). For example, often mortality occurs at higher rates in taller trees in forested as well as urban areas (Francis, 2000; Uriarte et al., 2019). However, we did not find a positive relationship between plant height and mortality. Variation in my model was however improved by inclusion of plant height, and was marginally significant ($p = 0.099$), which might suggest a potential relationship that is not perceivable due to my small sample size (10 spp. ≥ 10 stems, N = 164) compared to other studies (Uriarte el al. 2019: 24 spp. ≥ 40 stems, N = 1,4828; Francis 2000: 24 spp. ≥ 30 stems, N =1,076). Analyses did not detect a relationship between high wood density and tree survival which is considered to provide overall resistance to physical damage in other studies (Zimmerman et al., 1994; Francis, 2000; Uriarte et al., 2019). One possible explanation would be that lack of variation of mean wood density values for species included in the model reduced the potential explanatory power of the variable related to other variables. The range in wood density for the 10 species was 0.36 to 0.70 g/cm^3, with four species with values between 0.52 - 0.55 g/cm^3 and 3 species between 0.43 - 0.48 g/cm^3. By comparison, values of the 24 species in Uriarte et al. (2019) ranged from 0.26-0.79 g/cm^3. We did find however an inverse relationship with stem diameter and mortality that appears somewhat contradictory to findings in other hurricane studies (Table 6), but is consistent with high mortality rates observed in small-stem classes in urban vegetation (Steenberg, Millward, et al., 2019). These relationships may be influenced by other environmental variables not

considered in this study that may also influence mortality responses such as exposure to storms, differences in storms intensity, duration, rainfall patterns, frequency of recurrence (Van Bloem et al., 2006; Brokaw, Crowl, et al., 2012), type of forest, species composition and other traits that influence vegetation responses to wind exposure (Everham & Brokaw, 1996; McLaren et al., 2019). Natural and urban forests also differ in their structural traits and management regimes (Zhao et al., 2010) therefore one alternative is that in urban spaces, species may respond differently to hurricanes but long-term and larger-scale research studies are needed to evaluate these hypothesis.

Tree condition has been shown to be a good predictor of tree mortality in urban residential spaces (Koeser et al., 2013; Steenberg et al., 2019, Table 14) and an indicator of vulnerability to hurricane disturbances (Ostertag et al., 2005; Tanner et al., 2014). However, my model did not detect a significant association between tree health condition (defined by % dieback) and likelihood of mortality. A potential explanation for the inconsistency between my results and previous studies could be due to discrepancies between the way tree condition has been measured between the difference studies, some considering the occurrence of prior hurricane damage on stems and crowns (Ostertag et al., 2005; Tanner et al., 2014) or aggregated values of multiple indicators of condition (Steenberg, Millward, et al., 2019), neither which was considered in this study. Additionally, this study did not necessarily record instances of poor management (i.e., bad pruning) which can deteriorate tree condition (Steenberg, Millward, et al., 2019). Regardless, it should be noted that before the hurricane events 87% of the trees were in categories that included 0 to 25% dieback with very few considered to be in dieback categories above 25%. Thus, it is possible that there was not enough variation to test for

this effect at the time of the study. Regardless, results do point to a severe degradation in tree condition which has now become a potential a source of tree vulnerability to future hurricane disturbances that then makes the green infrastructure system of residential yards more vulnerable to future hurricane events. It should also be noted that past studies have shown that hurricane damage can result in further mortality as a result of delayed mortality manifested several years after hurricanes events (Everham & Brokaw, 1996; Lugo, 2008; Uriarte et al., 2019). If this is where the case for all hurricane events, a hypothesis is that the mortality estimate even when high, could be underestimated.

Table 14. Hurricane mortality responses of subtropical vegetation as a function of species variables. Excludes work in subtropical dry forest.

Description	System	Rationale	Examples of supporting literature	This study
stem diameter (dbh)	forest	• for some species mortality increased with stem size • larger diameter trees experience more damage	Zimmerman et al. 1994 Ostertag et al. 2005	Likelihood of mortality decreased with stem size
height (m)	forest	• large trees are particularly vulnerable to storms • taller trees experience more hurricane damage	Uriarte 2019 Ostertag et al. 2005	Not supported by this study. Relationship between high and mortality was found to be marginally significant for this sample (p=0.099)
	urban	• large trees are particularly vulnerable to storms • taller/overmatured trees are more sensitive to storm damage and mortality	Francis 2000 Steenberg et al. 2017*	
wood density (g/cm^3)	forest	• fast growing species experience greater damage by hurricanes • trees with high wood density tend to be more resistant to stem damage • high wood density species experienced low mortality • fast-growing low-density woods are more susceptible to wind damage	Ostertag et al. 2005 Canham et al., 2010 Uriarte et al., 2019 Zimmerman et al. 1995	Not supported by this study. Wood density did not add significant variation to mortality model.
	urban	• higher wood density has been found to be positively correlated with urban tree survival to hurricanes • trees with higher wood-specific gravity and greater branch flexibility are less likely to be affected by storm	Duryea et al., 2007 Francis 2000	
plant condition (varies)	forest	• structural damage due to hurricane is correlated with previous hurricane damage and past disturbance history • hurricane-damaged stems have higher mortality than undamaged stems	Ostertag et al. 2005 Tanner et al. 2014	Not supported by this study. Three categories of pre-hurricane conditions (bad, fair good) were not related to mortality
	urban	• trees in poor condition are more sensitive to stressors and disturbances and have higher rates of mortality	Steenberg et al. 2019	

