USING i-Tree Eco™ TO EVALUATE ECOSYSTEM SERVICES FOLLOWING FLOODPLAIN BUYOUTS

By

PETER FAHEY JR.

Bachelor of Environmental Science, Biological Sciences, 2019
University of Connecticut
Storrs, Connecticut

Submitted to the Graduate Faculty of the
College of Science and Engineering
Texas Christian University
in partial fulfillment of the requirements
for the degree of

Master of Science

August 2024

USING i-Tree Eco™ TO EVALUATE ECOSYSTEM SERVICES FOLLOWING FLOODPLAIN BUYOUTS

By

Peter Fahey Jr.

Thesis approved:

____

Major Professor

____

Esayas Gebremichael

____

____

____

For The College of Science and Engineering

Acknowledgments

I would like to thank the Texas Christian University College of Science and Engineering and the Department of Environmental and Sustainability Sciences for providing the resources and funding for this research.

Thank you to the faculty of the Department of Environmental and Sustainability Sciences for the guidance and wisdom along the way. Every faculty member was eager to provide me with encouragement, offer challenges, and of course, provide critiques. Each of you has made me into a more well-rounded environmental scientist and I am eternally grateful. Special thanks to Professor Becky Johnson, Professor Peter McKone and Dr. Rhiannon Mayne. Professor Johnson's environmental compliance class was certainly tough, but it taught me so much about the way business and organizations respond to environmental regulations. I also watched Professor Johnson's dog Macy a few times, and some snuggles were greatly appreciated as I was missing my dog Bertie back home in New York. Professor McKone provided invaluable real-world applications in our Wetlands Delineation and Management and Wetlands and Water Law classes. They were also some of my favorite classes I took at TCU. When it felt like this thesis was impossible to complete, Dr. Mayne reviewed my work, acknowledged where I slipped up, and helped me recenter myself to continue striving towards completing this study. Thank you all for all the help along the way.

I would also like to thank my committee members, Dr. Brendan Lavy, Dr. Michael Slattery, and Dr. Esayas Gebremichael. I have sincerely appreciated my relationship with each of you throughout my graduate education. You have all been instrumental in my success, and I would not have been able to complete this work without your support. Special thanks to Dr. Elyse Zavar as well for her expertise in buyout literature. Her research inspired me to pursue this project focused on buyout landscapes as it combined my interests of urban forestry and the management of environmental hazards and disasters.

Dr. Lavy, I cannot express how much you have helped me throughout my entire graduate education. After a brief email exchange where I showed my interest in the graduate program, we hopped on a call and discussed sustainability, potential research ideas, and went over some research you had recently worked on. From then on, I have relied on you for nearly everything.

This study truly would not have been completed without you. You could have given up on me in each instance where I gave up on myself, and we both know there were plenty, but you continued to tell me that I'm smart, I'm capable, and I'm deserving of this degree. Thank you so much for everything. Our lunch trips to Tropical Smoothie to talk about life, stress about due dates, or mull over my thesis are some of my fondest memories at TCU.

I would also like to thank the City of Arlington for allowing me to measure the trees in the urban buyout matrix. And of course, to the residents, who watched me lug around equipment and stopped to inquire about what I was doing hugging trees in their park. I would also like to thank Kenna Mollendor for her assistance in collecting field data. We were measuring trees in over 100-degree weather under the blazing Texas sun, and she continually assisted me with no complaints. I was able to collect so much more data with her assistance in the field.

Finally, I would like to thank my family and friends. Thank you to my mom, dad, and sisters Cheryl and Melissa for your unconditional support throughout all my endeavors. Special mention for my rescue Boston terrier, Kendra, for providing me with much-needed walk breaks and cheering me up when I felt down. Also, thanks to my dog Bertie at home in

New York. My friends have cheered me on every step of the way, thank you roast, I appreciate you.

Table of Contents

List of Figures

List of Tables

1. Introduction

1.1 Background

Municipalities are increasingly participating in floodplain buyout programs to reduce the risk and severity of flood damages (Zavar, 2015). Since the late 1980s, the U.S. federal government has supported over 43,000 property buyouts in response to flooding events (Mach, 2019). An increasing number of residential properties have experienced severe, repetitive flood damages since 2000, further emphasizing the importance of buyout programs to reduce future losses and ensure public safety (Sheppard, 2021). Buyout programs also present a unique opportunity within the urban floodplain, allowing program managers and residents to adopt a myriad of land uses following the acquisition of these parcels (Zavar, 2016). For example, municipalities have converted buyout landscapes to parks, adding playgrounds or trails to the area after acquisition. In other places, program managers have engaged in ecological restoration efforts to renew lost ecosystems (Atoba et al., 2020). These types of active land management strategies are encouraged by the Federal Emergency Management Agency (FEMA) and often provide vital social and ecological benefits to the surrounding area (FEMA, 1998). Municipalities have also taken more passive land management approaches after acquisition. It is common for buyout properties to remain vacant with minimal maintenance (Zavar & Hagelman, 2016). While vacant landscapes provide ecosystem services, more active management strategies, such as greening the landscape, could maximize these important benefits (Kim, 2016). Therefore, buyout sites can provide a multi-faceted approach to disaster recovery, where managers can remove residential structures from the landscape and use active management strategies like planting trees to improve the ecological importance of the urban forest through the provisioning of more ecosystem services.

A properly managed urban forest provides several benefits that align with established ecosystem services. Urban trees provide a wealth of value to communities, as they contribute to stormwater attenuation (Berland et al., 2017), improved water quality (Livesley et al., 2016), air pollution removal (Wu et al., 2019), oxygen production (Nowak et al., 2007), carbon storage and sequestration (Davies et al., 2011; McPherson, 1999), urban heat island reduction (Edmondson et al., 2016), and serve as cultural and spiritual icons (Hirokawa, 2011). Therefore, urban trees can serve as a proxy for empirical observations and quantification of ecosystem services (McPhearson et al., 2013). Quantifying ecosystem services is a useful tool to determine which, if any, ecosystem services are being provided by the landscape. This information can be used to highlight areas that already excel at or could benefit from improved provisioning of ecosystem services. For example, at buyout sites, avoided runoff would likely be one of the main ecosystem services prioritized by land managers in hopes of limiting future flooding events. Therefore, quantifying the urban forests' contribution to avoided runoff could identify tree species, or certain areas of the urban forest that could benefit from greater rates of avoided runoff through the urban forest.

Trees have been well studied for their contributions to the local environment, and individual tree species exhibit different rates of ecosystem services. It is important to note that one single species is often not sufficient in providing multiple ecosystem services, as trade-offs may exist between different services. These trade-offs have recently been explored as there is an increased interest in ecosystem services, though the mechanisms associated with them are complex and research on this topic is nascent (Felipe-Lucia et al., 2018).

Researchers have identified that managing forests to increase structural heterogeneity, maintain and preserve large trees, and allow for gaps in the canopy has the potential to counteract these trade-offs and promote several ecosystem services (Felipe-Lucia et al., 2018). Furthermore, research has found that forest stands with a higher species richness have greater provisioning of ecosystem services (Gamfeldt et al., 2013). Municipalities with thriving urban forests can expect greater benefits from them, so maximizing the potential of the urban forest would yield increases in the aforementioned ecosystem services. Therefore, it is important to maintain species diversity and provide ample space for tree growth within the urban forest to promote its overall health.

Urban environments tend to complicate the evaluation of ecosystem services due to their unique dynamics. These forests are constantly changing, and their trees often face different stressors than those they have evolved to withstand, such as the inadvertent introduction of nonnative pests and diseases (Alberti et al., 2003). Additionally, data collection in urban environments is hindered by limited access, funding, and time. One tool that can be used to simplify the quantification of ecosystem services provided by urban forests is i-Tree Eco™. The i-Tree Eco™ tool is the premier, peer reviewed tool, designed to quantify ecosystem services of urban and rural forests. It was created in partnership with the U.S. Forest Service as a public access tool to strengthen forest management and advocacy efforts. This program broadly allows for the estimation of urban forest structure, pollution reduction, public health impacts, carbon storage and sequestration, avoided runoff, energy effects, forecasting, bio emissions, pest impacts, and quantitative values (i-Tree, 2021b). Quantification of these services relies on ground-level tree measurements recorded in the field, providing more accurate estimations than similar programs (Nowak, 2008). Since its initial creation in 2005, i-Tree^{TM} has been used in many studies, largely focused in urban areas. This program is available for use worldwide and has previously been used to evaluate the urban forests of entire countries (Monteiro et al., 2019). Furthermore, i-Tree^{TM} has been used to determine ecosystem services at the city and local scale (Riondato et al., 2020). In Texas, many of the largest cities, including Houston, Dallas, Austin, and El Paso have conducted i-Tree Eco^{TM} analyses in the past (Foundation, 2021). Smaller municipalities like Arlington, Plano, Denton, and Argyle have also conducted i-Tree Eco^{TM} analyses (Barker et al., 2016; Pace & Kralik, 2014). While these analyses provide general information to city officials about urban forests at a greater spatial scale, limiting the study area to a single urban park system promises to identify more localized community effects that may influence ecosystem services. Such effects might be otherwise overlooked in broad-scale evaluations. For example, a broad understanding of canopy cover and structure across the city may not necessarily represent the urban forest's composition in a smaller park system. More localized analyses of ecosystem services allow land managers to make more well-informed decisions about the urban forest systems they manage.

Yet, the ecosystem services provided by urban trees might not be the only benefits of buyouts programs for the community. These sites are required to be maintained as public green spaces, providing residents with access to nature and allowing for increased oversight by park managers. A systematic review of urban green spaces and human well-being determined that both the number of green spaces and their vegetation cover improved well-being, particularly in terms of mental health and increased social opportunities (Reyes-Riveros et al., 2021). Conversely, researchers have found that while the number of green spaces is important, the accessibility and quality of urban green spaces significantly contribute to neighborhood satisfaction (Zhang et al., 2017). This indicates that while maximizing the landscape in terms of ecological productivity is important, providing opportunities for recreation and other land uses is essential to ensure that the space is being utilized by residents.

In addition to promoting public health, these open landscapes provide an opportunity to plant more trees as well. As trees contribute to several important processes within the landscape, planting more trees may have the potential to increase the amount of provided ecosystem services. An increase in the number of trees within the urban landscape will yield increases in the provisioning of ecosystem services if effectively managed and maintained (Sousa-Silva et al., 2023). Yet, options for increasing the number of trees in urban areas are limited. Much of the urban forest is located on private properties (McPherson, 1998; Pearce et al., 2013), where property owners dictate tree planting and removal based on their preferences (Conway, 2016; Lavy & Hagelman, 2017). This leaves municipalities with limited space to increase the urban forest and provide maintenance of these areas. Some urban municipalities have begun to acknowledge the lack of space they manage by exploring tree planting on private property in addition to public lands as a necessity to reach urban canopy goals (Morgan & Ries, 2022). Large-scale planting projects have been launched in major urban areas across the United States, including New York's MillionTrees initiative, Los Angeles' City Plants program, and Houston's plan to plant 4.6 million trees by 2030 as a part of their Climate Action Plan (Garrison, 2017; McPherson, 2014). These planting projects typically aim to increase the city's total canopy cover by lining city streets with trees to improve water quality and reduce negative effects of air pollution and heat stress on human health. On a localized scale, communities may implement tree planting efforts, as well as tree preservation ordinances, to manage, preserve, and grow their urban forests.

1.2 Problem statement

While the contribution of trees in urban forests have been well studied, less is known about the trees that remain at sites following buyout programs, as most research on buyout programs is centered on the social and economic aspects of property acquisition (Bendor et al., 2020; Curran-Groome et al., 2021; Loughran & Elliott, 2019). Little research has looked at the potential of buyout landscapes to contribute to ecosystem services of the surrounding area (Greer et al. 2021). Among the studies evaluating the ecological potential of floodplain buyouts, only a few have been published in peer-reviewed journals. One such peer-reviewed study identified 11,000 km^2 of land in coastal California that could benefit from a home buyout program followed by habitat restoration to derive desirable social, environmental, and economic benefits (Calil et al., 2015). Another study found that the creation and protection of open space is instrumental in reducing property damage associated with flooding events (Brody & Highfield, 2013). Recent studies have advocated for the consideration of strategic property buyouts that emphasize the ecological potential of these landscapes, as well as a myriad of economic and social benefits of buying out flood-prone vacant landscapes before they are ever developed (Atoba et al., 2021; Atoba et al., 2020). As previously mentioned, there has been an increasing prominence of tree planting programs in recent years, so buyout sites may prove to be an ideal environment for tree planting to provide more benefits to communities that have endured repetitive flooding events.

1.3 Purpose statement

Further insight into the contributions of urban trees to buyout landscapes and their potential to increase ecosystem services promises to inform research and management practices related to buyouts. The purpose of this research is to understand the extent to which an active management approach focused on increasing the number of trees across a buyout landscape would maximize ecosystem services. It is hypothesized that buyout sites are not utilizing all the available green space, indicating there may be an opportunity to improve species diversity and provided benefits through the addition of more trees and increasing the total canopy cover of the landscape. The aim of this research is to determine the potential of buyout landscapes to contribute urban ecosystem services through the implementation of tree planting programs. In addition, make recommendations for those managing buyout sites to maximize the use of the landscape through tree planting initiatives and the active maintenance of urban green spaces.

1.4 Research questions and objectives

This research aims to address the following questions:

1. What urban tree related ecosystem services are associated with buyout sites?
2. What extent of the buyout matrix is composed of available planting space, and can we plant trees within the matrix to effectively increase the output of ecosystem services?
3. What method of tree planting yields the greatest ecosystem services, the 10-20-30 rule for urban forestry or a mixed composition of the 10 most prevalent species already found in the landscape?

This research aims to address these research questions with three main objectives:

1. To identify and quantify the ecosystem services provided by the urban trees within the buyout matrix.
2. To determine the available planting space within the buyout matrix.
3. To compare which tree planting method yields the most ecosystem services, and what percentage of the available planting space needs to be afforested to yield the most ecosystem services.

2. Material and methods

The following sections will describe the field site, explain the methods for measurement and collection of data, the use of the i-Tree Eco™ software, as well as geospatial and statistical analyses in ArcGIS Pro and IBM SPSS.

2.1 Site and situation

This study focuses on a buyout program in Arlington, Texas, along Rush Creek (N 32.686983, W -97.1774785; Fig. 1). Prior to 2011, the site consisted of 23.5 hectares of residential homes and condominium complexes. According to the National Oceanic and Atmospheric Administration (NOAA), in September of 2010, Tropical Storm Hermine produced 40 cm of rain in the area, roughly half of Arlington's annual precipitation (NOAA, n.d.). Following the storm, the City of Arlington approved the Rush Creek Property Acquisition project to relocate individuals and remove affected homes from the floodplain to limit the risk associated with future flood damages (FEMA, 2022). The city purchased 49 residential properties and 14 condominium complexes (Naturally Resilient Communities, n.d.). The city had previously been granted $2 million by the Federal Emergency Management Agency to purchase five single-family homes along the creek in 2008. The city incorporated all properties into a contiguous urban park system, adding 9.5 hectares of land for a total park area of 33 hectares. The landscape includes open space, a tennis court, two playgrounds, two dog parks, and remaining residential structures the city was unable to acquire as part of the buyout program. The city maintains the park area by mowing and pruning existing trees. Site visits suggest limited active tree planting except for five small (diameter at breast height < 12 cm) irrigated trees in the northern portion of the study area. Existing trees reflect historical legacies of past residents with more vegetation in backyards of buyout parcels and the addition of some ornamental species in front yards (Locke et al., 2018). For the purposes of this study, I consider periodical mowing as a passive management strategy as it does not consider natural regeneration of the forest, health of the canopy, or improving the landscape's efficiency.

Figure 1: Map of the study area in Arlington, Texas. An inset map shows the distribution of buyout parcels within the landscape.

Arlington, Texas has a humid subtropical climate, with an annual average high temperature of 23.8°C, an annual average low temperature of 13.3°C, and annual precipitation of roughly 1,000 mm (NOAA, n.d.).This region is defined by the United States Forest Service (USFS) as the South Central climate region, and the South tree growth zone, with a mean number of freeze-free days each year of at least 240 days (McPherson, 1999). The city of Arlington is located within the Dallas-Fort Worth (DFW) Metroplex, roughly fifteen miles east of Fort Worth. The DFW Metroplex has experienced a 18% increase in population from 2010 to 2018 alone, driving increased residential development on flood prone areas (Lee, 2021). In addition, Texas is subjected to the most flood-related damages and fatalities out of every state (Brody et al., 2008). These factors make Arlington a suitable site for the evaluation of a buyout program, as the increased pressures of urban sprawl will cause increased flood damages. Therefore, existing and future developments must be met with sustainable and ethical solutions to protect the local environment and community members that reside there.

Furthermore, Arlington is a particularly interesting case, as the city has a history of urban forest management practices. The city has earned the distinction of Tree City USA by the Arbor Day Foundation in cooperation with the Forest Service for the past 23 years. This program requires cities to maintain a tree board, have a community tree ordinance, spend at least $2 per capita on urban forestry, and celebrate Arbor Day. As a part of their annual Arbor Day celebrations, the city gives out free trees to residents. Additionally, Arlington, Texas has had a tree preservation ordinance in place since 1987 to maintain large, native trees in the city. Furthering their initial tree preservation ordinances, the city of Arlington announced a 30,000-tree goal to expand their urban canopy, achieving 80% of this goal by 2020. While setting ambitious tree abundance goals can be effective, a lack of clear objectives hinders the success of these efforts. Establishing long-term management strategies, such as evaluating canopy cover, species diversity, and tracking individual gains and losses of trees increases the success rate of tree planting programs (Sousa-Silva et al., 2023).

2.2 Field measurements

I measured individual trees at buyout sites surrounding Rush Creek in Arlington, Texas from May 10th to September 17th, 2022. The Arlington Parks and Recreation Department granted me access to measure the trees located on the buyout sites. For the purposes of this study, a tree is defined as any woody plant with a diameter at breast height (DBH) of at least 12.7 cm (5 inches). These parameters were defined to prevent inclusion of woody stems that are not already well-established trees. For each tree, species, DBH, tree height, crown base height, crown width, crown light exposure, percent crown missing, crown condition, and geographic coordinates were recorded. These values were either directly, indirectly, or conditionally used to derive ecosystem services. Each tree was assigned a unique ID number to maintain organization of data while in the field. A total of 359 trees were identified, measured, and inventoried within the buyout matrix during the field season. The procedures followed during the field investigation are outlined below in detail.