Hurricane-driven changes in canopy and leaf structural traits (canopy cover, leaf area and leaf biomass) resulted in considerable changes for important ecosystem services (food provision, avoided runoff and pollution removal) that might have reduced the resilience capacity of already vulnerable urban systems and residential spaces. San Juan (and all Puerto Rico) is particularly vulnerable to food shortages because it imports almost 90% of its food supply (Muñoz-Erickson et al., 2014; Benach et al., 2019). Households located in the lowest part of the Río Piedras Watershed with higher urban

grey cover and housing density (such as Puerto Nuevo) are also located in flood hazard areas (as determined by observation of FEMA FIRM Panels) and experience moderate to high vulnerability to extreme heat events (Méndez-Lázaro et al., 2018). The lower part of the watershed is also located in a nonattainment area for sulfur dioxide SO_2, in other words bellow National Ambient Air Quality Standards (NAAQS) as defined by Clean Air Act of the United States (Sacks et al., 2018; U.S. EPA's EJSCREEN, 2019), due to local source emissions such as local power plants, high vehicle transit and ship traffic (Subramanian et al., 2018). Following hurricane María, the island experienced a shortage of food supply as airport and seaport closed, the lack of electricity led to spoil of food reserves and oil shortages limited the capacity to transport goods (Lugo, 2019). Likewise, hurricane María, and subsequent rainstorms brought record rainfall falls to the island (Ramos-Scharrón & Arima, 2019) and the deficiencies of drainage systems and abnormal rainfalls resulted in prolonged urban flooding for some communities (Lugo, 2019). Additionally, the increase in use in power generators had a significant immediate adverse effect on air quality in the San Juan Metropolitan Area, were observations for November to December 2017 (30 days) data showed that sulfur dioxide SO_2 emissions exceeded the EPA daily 1-hour threshold (75 ppb) almost 80% of the time (Subramanian et al., 2018). Given the current scenario, recovery efforts to replace the losses on ecosystem services in lower part of the watershed might be particularly useful to reduce the vulnerability of urban systems of San Juan in the face of current urban stressors and exposure to extreme disturbances.

In natural tropical ecosystems, ecological functions recover faster after a hurricane disturbance than the actual structure of the forest (Brokaw, Zimmerman, et al.,

2012). Lugo (2019) in his scientific memoir that narrates the effects of hurricanes Irma and María in Puerto Rico, emphasized on ecological recovery happening much faster than social-technological recovery. In fact, a lot of variable weather was observed after the hurricanes, small rain periods followed by lots of sun enhancing conditions for regrowth on the urban ecosystem. In general terms it has been hypothesized that the substantial amount of green cover in San Juan is a result of its moist and warm conditions which induce rapid growth of urban vegetation (Lugo & Helmer, 2004b; Muñoz-Erickson, 2014). San Juan Bay Estuary studies suggest that these urban forests are more dynamic than temperate regions and experience higher rates of mortality and in-growth rates (Tucker Lima et al., 2013). However, residential yards are privately own and subject to human selective pressures and management activities may have occurred after hurricanes crossed over the island that are not captured in this study design. In fact, of the few studies evaluating tree planting and removal decisions on urban residential yards, residents consistently self-reported concerns over tree health conditions as a one of the main reasons to remove trees (Summit & McPherson, 1998; Head & Muir, 2005; Kirkpatrick et al., 2012; Conway & Yip, 2016; Avolio et al., 2018; Guo et al., 2019). A related phenomenon for private yards is that residents may decide not to replace loss vegetation which can hinder the recovery of services of residential land uses. Therefore, the recovery of ecosystem services in residential green spaces needs to evaluate the social dimension and management decisions of residents.

5.1 Species selection and ecosystem services

More research is needed to improve urban forests species selection in residential areas that are both able to contribute to sustainable residential yards by providing multiple services, but are also resilient to extreme storm events. Generally, is recommended that maximization of potential co-benefits and focusing on multifunctionality can improve ecosystem services delivery and facilitate implementation of management strategies (Ordóñez-Barona et al., 2017; Keeler et al., 2019). Findings from his research show a lot of the species that provided more of these services are infrequent in residential yards. A limitation in my approach is that index developed in this study gives equal value to all services, but research shows that residents do not give different services the same value. For the RPWS, residents have prioritized food production, air purification, ornamental, shade and temperature reduction above other services (see Chapters 2 & 3). Additionally, species differed in their contribution but also in their losses of ecosystem services which calls for further examination how to consider species susceptibility to disturbance in ecosystem services recovery strategies. Particularly, experimental designs that take into account the ecosystem services approach and also evaluate the performance of urban vegetation under hurricanes and other urban disturbances (Yan & Yang, 2018).