First, geographic coordinates were recorded for each tree with a portable Trimble Catalyst Global Positioning System (GPS) unit capable of sub-foot accuracy. I mounted the Trimble Catalyst GPS unit on a 2-meter range pole to standardize the measurement height. Coordinates were recorded as close to the base of the tree as possible, with the rod firmly planted at the base of each tree. Location was only recorded coordinates when the GPS unit showed to be within 12.7 cm (5 inch) accuracy. Trimble Catalyst has a mobile manager app which integrates with the ArcGIS Online mobile application to directly upload these data points to a field map. In addition to the field map, kept a record of each tree's coordinates in a field book to verify the locations were accurate once plotted.

In tandem with obtaining each tree's location, species information of each tree was also recorded. In instances where it was difficult to discern the species, pictures of the leaves, acorns, crown structure, flowers, or other prominent characteristics were taken for later review. For example, within this landscape, some of the oak trees appeared to have experienced hybridization. It has been long well known that oak trees exhibit significant hybridization between species, complicating species identification (Palmer, 1948). Oak trees that could not be readily identified to the species level were categorized with best judgment based on optimal examples of each species known to be found within the landscape or within this growing region.

Diameter at breast height (DBH) was measured at the standard height of 1.3 meters (4 feet 5 inches) above ground. For multi-stemmed trees, DBH was recorded for each stem greater than 12.7 cm. In cases where DBH could not be measured at standard height, measurements were taken as close to the standard height of 1.3 meters as possible and the height at which DBH was measured was recorded in my field journal. For instance, a few trees had splits or galls occurring exactly at standard height, which would either make measurements impossible, or not representative of the actual tree's characteristics. If a tree were to split below standard DBH height, it was recorded as two individual stems. If a tree were to split above standard height, it was recorded as a single stem. In addition, urban trees have interesting and complex growing patterns, so many understory trees have curved bases. In those cases, to more accurately measure DBH, a measuring tape was placed at the base of the tree and followed the shape of the tree to the equivalent of 1.3 meters. DBH was then recorded at 1.3 meters from the base aligned with the tree shape.

Tree height was measured with the three-point measurement mode of the Nikon Forestry Pro II Laser Rangefinder. The rangefinder uses three measured angles to evaluate total tree height using triangulation. The first angle was taken at the viewer's natural eye height looking straight ahead. The next angle was taken at the ultimate point of the tree's canopy, followed by the third angle being taken at the base of the tree. These measurements were taken from at least 10 meters from the tree to ensure accuracy of the rangefinder. Sections of unmanaged urban canopy can grow quite dense, so where the top of the crown was not easily visible, measurements were taken as visually close to the top of the crown as possible.

All crown measurements were taken from the ground. Crown width was measured along two transects from north to south and east to west. A compass was used to properly follow these transects. A 100 foot (30 meter) measuring tape was staked into the ground at the edge of the canopy along both transects and measurements were recorded to a tenth of a foot accuracy.

Next, the condition of each tree's canopy was observed and recorded. I visually evaluated the percent of crown composed of dieback as well as the percent of the crown that was missing. These characteristics were measured following the standard methodology for visual estimations within i-Tree Eco's user manual (i-Tree, 2021a). For the purposes of this study, a range of 5% was given for these estimations. Only dead sections of the tree were included in the estimation of percent dieback. Determining the percentage of the missing crown was more intuitive. Essentially, the trees were first imagined to be in perfect condition for their size. Then, while looking at the current condition of the tree, it was estimated how much of the tree was missing from the optimal canopy.

Crown light exposure was based on how many sides of the tree receive sunlight from above, for a maximum of five total sides. The tree was viewed from a “birds-eye” view to mimic where sunlight would reach the canopy. The top was considered one side, as well as the four quadrants between the aforementioned transect lines. If it appeared that most of the side would receive sunlight in an average day, that was counted as one side. Therefore, trees in total sunlight throughout an average day would be a “5”, whereas a tree whose canopy is completely understory would be a “0”.

Additional comments were included during the field collection stage for several trees. For example, several sites contained remnants of residential structures or features that were not fully removed during the buyout process. In addition, a few trees even showed clear signs of former residential use, including metal attachments for bird feeders, PVC piping and concrete. Any other factors that may have influenced the data collection protocol were also recorded in the comments section for future references. All comments were reviewed before the field season ended to ensure all necessary trees were revisited and to make any corrections.

2.3 i-Tree Eco™

To determine the current ecosystem services provided by trees on the buyout landscape and model future scenarios under more active management practices, I used i-Tree Eco™, a widely used publicly available software. i-Tree Eco™ is a software application that utilizes random plot sampling to quantify forest structure, environmental effects, and value to communities using a series of equations and algorithms. The quantitative values derived from i-Tree Eco™ estimates can be used for comparative statistical analysis. Table 1 gives operational variables to be evaluated to address the conceptual variables.

Table 1. Conceptual and operation variables

Ecosystem Services	*Units*	*Source*
Carbon storage	kg C	i-Tree Eco
Gross carbon sequestration	kg C	
Air pollution removal	g/m2/yr	
Avoided runoff	cm3	
Total annual value	$/yr (US Dollars)	
Replacement value	$ (US Dollars)	
Available planting space	m2	Classified Land Cover Data

For this study, i-Tree Eco™ was used to quantify the ecosystem services provided by the trees in the buyout matrix, as well as quantifying different tree planting scenarios. This software integrates field measurements with air quality and hourly weather data to produce quantitative reports on ecosystem services. This software allows the user to quantitatively assess rates of avoided runoff, air pollution removal, and carbon storage and sequestration by trees in the study area (i-Tree 2021). First, all the trees were imported into i-Tree Eco™ for the individual evaluation of ecosystem services, with the software providing a quantitative report with approximate values and an estimation of error. To provide accurate estimations of local weather and air pollution effects, i-Tree Eco™ required local weather and hourly atmospheric pollutant data for a year. I collected available weather data from the weather station at Arlington Municipal Airport for 2017, as it was the most recent and extensive year with average precipitation for Arlington within the possible i-Tree Eco selections. An average precipitation year was used to establish a baseline for the urban forest and minimize the impact that fluctuations in rainfall may have on the provisioning of ecosystem services each year. For air pollution, I acquired data for six prominent pollutants, including carbon monoxide, ozone, nitrogen dioxide, sulfur dioxide, and particulate matter (PM), $PM_{2.5}$, and PM_{10} from various recording stations ranging from 9 km to 33 km from the field site. These stations were selected based on their proximity to the study area. Once the field inventory was entered into i-Tree Eco the quantitative report was run.

2.4 Analysis

To compare ecosystem services within the study area, geospatial and statistical analyses were conducted. Tree measurements were uploaded to i-Tree Eco™, and the ecosystem services associated with each tree in the study area were calculated. These estimations provided a baseline for the ecosystem services currently provided in the study area and establish tree planting scenarios to determine changes in provided ecosystem services. I analyzed four planting scenarios to determine which, if any, would significantly increase existing ecosystem services. To accomplish this task, I created a geographic information system (GIS) of the tree inventory and determined the available planting space, using ArcGIS Pro 2.4 (ESRI 2019). I used a classified land cover layer (Halff Associates Inc., 2022) with six land cover classifications (i.e., water, tree canopy, low vegetation, barren, impervious surfaces, and impervious roads) to calculate available planting space. Within the buyout matrix, only barren land within the buyout properties and surrounding urban park land was considered as available planting space. Available planting space was calculated by summing all permeable land within the landscape. This reflects any land that was not already forested or covered by impermeable surfaces. During field surveys, I did not observe any further obstructions to the landscape, such as telephone wires or other structures, that would further limit the available planting space.

Next, I determined the number of trees that could be planted in the available space on the buyout parcels. After removing outliers, I used the median crown width of existing trees to create a hexagonal tessellation with equal sides of 5.34 m (area = 74 m^2) to represent the average tree crown across the entire landscape (Fig. 2). If the centroid of the hexagonal overlay was over forested canopy and the canopy visually covered at least 50% of the area of the hexagon, the area was deemed not suitable for planting. Similarly, if the centroid of the hexagon landed outside of the study area this space was not selected for planting. I used these selection rules to ensure that each proposed tree would have the appropriate space necessary for growth and park managers would be able to feasibly plant these trees within the landscape. From these available planting space estimates, I modeled four tree-planting scenarios to maximize ecosystem services.

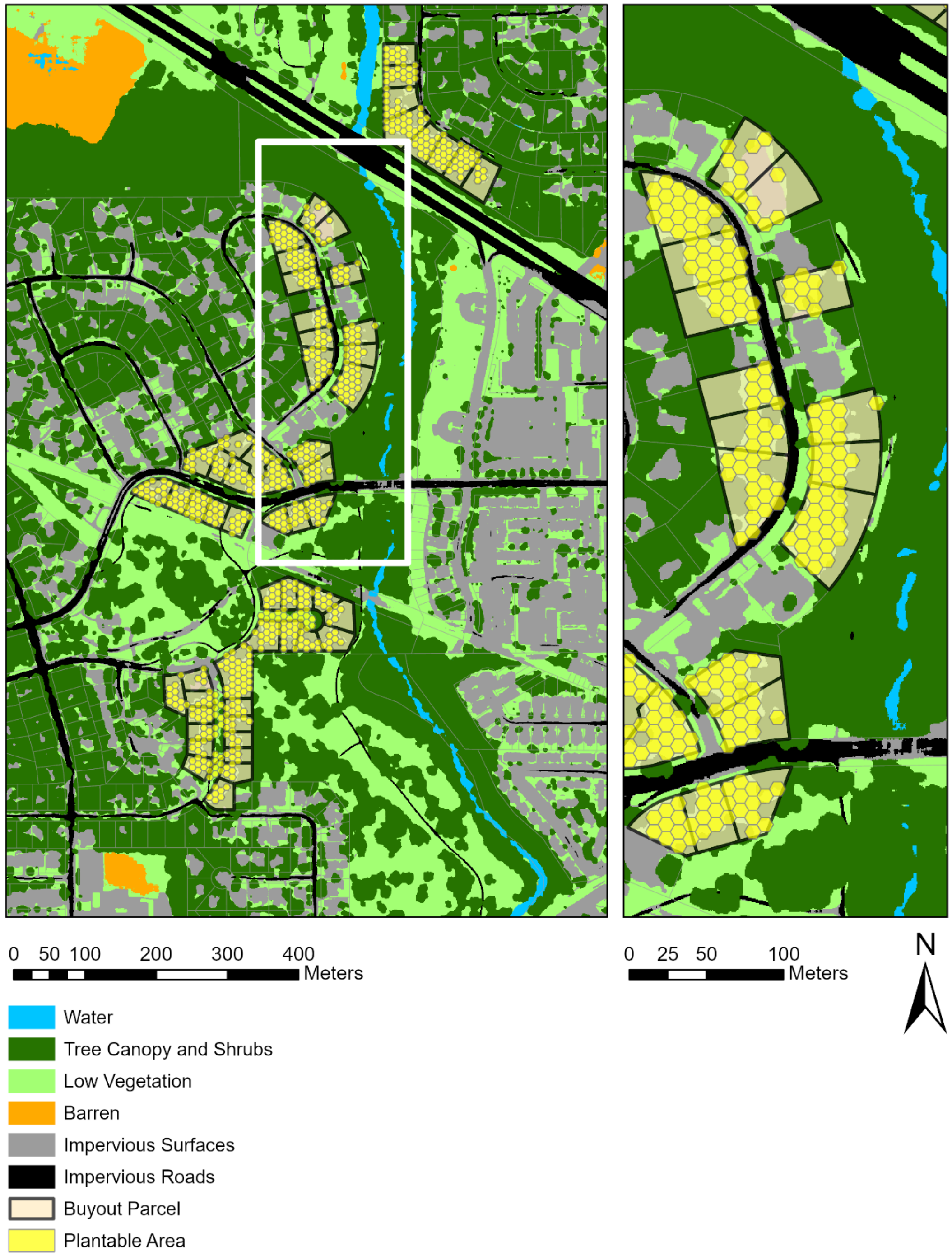

Figure 2: Map depicting the spatial determination of available planting space using hexagonal tessellation on ArcGIS Pro.

Based on the available planting space and existing urban forest composition, I modeled four tree planting scenarios to determine the number of trees that would be needed to provide a significant increase in ecosystem services. This was done twice under two different planting conditions: four scenarios following a top 10 most prevalent species approach, and four scenarios modeled on Santamour's 10-20-30 rule for tree planting (Santamour, 1990). For the top 10 approach, I based each scenario's proposed tree plantings on the calculated mean crown spread, tree height, DBH, and canopy condition of those observed within the landscape. For the 10-20-30 scenarios, only 10% of any given species was added to the landscape. Under both approaches, the proposed tree planting under each scenario was combined with the observed tree inventory to determine what the existing landscape would look like under more forested conditions (Table 2).

Table 2. Table of the trees required for each planting scenario. These trees are added to the existing park matrix during i-Tree Eco™ analysis.

Tree species	***Scenario 1 - 25%***	***Scenario 2 - 50%***	***Scenario 3 - 75%***	***Scenario 4 - 100%***
Ulmus americana	12	24	36	48
Quercus buckleyi	8	16	24	32
Quercus macrocarpa	4	8	12	16
Ulmus crassifolia	35	69	105	139
Quercus virginiana	27	54	81	108
Pinus taeda	6	12	18	24
Carya illinoinensis	37	74	111	148
Quercus stellata	23	46	69	92
Celtis laevigata	9	18	27	36
Quercus alba	7	14	21	28
Total	168	336	504	671

3. Results

3.1 Descriptives

I measured and inventoried 359 trees across 29 species from May 17 to September 25 of 2022 (Fig. 3). The three most common species in the study area were *Carya illinoinensis* (18.6%), *Ulmus crassifolia* (15.9%), and *Quercus virginiana* (14.8%). Of the 29 identified species, only two species, *Morus alba* and *Pistacia chinensis*, were nonnative to Texas. In addition to being introduced species, both *Morus alba* and *Pistacia chinensis* are listed as invasive species by the state of Texas as well. These invasive trees comprise only 1.2% of the urban forest population. A majority of the identified trees were within the 30 to 46 and 46 to 61 cm DBH classes, indicating a young and maturing forest (Morgenroth et al., 2020) (Fig. 4). The average tree in the landscape had minimal dieback (between 10 and 15%) and a low percent missing from the canopy (between 25 to 30%). This indicates that the forest stand is healthy, with most trees maintaining their canopy. In addition to the forest inventory, I analyzed the available planting space within the buyout matrix in ArcGIS and found that there was 49,654 m^2 of available space. When using a mature tree-size of 74 m^2 to represent the average mature tree size of the existing canopy, this results in 671 available spaces to plant trees.

Figure 3. Map of the existing trees within the buyout matrix. Each point represents a tree that was measured during the field season.

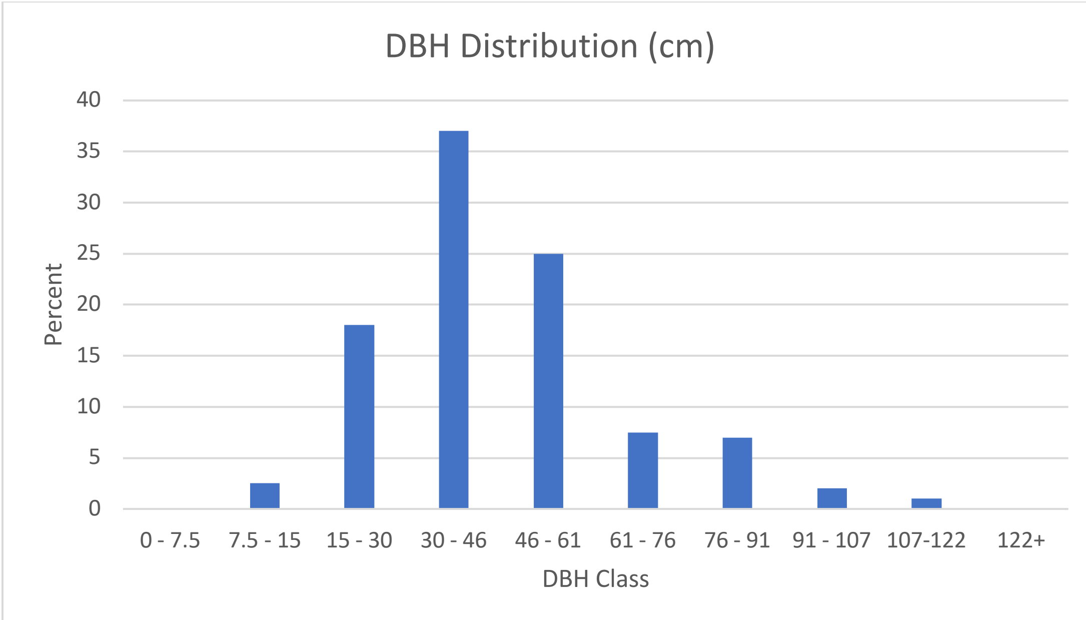

Figure 4. The distribution of inventoried trees based on their DBH class.

3.2 Current provided ecosystem services

At present, the urban forest within the study area stored an estimated 237,000 kgs of carbon. This value represents all of the carbon stored within each tree at the time of sampling and does not accrue annually. *Quercus virginiana* stored the most carbon on average among each of the observed tree species at roughly 950 kgs of carbon each. Furthermore, *Quercus virginiana* accounted for nearly 50,000 kgs (~20%) of the total carbon storage while only making up roughly 15% of the total population. *Ulmus crassifolia* and *Carya illinoinensis* also provide significant carbon storage within this urban forest, storing roughly 41,000 and 42,000 kgs of carbon, respectively.