Some species-specific results are also available from this study. For example, plantain and banana are an important source of food, but findings confirm they are highly susceptible to hurricanes. Their individual contribution relative to other structurally stronger species is moderate, the tradeoff is that they usually experience fast recovery and rapid fruit production (Crane et al., 2006; Díaz, 2006). Another mid-term food alternative

(fruits in four to seven years) is *Artocarpus atilis* (breadfruit) which was also a good contributor of carbon storage and carbon sequestration, has high growth rate (Francis & Lowe, 2000) and has also been recommended for shade and ornamental services (Little et al., 2001). Other small-medium trees that are mid-term food providers with moderate contribution to carbon sequestration are *Annona muricate* (soursop) which experienced no mortality, *Citrus sinensis* (valencia orange) with experienced low mortality, and *Psidium guajava* (*guava*) which experienced low-moderate mortality. For ornamental services, the shrub *Hibiscus rosa-sinensis* experienced relatively low losses of individuals but provided low to moderate contribution of carbon sequestration. These shrubs flower all year (Whistler, 2000) and 12 of 34 individuals were observed flowering less than a month after the hurricanes.

Palms are commonly recommended species for urban areas because of their cultural and aesthetic benefits, a common example is the endemic *Roystonea borinquena* (Schubert, 1979). *Roystonea borinquena* also provides an important food source to local birds and bees (Francis & Lowe, 2000; Servicio de Extensión Agrícola, 2018). Recent observation have highlighted the survival of *Roystonea borinquena* and multiple stem palms such as *Ptychosperma macarthurii* and *Dypsis lutescens*, due to their positive performance after hurricanes Irma and María events (Servicio de Extensión Agrícola, 2018; this study). However, these species do not necessarily provide as much services (e.g., air pollution removal, runoff reduction) as other species less commonly used in this study. In addition, many palm species including several species occurring at RPWS (*Roystonea borinquena*, *Ptychosperma macarthurii*, *Dypsis lutescens* and *Cocos nucifer*a) are important emitters of volatile organic compounds (Klinger et al., 2002)

which can reduce air quality in cities (Churkina et al., 2017). More research would be needed to evaluate trade-offs among ecosystem services, disservices and hurricane resistance to be able to make better recommendations.

5.2 Limitations and further research

While the occurrence of hurricanes provided an opportunity for us to evaluate the potential effects on vegetation, it limited our capacity to evaluate the extent of ecosystem services provided by yards within the Rio Piedras Watershed at a larger scale and our ability to analyze species-specific aspects of ecosystem function and their associated provision of ecosystem services. Tree inventories are an important step in managing the urban forests and modeling tools such as i-Tree Eco modeling suite are useful for urban forest management and post-disaster planning. Nevertheless, limitations are to be considered for analysis results should be considered as estimates and not as absolute values. Further research on an increased sample size is recommended and further analysis to identify if efforts invested on inventories provide the required information for decision-makers to incorporate ecosystem services to urban forest management and green infrastructure planning (Dobbs et al., 2017; Wolf, 2017). Developing a complete comprehensive inventory analysis and prioritizing management variables to reduce the intensity of field work is a priority.

The vulnerability of vegetation to hurricane disturbance is associated to a number of other factors that were not considered in this study. In natural ecosystems of the Caribbean, factors that have been found to be associated with hurricane-driven damages to trees are the hurricane size and intensity, topography, and the susceptibility of the

ecosystem to damage (Tanner et al., 2014). In the context of urban social-ecological systems, the vulnerability of a system is similarly defined by its exposure to a disturbance (defined magnitude, frequency duration and extent) and the sensibility of the system (defined by the capacity to respond to disturbance forces) (B. L. Turner et al., 2003; Steenberg et al., 2017). There is a multiplicity of variables documented for different social, ecological and spatial contexts that could influence hurricane damage to vegetation such as their interactions with the light environment, tree architecture, interactions with neighboring vegetation, potential spatial dependence between individuals, interaction with surrounding buildings, wind intensity, soil conditions and available root space, etc. (Zimmerman et al., 1994, 2014; Uriarte et al., 2004, 2019; Ostertag et al., 2005; Duryea, Kampf, & Littell, 2007; Canham et al., 2010; Staudhammer et al., 2011; Brokaw, Crowl, et al., 2012; Steenberg, Millward, et al., 2019). Further research would benefit from comprehensive social-ecological vulnerability framework analysis to synthesize recent findings in the local context of San Juan.

6. Conclusion

Evaluating and documenting the dynamics of the ecosystem services provided by urban green infrastructure can help us understand, design, plan and manage our green spaces to provide us with the functions we need to have healthy cities in a way that they are resilient and sustainable. Increasing our understanding of hurricanes effects on vegetation can inform management decisions and restoration efforts. This information is valuable to evaluate the resilience of individual tree species to these large-scale events and provides preliminary data on the potential loss of ecosystem services after hurricanes.

As Puerto Rico continues to face social and ecological challenges that can be addressed to some extent by improving the urban form, green infrastructure or nature-based solutions could provide a realistic alternative to cope with some stressors and reduce vulnerability by providing needed services to improve wellbeing and increase resilience.

Chapter 5. Conclusions and recommendations

The combined results of this dissertation research support the inherent complexity of residential social-ecological systems. Findings enrich the exiting body of knowledge for residential areas of the Río Piedras Watershed, a tropical watershed, and emphasize the role of social-ecological interactions and scale in understanding the diversity, composition, structure and function of urban vegetation. Nevertheless, there are limitations to be considered when interpreting these findings and areas of opportunity for further improvement. This work represents a snapshot of the conditions on 2011, 2014, 2016, and 2017 for certain social and biophysical characteristics. Social-ecological interactions are dynamic and therefore their outcomes need to be monitored through time. The effects of hurricanes Irma and María on residential vegetation while sudden could have long-lasting social and ecological effects not considered by this research. Notwithstanding, results do support that the social-ecological information collected in urban residential spaces could be used as tools that improve residential sustainability planning in urban settings by considering the variation in individual needs and attitudes towards green spaces and other social and economic factors of households. Only then can urban planning ensure an equitable distribution of functions that contribute to well-being, and develop sustainable goals that reflect the common visions of the residents as well as other institutional structures (Elmqvist et al., 2019). By considering social and ecological factors, data from this study can be used to facilitate the alignment of reforestation goals generated from different proposed frameworks for green infrastructure planning (e.g., species conservation, ecosystem services, climate change mitigation and adaptation) to develop resilience to urban stressors and extreme weather events in residential social-

ecological systems. It can also help guide the integration of private yards into holistic approaches of green infrastructure planning and urban forestry strategies in this city and help create more effective urban reforestation strategies by promoting an all-lands approach. Below I list a series of management possibilities and strategies to improve and expand the urban forest of the Río Piedras Watershed and other areas of the San Juan Metropolitan Area with emphasis on residential yards based on my results:

1. **Promote and diversify community-level outreach programs to raise awareness on the multiple benefits of planting trees**. Results show that residents are aware of many services provided by trees but also and that this awareness may differ across geographic areas and may be limited for important benefits for climate change adaptation and/or mitigation (i.e., runoff reduction, reducing heat island effect, natural hazard moderation, erosion control, carbon sequestration) which seem to be overlooked.
2. **In selecting species for urban reforestation programs that target residential yards, one strategy may be to seek species that optimize those ecosystem services with high residential awareness.** Results showed that residents were particularly aware of tree services such as shade provision, lowering temperature, food provision and ornamental value.
3. **In the promotion of tree plantings in residential spaces using conservation efforts may consider alternative native planting campaigns such as re-branding campaigns to highlight native origin (i.e., eco-labeling) while incorporating preferred ecosystem services (i.e., food provision, air purification, ornamental value) and creating markets for these preferred**

plants functions. Results showed that residents do value native plants and showed positive attitudes towards them which should be considered in the selection of species**.** While, more research is needed to clarify the disparities on whether native species really do better in urban environments, there are social opportunities taking place in San Juan. Research efforts that evaluate performance of different native and non-native species are high priority to improve species selection. A starting point for further research may well be selecting among most common species documented in residential areas in recent studies (Brandeis et al., 2014; Vila-Ruiz et al., 2014; Meléndez-Ackerman et al., 2018), available plants in local nurseries (Torres-Camacho et al., 2017) and non-governmental and governmental nurseries leading current reforestation efforts.

4. **In setting local green infrastructure standards (tree cover, diversity, quantity of different ecosystem services), include goals above those standards to offset potential losses following extreme weather events.** Results here showed that plant mortality in residential sites is considerable and much higher than in non-urban sites for hurricane disturbances. However, research is still limited on how urban factors may influence tree mortality (including delayed mortality following hurricane events).
5. **Improve existing resources for local stakeholders.** I recommend the improvement of tree selection information available for different stakeholders or expanding on current knowledge sharing networks so that it reaches a broader audience. For example, tree selection lists and the criteria used behind their selection is currently difficult to obtain. A great start is the new *Urban Forestry*

Manual for Puerto Rico and the U.S. Virgin Islands, but the manual could increase information on species provision of ecosystem services. The i-Tree set of tools have proven extremely useful and is recommended by numerous local institutions and planning guidelines, but specific tools that could be of use for residents plating decisions such as i-Tree Design and i-Tree Planting, are not available for Puerto Rico. More efforts are needed to increase the applicability of the tools outside continuous United States.

6. **Increase the number of trained professionals**. Many urban forestry academic findings point to the important role of arborists in providing accurate information for tree planting, but the number of currently certified arborists in Puerto Rico is small. Efforts aimed at increasing the number of trained arborists and other forestry professionals on the island could provide a significant improvement, as long as they come in hand with employment opportunities for professional practice.
7. **Goal definition and integration of institutional frameworks.** There is a substantive body of knowledge that emphasizes the need for clear definitions on concepts, frameworks and approaches. The development of future policies, plans and manuals related to San Juan green spaces, may benefit from careful examination of the definition used for green infrastructure, ecosystem services, urban forestry, nature-based solutions, etc. This includes an explicit articulation of how each concept and their components are defined (i.e., what is an urban forest, types of green infrastructure, functions or values, ecosystem services) and what specific ecological or social functions (services and/or disservices) are referenced.