Photosynthesis gives trees the ability to absorb carbon dioxide and water to produce sugars that are either expended for energy or converted into organic compounds that could be used to promote tree growth in the form of limbs, branches, leaves, or fruiting bodies. The carbon that was absorbed into the tree is not expended, but rather stored as biomass within the tree's tissues. Therefore, trees act as natural carbon sinks that remove carbon from the atmosphere, which is especially notable in urban areas where carbon dioxide tends to be more readily released into the atmosphere from transportation, industry, heating and cooling of buildings, waste storage such as landfills, and residential sources. Within the study area, an estimated 7,860 kgs of carbon were sequestered annually. *Ulmus crassifolia* contributed the most to carbon sequestration, followed closely by *Quercus virginiana*, at roughly 2,047 and 1,555 kgs of carbon sequestered each year, respectively. *Carya illinoinensis*, *Ulmus americana*, and *Quercus stellata*, made sizable contributions to carbon sequestration as well, annually sequestering roughly 1,120, 920 and 650 kgs of carbon each, respectively.

The trees in the study site also removed 239 kgs of air pollution from the atmosphere annually. Of the six air pollutants analyzed in this study, ozone was the most reduced by the urban forest, accounting for 165 of the 239 total kgs reduced, according to i-Tree Eco™ estimations. Removal of PM_{10} amounted to 48 kgs. Smaller contributions in the removal of carbon monoxide, nitrogen dioxide, PM 2.5, and sulfur dioxide accounted for the remaining 26 kgs of air pollution removal. There was little variation among tree species on their contributions to removing air pollution, ranging from 421 g to 784 g removed annually. *Celtis laevigata* removed the least air pollution per tree, and *Quercus alba* removed the most air pollution per tree within the study site.

The urban forest also intercepted 266 cubic meters of water during the year. Similarly to air pollution removal, there were little differences among species in their ability to intercept water and reduce surface runoff. *Quercus alba* was responsible for the highest rates of avoided runoff at 2.6 cubic meters per tree. Conversely, *Quercus macrocarpa*, *Pinus taeda*, and *Celtis laevigata* contributed the least to avoided runoff at 0.5 cubic meters per tree. On average, most species intercepted .75 cubic meters of water per tree. Since this study area exists within a floodplain, intercepted runoff is especially important and tree species such as *Quercus alba* that significantly contribute to avoided runoff may be prioritized by urban forest managers looking to prevent further flooding events in addition to greening the space.

As mentioned, urban trees provide a wealth of economic value to communities as well. The total annual benefits provided by the trees in the study site were a summation of each of the monetary values of the provided ecosystem services. Essentially, this means that the monetary values associated with gross carbon sequestration, air pollution removal, and avoided runoff are summed as total annual benefits. Each year, the trees in this urban matrix provide an estimated $3,659 of value in the form of ecosystem services. Generally, *Carya illinoinensis* and *Quercus alba* provided the greatest total annual benefits to the landscape, though each species may be more proficient in providing certain ecosystem services than others. For example, *Pinus taeda* generally provided more carbon storage than *Celtis laevigata*, but *Celtis laevigata* provided more value in avoided runoff than *Pinus taeda*. Replacement value was also determined, which represented the cost of replacing all the trees in the urban park matrix. At present, the replacement value of the urban forest within the study site was $1.35 million dollars. *Ulmus americana* had the greatest monetary value at $4,166 per tree, while *Celtis laevigata* provided the least value at $1,266 per tree.

3.3 Available Planting Space

Across the study area, 671 planting spaces of 74 m^2 were identified as potential planting locations utilizing the GIS, equating to 49,654 m^2 of available space. This was determined by summing each hexagon not already forested or covered in impervious surfaces in the GIS. Each planting space was represented by a 74 m^2 hexagon to represent the average tree crown spread across the canopy and the general crown shape. This was done to ensure that proposed tree plantings would have the resources necessary to thrive and grow to maturity. Since there were 671 available planting spaces, to plant 25% of the available space, 168 additional trees were added to the existing forest within the landscape. This resulted in scenario 1 having 168 trees planted, scenario 2 had 336 trees planted, scenario 3 had 504 trees planted, and scenario 4 had all 671 available planting spaces planted.

3.4 Top 10

Under the top 10 planting scenarios, the distribution of tree plantings was representative of the trees that already existed within the landscape. Each of the four planting scenarios were sent into i-Tree Eco for processing and the results of their projected ecosystem services were received on May 1st, 2023. I found that under each planting scenario there was an increase in the ecosystem services derived from the trees in the landscape. This was expected as additional trees in the landscape would provide additional benefits across all ecosystem services. Furthermore, the results of the ecosystem services analysis displayed the projected ecosystem services stayed constant by species, as the mean values of the observed trees were used for the scenario modeling. Essentially, this means that each individual tree added to the landscape provided the same benefits as other members of the same species. In terms of the scenarios, this meant that the addition of 25% planted space for each sequential scenario would add a multiple of the first planting scenario to the landscape. The average values of each tree were used during analysis to determine the potential of trees once they reach a mature size.

To create these tree planting scenarios under the top 10 method, outliers were removed from the existing data based on observed DBH. Since the data is not normally distributed, I determined the interquartile range of the data and removed trees that did not fall between this range. Then, found the average values of DBH, N-S width, E-W, width, crown light exposure, total height, crown height, and base height of the trees already existing within the landscape (Table 3). Therefore, each tree species used in the tree planting scenarios were based on the existing landscape, not based on a standard DBH or crown size.

Table 3. Table showing the values used for i-Tree Eco^{TM} analysis for the top 10 tree planting method.

Species	*% of Pop*	*#*	*DBH*	*N/S Width*	*E/W Width*	*CLE*	*Total Height*	*Top Height*	*Base Height*
American elm	7.1	12	22.5	47.7	44.2	3	44.2	44.2	17.1
Buckley oak	4.6	8	15.9	36.8	35.3	3	43.4	43.4	14.5
Bur oak	2.5	4	14.4	23.8	26.4	3	37.1	37.1	13.8
Cedar elm	20.8	35	16.4	33.5	33.3	3	41.2	41.2	19.3
Live oak	16.1	27	17.6	38.6	38	3	39.3	39.3	16
Loblolly pine	3.7	6	17.9	29.6	27.9	3	53.7	53.7	33
Pecan	22.1	37	18.5	38.6	40.2	3	43	43	17.7
Post oak	13.6	23	15.9	32.3	32.6	3	41.2	41.2	17.7
Sugarberry	5.4	9	12.2	26.8	28	3	36.8	36.8	13.6
White oak	4.1	7	18.7	38.8	36	3	41.5	41.5	19

Under each planting scenario, an estimated 101,100 kg of carbon storage, 2,950 kg/yr of carbon sequestration, 111,300 g of air pollution removal, 128 m^3/yr of avoided runoff, and $515,350 of replacement value would be added to the landscape (Table 4). The total annual value was the summation of the individual monetary benefits of each ecosystem service and was estimated at an additional $1,590 for each additional planting scenario. Notably, the value of each ecosystem service nearly doubled the existing forest under scenario 2 when half of the available space in the landscape was planted.

Table 4. Ecosystem services provided by each individual tree within the tree planting scenarios.

Tree species	*N*	*Carbon storage (USD)*	*Carbon storage (kg)*	*Gross carbon sequestration (kg/yr)*	*Air pollution removal (g/yr)*	*Air pollution removal ($/yr)*	*Avoided runoff (m3/yr)*	*Avoided runoff ($/yr)*	*Total annual benefits (USD)*	*Replacement value (USD)*
Ulmus americana	*12*	125.92	669.8	27.1	671.6	5.49	1	2.45	11.59	4,166.65
Quercus buckleyi	*8*	93.70	498.4	12	641.9	5.24	1	2.34	8.62	2,469.65
Quercus macrocarpa	*4*	70.32	374	11.5	479.5	3.92	0.7	1.75	6.68	2,671.74
Ulmus crassifolia	*35*	94.00	500	25.4	519.8	4.25	0.8	1.9	10.15	2,876.83
Quercus virginiana	*27*	154.77	823.3	31.3	620.7	5.07	1	2.27	11.81	3,547.39
Pinus taeda	*6*	75.06	399.2	16.8	460.7	3.76	0.7	1.68	7.37	2,905.02
Carya illinoinensis	*37*	82.68	439.8	11.1	605.4	4.95	0.9	2.21	8.64	3,641.86
Quercus stellata	*23*	100.00	531.9	14.5	619.9	5.06	1	2.26	8.61	2,816.15
Celtis laevigata	*9*	9.96	53	1.9	421.1	3.44	0.7	1.54	4.52	1,266.27
Quercus alba	*7*	148.51	790	18.8	784.4	6.41	1.2	2.86	10.95	3,564.76
Total	168									

To further explain, the baseline value of avoided runoff contributed by the trees already existing within the park matrix was estimated at 266.2 m^3/yr. Planting scenario 1 required the addition of 168 trees to the existing park matrix to plant 25% of the available space and resulted in an estimated 390.8 m^3/yr of avoided runoff. Planting scenarios 2, 3, and 4, had estimated avoided runoff values of 518.5 m^3/yr, 646.2 m^3/yr, and 773.9 m^3/yr. Therefore, in the case of avoided runoff, roughly an additional 128 m^3/yr of precipitation was intercepted by trees within the park matrix for every 25% of available space that was planted. The other ecosystem services exhibit the same results, whereas each sequential scenario adds a multiple of the first scenario to the existing park matrix. Results of the i-Tree Eco™ analyses are presented in Table 5.