Research also points out to the importance of good communications between stakeholders as well as clear definition of management goals, objectives and steps to achieve them, for effective outcomes to be achieved.

8. **Development of an urban forest comprehensive plan.** The lack of a comprehensive plan for the city of San Juan represents a challenge for effective management of the urban forest resource in this city.

Current institutional guidelines and opportunities to consider:

9. **Urban forests as priority landscapes:** The Puerto Rico Forest Action Plan (PRFAP) of 2016 defines urban areas, large and small, as priority landscapes with "*the intent of increasing the biodiversity and health of urban forests, establishing and/or maintaining, green infrastructure with all its associated benefits, and reducing tree hazards and flooding hazards that affect public safety*." For guidance, we highlight related data needs outlined in the plan and strategies provided for active and sustainable management of private urban forests:
    i. Data are needed on (1) the extent, composition, health, and restoration of urban forests (2) ecosystem services and other benefits from public and private forest land (3) disturbances affecting forests (hurricanes, floods, fires, pests, etc.)
    ii. Strategies for active and sustainable management of private urban forests should include: (1) increase capacity of communities to manage trees (i.e., promote municipal tree boards), (2) increase tree canopy cover and condition, (3) acquire community open space to protect key forested areas,

(4) hazard tree mitigation, (5) increase use of native plant material (native tree propagation and use), (6) develop educational programs, activities (i.e., demonstration forests projects), (7) develop nursery quality standards, (8) introduce agroforestry concepts and (9) promote arboriculture in university curricula.

iii. Strategies to control or reduce hurricane harmful effects should include: (1) conduct urban forest inventory, (2) develop urban forest management plan, (3) perform hazard tree mitigation (4) promote adequate tree selection.

10. **Capitalizing on urban forest inventory efforts.** For 2020, the USDA Forest Service will be conducting the Forest Inventory Analysis (FIA) which is part of a US Nationwide field-based inventory conducted every 4 years of all forest resources. The FIA Program has been incorporating urban analysis following i-Tree Eco methodology to conduct urban forest analysis and assessment of ecosystem services. The Urban FIA will be also complemented with the Urban National Landowner Survey and collecting social dimensions data related to urban forests and green space management, including that of private landowners. In Puerto Rico, the 2020 Urban FIA is scheduled to expand on the urban forest inventory conducted in 2001 and 2011 in the San Juan Bay Estuary which consisted of 108 plots (Brandeis et al., 2014), to include 200 plots and two new municipalities (Guaynabo y Bayamón). This provides an extraordinary opportunity to improve current best available science and conduct comprehensive inventory of urban forest resources that allows for the integration of social

surveys or other methods of analysis to incorporate local values into decision-making.

11. **Consider the integration of forest recovery efforts with emergency management responses following hurricane events.** Recovery efforts in Puerto Rico following the 2017 season, brought the green infrastructure concept, as a nature-based solution, outside of academic grounds and isolated case studies to a promoted approach in hurricane recovery guidelines in Puerto Rico, particularly by U.S. Federal Agencies involved in these efforts (Santiago Fink, 2018). For example, the recovery plan that established the guidelines of the recovery efforts of Puerto Rico (Puerto Rican Government, 2018) incorporates the language of these frameworks and for recovery and conservation strategies for urban forests (NCR 5), specifically identified the restoration of ecological functions and the provision of ecosystem services as potential benefits: "*Through both public and private collaborations, DNER will develop and implement landscape habitat conservation strategies to restore the function and structure of urban and rural forests, which will lessen erosion and sedimentation challenges and provide other ecosystem services, such as enhancing air quality and managing stormwater runoff*." While several important opportunities can arise from recovery efforts, their reach to private lands is however, arguably fairly limited in Puerto Rico.
12. **Consider how initiatives in residential green spaces can help meet targets of the Sustainable Development Goals of the 2030 Agenda for Sustainable Development.** Urban forests and green infrastructure can help meet specific targets of multiple Sustainable Development Goals developed by the United

Nations by providing ecosystem services for all citizens, as direct sources of food, reducing air pollutants, contribute to biodiversity conservation, and help mitigate climate change and other extreme weather events by sequestering carbon, reducing greenhouse emissions, reducing the urban heat island effects and mitigating flooding (Salbitano et al., 2016).

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Appendices

Appendix A. Supplementary materials for Chapter 2

Table A1. Survey questions included in our study.

Survey question	Measure	Range of answers
Do you prefer having on your property?	closed	yes / no
Of you answer yes. Why?	open	up to three responses
If you answered no. Why not?	open	up to three responses
In reference to trees on your property, do you see them as a benefit?	closed	yes / no
Of you answer yes. Which (benefit)?	open	up to three responses
In reference to trees on your property, do you see them as a problem?	closed	yes / no
Of you answer yes. Which (problem)?	open	up to three responses
In reference to trees on your property, do you see them as a benefit?	closed	yes / no
Of you answer yes. Which (benefit)?	open	up to three responses
In reference to trees on your property, do you see them as a problem?	closed	yes / no
Of you answer yes. Which (problem)?	open	up to three responses

Table A2. Examples of verbatim responses coding to questions of tree benefits and problems.

Item category	Verbatim responses	Code
Ecosystem services	"shade", "shade for cars"	shade
	"freshness", "they hold the heat"	lower temperature
	"provide oxygen" "without trees we cannot breathe", "lung"	oxygen production
	"they purify the air", "they reduce pollution"	air purification
	"beauty", "decoration", "visual decoration, green environment, more tropical"	aesthetic value / ornamental
	"fruits", "edibles", "nourishment"	food provision
	"animals", "attract birds"	flora & fauna habitat
Ecosystem disservices	"falling leaves", "they shed branches, leaves and seeds", "they fill the roof with leaves and cog up the roof"	maintenance hardship
	"the roots break the sidewalk", "they crack the walls"	reduced structural integrity
	"they are too high and they collide with power lines", if they grow too much is not good, power cutoffs"	power lines obstruction
	"they attract termites"	induces pests

Tables A3. Frequencies of responses of ecosystem services (Table S.3.1) and disservices (Table S.3.2)
Table S.3.1. Frequencies of open-ended explanatory responses to preference for home trees (P), positive attitudes towards home (H) and neighborhood (N) trees. Each resident had up to three responses by question. Ecosystem services are classified by type according to the Millennium Ecosystem Assessment (2005).

Ecosystem Services	**P**	**H**	**N**	**Ecosystem Services**	**P**	**H**	**N**
Regulation				*Cultural*			
shade	138	165	130	aesthetic value	80	62	97
lower temperature	132	116	106	spiritual	16	10	8
air purification	38	37	52	recreation	6	2	0
natural hazard moderation	10	6	9	likes or prefers them	8	0	0
erosion control	5	4	4	privacy	2	5	3
carbon sequestration	2	3	2	relaxation	3	3	2
noise reduction	2	2	2	family tradition	2	0	0
pollution moderation	0	0	1	improves economy	1	1	0
				tranquility	1	0	1
Support				small trees	1	0	0
oxygen production	78	68	79	comfort	1	0	0
flora & fauna habitat	16	11	10	liking to planting	1	0	0
improves environment	13	0	5	green	0	1	0
humidity	1	1	0	bird singing	0	0	1
bird nesting	1	0	0	neighbor interaction	0	0	1
help reforestation of the island	1	0	0	neighborhood well-being	0	0	1
soil fertilization	1	0	0				
soil drainage	1	0	0	*Provision*			
moisture in soil	0	1	0	food	71	154	73
ecological balance	0	1	0	increased property value	1	6	2
clean environment	0	0	1	medicine	2	0	1
				brings rainwater	1	0	2
Well-being				economic value	1	1	0
human well-being	16	13	6	saves electricity	1	0	0
safety	1	0	0	improves economy	1	0	0

Table A4. Frequencies of open-ended explanatory responses to non-preference for home trees (P), negative attitudes towards home (H) and neighborhood (N) trees. Each resident had up to three responses for home or neighborhood trees. Ecosystem disservices are classified by type according to the Döhren & Haase (2015).

Ecosystem Disservices	R	H	N	Ecosystem Disservices	R	H	N
Economic impact				*Ecological impact*			
maintenance hardship	28	51	65	induces pests	0	17	13
reduced structural integrity	12	46	27	African tulip	0	0	1
power lines obstruction	0	16	26	reduces sunlight for lower plants	0	0	1
property damage due to natural hazards	1	9	7	termites	0	1	0
car damage due to fruit drops	0	1	0				
bird droppings	0	0	1	*Other*			
				does not like trees	2	0	0
Health impact				lack of space	7	1	1
cause asthma	0	1	0	not enough space	1	0	0
humid environment for children	0	1	0	too big	1	0	0
				nowhere to plant it	1	0	0
Psychological impact				block sunlight needed to dry clothes	0	0	1
leads to neighbor disputes	1	8	9	trees inappropriate for urban areas	1	0	0
increased risk to personal injury	1	7	4	disregard for unfruitful trees	1	0	0
fruit theft	1	1	0	terrain not apt for planting	1	0	0
lowers visibility	0	2	3	too few	0	0	1
facilitates criminal activity	0	0	5	un-nested trees	0	0	1
public obstruction	0	0	1				

Appendix B. Supplementary materials for Chapter 3

Table B1. Survey questions asked to residents

Survey question	Items
Preference should be given to Puerto Rican plants over plants form other places.	strongly agree
	somewhat agree
	neither agree nor disagree
	somewhat disagree
	strongly disagree
Why?	open-ended
Would you be willing to exchange at this moment some of your non-native plants for native plants?	yes
	no
If you answer yes. What type of plant based on its form?	big trees
	small trees
	shrubs
	small herbs (includes grass and small ferns)
	big herbs
	palms
	tree fern
If you answered yes. What type of plant based on its uses?	ornamental
	food
	medical
	wood
	shade
	other
If you were given a plant, how would you prefer it?	seed
	little plant
	young plant
	mature plant (already flowered)
	cutting / slip
	other
Organize from highest (5) to lowest (1) the importance of the following plants effects:	aesthetics / beauty / ornamental
	food
	shade
	air purification
	habitat / space for wildlife

Appendix C. Supplementary materials for Chapter 4

Table C1. Frequency ranking, changes in abundance and estimated mortality of the 25 most common species.