Table 5. Table displaying the results of the i-Tree Eco analyses of each scenario under the top 10 planting method.

Ecosystem Services	***Baseline***	***Scenario 1 - 25%***	***Scenario 2 - 50%***	***Scenario 3 - 75%***	***Scenario 4 - 100%***
Carbon storage (kg)	236,873.40	337,982.90	439,092.40	540,201.90	641,311.40
Gross carbon sequestration (kg/yr)	7,859.40	10,814.60	13,769.80	16,725.00	19,680.20
Air pollution removal (g/yr)	239,049.80	345,192.50	456,511.50	567,830.50	679,149.50
Avoided runoff (m3/yr)	266.20	390.8	518.5	646.2	773.9
Total annual value (USD)	3,659.27	5,234.51	6,822.65	8,410.79	9,998.93
Replacement value (USD)	1,353,664.70	1,869,021.79	2,384,378.88	2,899,735.97	3,415,093.06

Statistical analyses using Mann-Whitney U tests allowed me to determine which, if any of the ecosystem services demonstrated statistical significance. A Mann-Whitney U test was completed for each of the four planting scenarios compared to the existing landscape to determine if the planted trees truly improved the provisioning of ecosystem services. Results of the Mann-Whitney U tests depicted the following. Although there were increases in provided services, in the first top 10 planting scenario, these increases were not statistically significant for any of the ecosystem services. Under the second planting scenario, where 50% of the available planting space would be planted, there was only one statistically significant increase in ecosystem services which was observed for avoided runoff. Under the third planting scenario, statistically significant increases in estimated ecosystem services occurred for: carbon storage (kg), air pollution removal (g/yr), and avoided runoff (m^3/yr). While the fourth planting scenario, planting 100% of the available space, would cause an increase in net benefits, there were no additional statistically significant increases in ecosystem services from 75 to 100% of the landscape being planted. Therefore, three ecosystem services did not demonstrate statistically significant increases under any of the top 10 planting scenarios: gross carbon sequestration (kg/yr), total annual benefits ($/yr), and replacement value ($).

3.4 10-20-30

Under the 10-20-30 planting scenarios, the composition of tree plantings followed the benchmark set by Santamour: no more than 10% of any species, 20% of any genus, and 30% of any family (Santamour, 1990). For these scenarios, I selected 10 native species already present within the park matrix. This was to ensure that the selected species could survive and grow in the specific conditions of the study site. Following selection, only two genera reached the 20% benchmark set, those being *Quercus* and *Ulmus*. Furthermore, this resulted in 17 trees of each of the 10 species being planted for every 25% of landscape planted in each scenario, aside from *Acer negundo* and *Celtis laevigata* which only added 16 trees per planting scenario (Table 6). Each scenario required the addition of 168 trees, so I needed to select which two species would have one less tree planted than the others. I selected *Acer negundo* and *Celtis laevigata* as the species with one less tree due to their smaller average size within this urban forest and therefore, less expected benefits (Table 7). Simply, larger trees should yield greater values of ecosystem services, so it would likely be more beneficial to plant an additional larger tree, such as *Quercus virginiana* than it would be to plant an additional *Acer negundo* or *Celtis laevigata.*

Table 6. Table showing the number of trees of each species required for the 10-20-30 tree planting scenarios

Tree species	***Scenario 1 - 25%***	***Scenario 2 - 50%***	***Scenario 3 - 75%***	***Scenario 4 - 100%***
Ulmus crassifolia	17	34	51	68
Ulmus americana	17	34	51	68
Carya illinoinensis	17	34	51	68
Quercus virginiana	17	34	51	68
Quercus stellata	17	34	51	68
Celtis laevigata	16	32	48	64
Pinus taeda	17	34	51	68
Acer negundo	16	32	48	64
Pyrus calleryana	17	34	51	68
Quercus alba	17	34	51	68
Total	168	336	504	672

Table 7. Table showing the values used for i-Tree Eco™ analysis for the 10-20-30 tree planting method.

Species	***% of Pop***	***#***	***DBH***	***N/S Width***	***E/W Width***	***CLE***	***Total Height***	***Top Height***	***Base Height***
Ulmus americana	10	17	22.5	47.7	44.2	3	44.2	44.2	17.1
Acer negundo	10	16	5.4	9.8	12.3	3	17.1	17.1	13.6
Pyrus calleryana	10	17	17.7	33.2	32.7	3	34	34	6.4
Ulmus crassifolia	10	17	16.4	33.5	33.3	3	41.2	41.2	19.3
Quercus virginiana	10	17	17.6	38.6	38	3	39.3	39.3	16
Pinus taeda	10	17	17.9	29.6	27.9	3	53.7	53.7	33
Carya illinoinensis	10	17	18.5	38.6	40.2	3	43	43	17.7
Quercus stellata	10	17	15.9	32.3	32.6	3	41.2	41.2	17.7
Celtis laevigata	10	16	12.2	26.8	28	3	36.8	36.8	13.6
Fraxinus albicans	10	17	17.4	31.2	40.6	3	42.3	42.3	22.4

Similarly to the top 10 planting scenarios, the 10-20-30 planting scenarios also yielded greater benefits to the landscape through each sequential scenario, as expected. Under each planting scenario, an estimated 94,800 kg of carbon storage, 2,580 kg/yr of carbon sequestration, 101,000 g of air pollution removal, 115 m^3/yr of avoided runoff, and $446,500 of replacement value would be added to the existing landscape. The total annual value was the summation of the individual monetary benefits of each ecosystem service, which was estimated at an additional $1,420 for each additional planting scenario. Furthermore, planting 50% of the available space in the landscape yields nearly double the benefits of the existing forest for each ecosystem service under the 10-20-30 scenario as well.

Table 8 displays the results for each planting scenario under the 10-20-30 tree planting method.

Table 8. Table displaying the results of the i-Tree Eco analyses of each scenario under the 10-20-30 planting method.

Ecosystem Services	*Baseline*	*Scenario 1 - 25%*	*Scenario 2 - 50%*	*Scenario 3 - 75%*	*Scenario 4 - 100%*
Carbon storage (kg)	236,873.40	331,686.60	426,499.80	521,313.00	616,126.20
Gross carbon sequestration (kg/yr)	7,859.40	10,437.40	13,015.40	15,593.40	18,171.40
Air pollution removal (g/yr)	239,049.80	332,925.30	433,869.80	534,814.30	635,758.80
Avoided runoff (m3/yr)	266.20	377.1	492.2	607.3	722.4
Total annual value (USD)	3,659.27	5,050.51	6,471.29	7,892.07	9,312.85
Replacement value (USD)	1,353,664.70	1,800,164.41	2,246,664.12	2,693,163.83	3,139,663.54

Mann-Whitney U tests were also completed for the 10-20-30 scenarios to determine statistical significance of ecosystem services. Under the first planting scenario, none of the ecosystem services showed a statistically significant improvement in their provisioning within the urban forest. Increases in gross carbon sequestration (kg/yr) became statistically significant once 50% of the available space was planted under scenario 2 and remained significant when 75% and 100% of the available space was planted under scenarios 3 and 4, respectively. No other ecosystem services were significant when half of the available planting space was planted. When 75% of the available space was planted under scenario 3, the increase in replacement value ($) was also shown to be statistically significant and remained that way when all the available space was planted in scenario 4. Planting all available space under scenario 4 yielded no additional statistically significant ecosystem services. Furthermore, there were no statistically significant increases in the provisioning of carbon storage (kg), air pollution removal (g/yr), avoided runoff (m^3/yr), or total annual benefits ($) across any of the four planting scenarios.

3.5 Analyzing Top 10 and 10-20-30 Planting Scenarios

After receiving the results of the i-Tree Eco™ analyses for each planting scenario, I then compared the top 10 and 10-20-30 planting scenarios using a series of Mann-Whitney U tests (Tables 8, 9, and 10). For these tests, each planting scenario under the top 10 tree planting method was tested directly against its counterpart of the 10-20-30 scenarios: Scenario 1 (top 10) and Scenario 1 (10-20-30), Scenario 2 (top 10) and Scenario 2 (10-20-30), Scenario 3 (top 10) and Scenario 3 (10-20-30), Scenario 4 (top 10) and Scenario 4 (10-20-30). The Mann-Whitney U test allowed me to determine if there were statistically significant differences between the two planting methods, top 10 and 10-20-30. If the differences were deemed statistically significant then I could conclude that there is indeed variation among the two different planting methods.

Table 9. Results of Mann-Whitney U tests for the top 10 planting scenarios.

	Scenario 1 - Top 10 (25%)		*Scenario 2 - Top 10 (50%)*		*Scenario 3 - Top 10 (75%)*		*Scenario 4 - Top 10 (100%)*	
Ecosystem Services	***U value***	***P value***	***U value***	***P value***	***U value***	***P value***	***U value***	***P value***
Carbon storage (kg)	90471	0.269	116501.5	0.078	142532	0.027*	168562.5	0.012*
Gross carbon sequestration (kg/yr)	92524	0.579	120607.5	0.376	148691	0.268	176774.5	0.205
Air pollution removal (g/yr)	90895	0.322	116133	0.066	141371	0.016*	166609	0.005*
Avoided runoff (m^3/yr)	89448	0.167	113869	0.019*	138290	0.003*	162711	<0.001*
Total annual benefits (USD/yr)	93005.5	0.67	121201	0.448	149396.5	0.326	177592	0.253
Replacement value (USD)	93556.5	0.781	122672.5	0.657	151788.5	0.578	180904.5	0.525

* statistically significant

Table 10. Results of Mann-Whitney U tests for the 10-20-30 planting scenarios.