Rank	Species	Family	Code	Pre-alive	Post-alive	Post-stand	Mortality (%)
1	*Musa x paradisiaca*	Musaceae	MUSPAR	88	30	33	65.9
2	*Hibiscus rosa-sinensis*	Malvaceae	HIBROS	39	31	34	20.5
3	*Ptychosperma macarthurii*	Arecaceae	PTYMAC	30	22	24	26.7
4	*Ficus benjamina*	Moraceae	FICBEN	27	25	25	7.4
5	*Musa acuminata*	Musaceae	MUSACU	27	12	12	55.6
6	*Dypsis lutescens*	Arecaceae	DYPLUT	22	17	17	22.7
7	*Duranta sp.*	Verbenaceae	DURSPS	17	16	16	5.9
8	*Psidium guajava*	Myrtaceae	PSIGUA	14	11	11	21.4
9	*Codiaeum variegatum*	Euphorbiaceae	CODVAR	13	11	11	15.4
10	*Citrus aurantifolia*	Rutaceae	CITAUR	9	8	8	11.1
11	*Mangifera indica*	Anacardiaceae	MANIND	11	8	8	27.3
12	*Annona muricata*	Annonaceae	ANNMUR	10	10	10	00.0
13	*Citrus sinensis*	Rutaceae	CITSIN	8	7	7	12.5
14	*Dracaena marginata*	Asparagaceae	DRAMAR	8	7	8	12.5
15	*Cajanus sp.*	Fabaceae	CAJSPS	8	0	0	100.0
16	*Roystonea borinquena*	Arecaceae	ROYBOR	7	7	7	0.0
17	*Adonidia merrillii*	Arecaceae	ADOMER	6	6	6	0.0
18	*Cestrum diurnum*	Solanaceae	CESDIU	6	6	6	0.0
19	*Persea americana*	Lauraceae	PERAME	5	4	4	20.0
20	*Averrhoa carambola*	Oxalidaceae	AVECAR	4	4	4	.00
21	*Duranta erecta*	Verbenaceae	DURERE	4	4	4	.00
22	*Schefflera arboricola*	Araliaceae	SCHARB	4	4	4	.00
23	*Tabernaemontana divaricata*	Apocynaceae	TABDIV	4	4	4	.00
24	*Clerodendrum quadriloculare*	Lamiaceae	CLEQUA	4	3	3	25.0
25	*Malpighia emarginata*	Malpighiaceae	MALEMA	4	3	3	25.0

Table C2. Table with frequencies of all individuals pre and post hurricanes and estimated mortality.

Species	Family	Code	Pre-alive	Post-alive	Mortality (%)
Acalypha wilkesiana	Euphorbiaceae	ACAWIL	1	0	100.0
Adonidia merrillii	Arecaceae	ADOMER	6	6	0.0
Allamanda blanchetii	Apocynaceae	ALLBLA	1	1	0.0
Annona muricata	Annonaceae	ANNMUR	10	10	0.0
Annona reticulata	Annonaceae	ANNRET	1	1	0.0
Araucaria heterophylla	Araucariaceae	ARAHET	1	1	0.0
Ardisia elliptica	Primulaceae	ARDELL	2	2	0.0
Ardisia solanacea	Primulaceae	ARDSOL	1	0	100.0
Artocarpus altilis	Moraceae	ARTALT	3	3	0.0
Averrhoa carambola	Oxalidaceae	AVECAR	4	4	0.0
Azadirachta indica	Meliaceae	AZAIND	1	1	0.0
Bougainvillea glabra	Nyctaginaceae	BOUGLA	2	2	0.0
Breynia disticha	Phyllanthaceae	BREDIS	1	1	0.0
Brunfelsia pauciflora	Solanaceae	BRUPAU	1	1	0.0
Caesalpinia ferrea	Fabaceae	CAEFER	3	3	0.0
Cajanus	Fabaceae	CAJSPS	8	0	100.0
Callistemon citrinus	Myrtaceae	CALCIT	2	1	50.0
Calophyllum antillanum	Calophyllaceae	CALANT	1	1	0.0
Carica papaya	Caricaceae	CARPAP	3	1	66.7
Caryota mitis	Arecaceae	CARYMIT	1	0	100.0
Cestrum diurnum	Solanaceae	CESDIU	6	6	0.0
Chrysobalanus icaco	Chrysobalanaceae	CHRICA	4	1	75.0
Chrysophyllum cainito	Sapotaceae	CHRCAI	2	0	100.0
Citharexylum spinosum	Verbenaceae	CITSPI	1	1	0.0
Citrus	Rutaceae	CITSPS	2	1	50.0
Citrus aurantifolia	Rutaceae	CITAUR	11	8	27.3
Citrus limon	Rutaceae	CITLIM	2	2	0.0
Citrus reticulata	Rutaceae	CITRET	1	1	0.0
Citrus sinensis	Rutaceae	CITSIN	8	7	12.5
Citrus x jambhiri	Rutaceae	CITJAM	1	1	0.0
Clerodendrum quadriloculare	Lamiaceae	CLEQUA	4	3	25.0
Coccoloba pubescens	Polygonaceae	COCPUB	1	0	100.0
Coccoloba uvifera	Polygonaceae	COCUVI	2	2	0.0
Cocos nucifera	Arecaceae	COCNUC	2	1	50.0
Codiaeum variegatum	Euphorbiaceae	CODVAR	13	11	15.4
Cordyline fruticosa	Asparagaceae	CORFRU	1	1	0.0
Cupressus sempervirens	Cupressaceae	CUPSEM	4	0	100.0
Cyrtostachys renda	Arecaceae	CYRREN	2	2	0.0
Dovyalis hebecarpa	Salicaceae	DOVHEB	1	1	0.0
Dracaena	Asparagaceae	DRASPS	2	2	0.0
Dracaena fragrans	Asparagaceae	DRAFRA	2	2	0.0

Dracaena marginata	Asparagaceae	DRAMAR	8	7	12.5
Dracaena reflexa	Asparagaceae	DRAREF	1	1	0.0
Duranta	Verbenaceae	DURSPS	17	16	5.9
Duranta erecta	Verbenaceae	DURERE	4	4	0.0
Dypsis lutescens	Arecaceae	DYPLUT	22	17	22.7
Euphorbia	Euphorbiaceae	EUPSPS	1	0	100.0
Ficus benjamina	Moraceae	FICBEN	27	25	7.4
Ficus carica	Moraceae	FICCAR	1	1	0.0
Ficus lyrata	Moraceae	FICLYR	1	1	0.0
Flacourtia indica	Salicaceae	FLAIND	1	1	0.0
Graptophyllum pictum	Acanthaceae	GRAPIC	1	1	0.0
Hibiscus rosa-sinensis	Malvaceae	HIBROS	39	31	20.5
Inga vera	Fabaceae	INGVER	1	0	100.0
Lagerstroemia indica	Lythraceae	LAGIND	1	1	0.0
Leea guineensis	Vitaceae	LEEGUI	4	1	75.0
Livistona	Arecaceae	LIVSPS	1	1	0.0
Malpighia emarginata	Malpighiaceae	MALEMA	4	3	25.0
Mammea americana	Calophyllaceae	MAMAME	1	1	0.0
Mangifera indica	Anacardiaceae	MANIND	11	8	27.3
Morinda citrifolia	Rubiaceae	MORCIT	3	3	0.0
Moringa oleifera	Moringaceae	MOROLE	1	1	0.0
Murraya paniculata	Rutaceae	MURPAN	3	3	0.0
Musa acuminata	Musaceae	MUSACU	27	12	55.6
Musa x paradisiaca	Musaceae	MUSPAR	88	30	65.9
Mussaenda frondosa	Rubiaceae	MUSFRO	1	1	0.0
Myrciaria floribunda	Myrtaceae	MYRFLO	1	1	0.0
Ochna serrulata	Ochnaceae	OCHSER	2	2	0.0
Persea americana	Lauraceae	PERAME	5	4	20.0
Phoenix dactylifera	Arecaceae	PHODAC	1	1	0.0
Phyllanthus acidus	Phyllanthaceae	PHYACI	2	2	0.0
Pimenta racemosa	Myrtaceae	PIMRAC	3	2	33.3
Pithecellobium dulce	Fabaceae	PITDUL	1	1	0.0
Plumeria	Apocynaceae	PLUSPS	2	1	50.0
Plumeria alba	Apocynaceae	PLUALB	1	1	0.0
Plumeria rubra	Apocynaceae	PLURUB	1	1	0.0
Polyscias guilfoylei	Araliaceae	POLGUI	1	1	0.0
Psidium guajava	Psidium guajava	PSIGUA	14	11	21.4
Pterocarpus indicus	Fabaceae	PTEIND	2	1	50.0
Ptychosperma macarthurii	Arecaceae	PTYMAC	30	22	26.7
Punica granatum	Lythraceae	PUNGRA	1	0	100.0
Roystonea borinquena	Arecaceae	ROYBOR	7	7	0.0
Roystonea regia	Arecaceae	ROYREG	3	1	66.7
Salvia	Lamiaceae	SALSPS	1	0	100.0
Schefflera	Araliaceae	SCHSPS	1	1	0.0
Schefflera arboricola	Araliaceae	SCHARB	4	4	0.0

Schinus terebinthifolius	Anacardiaceae	SCHTER	3	3	0.0
Syzygium cumini	Myrtaceae	SYZCUM	1	1	0.0
Syzygium jambos	Myrtaceae	SYZJAM	1	0	100.0
Syzygium malaccense	Myrtaceae	SYZMAL	1	1	0.0
Tabernaemontana divaricata	Apocynaceae	TABDIV	4	4	0.0
Tamarindus indica	Fabaceae	TAMIND	1	1	0.0
Tecoma stans	Bignoniaceae	TECSTA	1	1	0.0
Theobroma cacao	Malvaceae	THECAC	1	1	0.0
Thespesia grandiflora	Malvaceae	THEGRA	1	1	0.0