	Scenario 1 - 10-20-30 (25%)		*Scenario 2 - 10-20-30 (50%)*		*Scenario 3 - 10-20-30 (75%)*		*Scenario 4 - 10-20-30 (100%)*	
Ecosystem Services	***U value***	***P value***	***U value***	***P value***	***U value***	***P value***	***U value***	***P value***
Carbon storage (kg)	93426	0.754	122411.5	0.617	151397	0.532	180382.5	0.474
Gross carbon sequestration (kg/yr)	88978	0.133	113515.5	0.016*	138053	0.003*	162590.5	<0.001*
Air pollution removal (g/yr)	94165.5	0.908	123970	0.867	152912.5	0.722	181855	0.624
Avoided runoff (m^3/yr)	93519.5	0.773	121752.5	0.52	149985.5	0.379	178218.5	0.293
Total annual benefits (USD/yr)	91927.5	0.475	120019.5	0.312	148111.5	0.226	176203.5	0.176
Replacement value (USD)	90747.5	0.303	117054.5	0.1	143361.5	0.04*	169668.5	0.019*

* statistically significant

Table 11. Results of the Mann-Whitney U tests directly comparing the top 10 and 10-20-30 tree planting methods.

	Scenario 1 - Top 10 vs 10-20-30		***Scenario 2 - Top 10 vs 10-20-30***		***Scenario 3 - Top 10 vs 10-20-30***		***Scenario 4 - Top 10 vs 10-20-30***	
Ecosystem Services	***U value***	***P value***	***U value***	***P value***	***U value***	***P value***	***U value***	***P value***
Carbon storage (kg)	135673.5	0.518	234658.5	0.359	361395.5	0.288	515884.5	0.248
Gross carbon sequestration (kg/yr)	133271.5	0.258	226232.5	0.041*	343323.5	0.005*	484544.5	<0.001*
Air pollution removal (g/yr)	131361	0.129	221642.5	0.008*	334704	<0.001*	470545.5	<0.001*
Avoided runoff (m^3/yr)	132818.5	0.219	226287.5	0.04*	344584.5	0.006*	487709.5	<0.001*
Total annual benefits (USD/yr)	130307.5	0.083	216822	<0.001*	323566.5	<0.001*	450541	<0.001*
Replacement value (USD)	130611.5	0.095	218278.5	0.002*	327441.5	<0.001*	458100.5	<0.001*

* statistically significant

For the first planting scenarios, none of the ecosystem services were shown to have statistical significance. Thus, the differences between the top 10 and 10-20-30 planting methods were indistinguishable when only 25% of the available planting space was planted with trees. Under the second planting scenarios, every ecosystem service was shown to have statistically significant differences between the two planting methods, aside from carbon storage (kg). Therefore, once half of the available planting space was planted there were significant differences in the urban forest's ability to accumulate ecosystem services under the two different tree planting methods. Gross carbon sequestration (kg/yr), air pollution removal (g/yr), avoided runoff (m^3/yr), total annual benefits ($/yr), and replacement value ($) continued to show statistically significant differences in their provisioning across scenario 3 and 4 as well. With 100% of the landscape planted in scenario 4, each ecosystem service aside from carbon storage (kg) demonstrated high degrees of statistical significance (<0.001). Differences in the provisioning of carbon storage (kg) were not statistically significant between any of the planting scenarios, indicating little variation between the two methods. Furthermore, the top 10 planting scenarios depicted higher net ecosystem services across every scenario when compared directly to the 10-20-30 scenarios (Figure 5).

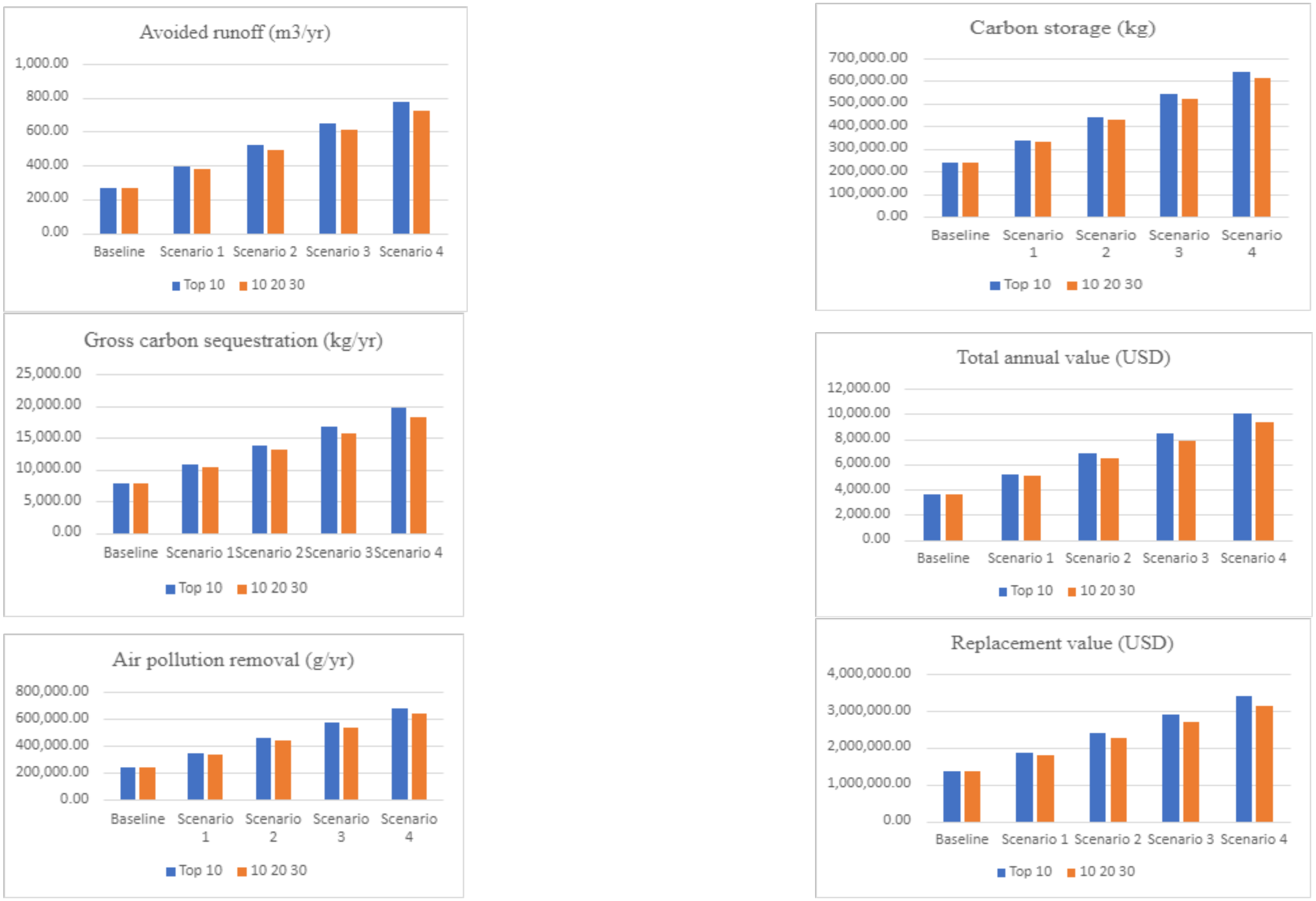

Figure 5. Graphs of each ecosystem service under the two proposed planting methods compared to the baseline existing forest.

4. Discussion

In this study, I identified and quantified critical ecosystem services provided by the trees in an urban buyout landscape: carbon storage, gross carbon sequestration, air pollution removal, avoided runoff, total annual benefits, and replacement value. Moreover, I determined that implementing active management in the form of planting trees in the available space resulting from the acquisition of buyout sites is an effective way to improve the provisioning of the associated ecosystem services. This remains true for both proposed methods of tree planting, following the composition of the top 10 most common species, or using the 10-20-30 rule for urban forestry. Thus, using i-Tree Eco^{TM}, I was able to determine the ecosystem services provided by the current urban forest following a buyout program, as well as determine which method of tree planting would be most effective in practice.

The results of this study demonstrate that planting 25% of the available space within this urban forest landscape would not provide any significant increase in ecosystem services under both the top 10 and 10-20-30 planting methods. Therefore, to make any significant improvements to the landscape, at least 50% of the available space must be planted with trees. With half of the available space planted, an observed increase was found only in the provisioning of avoided runoff under the top 10 scenario. Under the 10-20-30 method, planting half of the available space only resulted in an increased provisioning of gross carbon sequestration. Scenario 2 was also noted to have significant differences between the two planting methods across all ecosystem services except for carbon storage, indicating that there was a clear difference between the ecosystem services expected by each planting method.

It was found that the optimization of ecosystem services was achieved under scenarios 3 and 4 for both the top 10 and 10-20-30 method. Under the top 10 method, results showed increases in carbon storage, air pollution removal, and avoided runoff. Interestingly, the 10-20-30 method yielded increases in gross carbon sequestration and replacement value. Therefore, the top 10 and 10-20-30 methods depicted quite different results. In this specific case of an urban floodplain buyout, the top 10 method would likely be preferred by forest managers, as contributions to avoided runoff are likely a higher concern than the replacement value of the forest itself. Alternatively, in other landscapes, for example, where trees can be harvested for lumber, providing increases in the overall replacement value of the urban forest may be much more favored to increase profits. Therefore, both methods of tree planting excelled in their provisioning of different ecosystem services, allowing management the opportunity to select which ecosystem services they value the most.

Although scenarios 3 and 4 both optimized ecosystem services, in practice, planting all the available space would be an arduous task that would not result in any statistically significant improvements when compared to planting 75% of the available space. Additionally, tree planting can be costly, so it would be more economically advantageous to plant 75% of the available space. It would be unlikely that an urban park system would elect to plant trees across all of the available space, as other pressures such as those for recreation and development often drive stakeholder decisions following disasters (Zavar, 2016). It is also favorable to allow for gaps in the canopy to ensure tree health, further emphasizing why planting all space would not be advantageous (Felipe-Lucia et al., 2018). I therefore concluded that scenario 3 would be the most effective management strategy for the study area to ensure the optimization of ecosystem services. This scenario leaves open space for other land use types within the landscape while providing significant increases in ecosystem services, which is ideal for both the environment and those who manage and recreate in these landscapes.

Although these modeled planting scenarios yielded significant improvements to the landscape, it is important to recognize the current composition and structure of the forest stand. Most of the identified trees had DBH's of 61 cm or below, which are considered small to medium in size. From these results, it could be inferred that many of these trees will continue to grow, yielding greater values of ecosystem services. It is well known that larger trees contribute more ecosystem services than smaller trees, so most of these trees may provide even more in the future, leading to an underestimation of projected future planting scenarios. For example, healthy large trees have been demonstrated to remove 60 times as much air pollution as healthy small trees (Selmi et al., 2016). Large diameter trees have greater potential for carbon storage and sequestration than smaller counterparts as well (Lutz et al., 2012). Therefore, there is potential for the landscape to yield even greater values of ecosystem services than expected in this study.

Furthermore, future studies should incorporate expected growth rates of trees to determine the impact on ecosystem services. This study assumes that planted trees will be able to reach maturity and accrue optimal rates of ecosystem services, but that may not be the case in a real-world application. Projecting this data out several years and including expectations of extreme weather events and expected tree mortality in addition to the growth rate of local trees promises to yield more conclusive results on the rates of ecosystem services accrued by the trees within the buyout landscape.

During field sampling, saplings and seedlings were not recorded within the landscape. Therefore, as the urban forest continues to age, it is believed that some of these young trees will mature to an adult size and begin to contribute ecosystem services within the landscape. Similarly, dead trees were identified and recorded while in the field, but inferences about mortality were not interpreted from this data as these measurements were only taken over one field season. Urban environments are ever-changing, facing a variety of different stressors year after year, so one field season was not an ample amount of time to make educated inferences about mortality rates. For example, recent persistent drought events in Texas could have contributed to the death of several of these trees, though there was no way of identifying when these trees died or what caused their mortality. Future work should record and identify seedlings and saplings within the landscape and observed mortality over multiple field seasons to understand the rates of natural regeneration and mortality. Incorporating rates of natural regeneration and mortality into the projection of tree modeling scenarios would create a better understanding of the urban forest's dynamics.

While planting scenarios prove to be beneficial to the landscape in this study, there are several drawbacks to the implementation of these programs as well. In terms of ecosystem services, many planted trees may not survive long enough to provide a significant contribution to the landscape. The implementation of urban tree planting efforts can be quite costly, involving the individual costs to plant each tree as well as management costs to ensure the trees are planted and maintained. Conversely, the economic, social, and environmental benefits of tree planting programs tend to outweigh the costs. Using i-Tree Eco^{TM} provides municipalities with a more localized understanding of their urban forests and can better inform planting scenarios. For example, if Arlington wanted to provide more sun shading and aesthetic values for their residents, they could prioritize drought-tolerant *Quercus* species for their broad canopies and colorful leaves in the autumn. In a flood zone, it is far more likely that Arlington would emphasize the flood reduction ability of urban trees, so planting *Carya illinoinensis* may be preferred due to providing higher rates of avoided runoff than other species in the study area.

Although i-Tree Eco is a useful tool to quantify ecosystem services, there are still several limitations to the use of this software. Urban forests are complex and dynamic systems, requiring careful attention to the individual structure of each tree to derive accurate field observations. The calculation of individual ecosystem services relies heavily on observed tree measures, so the field data is only as accurate as the observations. To address these limitations, I followed the standard guidelines set in the i-Tree Eco™ user manual. In addition, i-Tree Eco™ only produces an estimate of provided ecosystem services for one year. Since urban forests are constantly changing, improved reliability of these measures could be achieved if the study area was revisited during future growing seasons. There is a forecasting tool within i-Tree Eco™ that allows for the estimation of future events such as planting trees or determining the impact of severe weather, but it was not utilized in this study as it was impossible to change the composition of proposed tree plantings and therefore, I could not compare the two proposed tree planting methods. Overall, though this tool was incredibly useful in the estimation of ecosystem services, there were still limitations in the effectiveness of its use.

Similarly, I acknowledge there are limitations associated with the modeling of the tree planting scenarios. The mean values of the observed field characteristics were used in modeling, but real-world application of tree planting efforts would likely not involve trees of such large sizes due to high cost and feasibility of transport. Therefore, these modeled scenarios represent what could be achieved assuming planted trees reach the mean size and age of the current landscape. In addition, tree mortality rates were not accounted for in these scenarios. Mortality may cause significant changes to the composition and structure of the landscape, reducing the overall amount of provided ecosystem services. Urban forest canopies have quite complex regeneration and mortality patterns, so no assumptions were made about these factors. Future longitudinal studies inventorying tree seedlings and sapling and annual tree mortality within the study area may improve the reliability of these planting scenarios by adding these factors in scenario modeling.

While not included in this study's scope, the optimization of ecosystem services does not rely solely on ecological values, it also includes cultural services. These cultural services can be in the form of spiritual connections, aesthetic values, and educational and recreational opportunities. This further emphasizes that planting all available space would not be the most effective way to optimize ecosystem services in a buyout landscape. Although planting all the available space yields the highest net ecological benefits, these cultural services would likely be greatly hindered by a fully enclosed urban forest. One study compared New York City’s forests to more manicured park areas and found that park visitors that do not visit the natural forests in the area do so due to a lack of accessibility and a limited sense of safety within natural areas (Sonti et al., 2020). Since this study area already exists within an urban park matrix it would likely be important to highlight increased use of the park area as opposed to driving visitors away from recreating in the landscape. Further research should conduct a socio-cultural analysis of the urban park matrix to ensure residents would benefit from increased tree cover in their neighborhood.

5. Conclusions

Buyout landscapes and similar areas that are converted to green spaces have the potential to provide essential ecosystem services within urban areas, including carbon storage and sequestration, air pollution removal, avoided runoff, and replacement value. Moreover, these landscapes may be optimal locations to seek improvements in ecosystem services through an active management approach. I found that planting trees in the available space in the landscape could lead to significant improvements in the provisioning of all but one of the ecosystem services. It was determined that carbon storage, avoided runoff and air pollution removal could all exhibit significant improvements in their provisioning under scenario 3 of the top 10 method, which are of particular importance in urban areas to improve public health and reduce flood risk. Alternatively, the 10-20-30 method provided significant improvements in the provisioning of carbon sequestration and replacement value. This allows forest managers to select which tree planting method to employ based on which ecosystem services they would like to prioritize. Therefore, this research suggests that floodplain buyouts are not just effective as a recovery tool for flood hazards and disasters; they also provide opportunities for urban forest managers to oversee and maintain the area, as well as take the necessary actions to increase the collective benefits of the urban landscape. Furthermore, this study emphasizes the need for active management of buyout landscapes. These sites are ideal landscapes for active management, such as implementing a tree planting program, as the land is required to remain public, green space. While I focused on one urban buyout matrix, there are thousands of buyout landscapes across the United States that likely employ similar passive management techniques, ultimately not achieving optimal use of the landscape. As pressures from climate change and future flooding persist, it becomes increasingly important to optimize urban forests to reduce flood risk, improve air quality, and store carbon within urban areas.

References

Alberti, M., Marzluff, J. M., Shulenberger, E., Bradley, G., Ryan, C., & Zumbrunnen, C. (2003). Integrating Humans into Ecology: Opportunities and Challenges for Studying Urban Ecosystems. *BioScience*, *53*(12), 1169-1179.

Atoba, K., Newman, G., Brody, S., Highfield, W., Kim, Y., & Juan, A. (2021). Buy them out before they are built: evaluating the proactive acquisition of vacant land in flood-prone areas. *Environ Conserv*, *48*(2), 118-126. https://doi.org/10.1017/s0376892921000059

Atoba, K. O., Brody, S. D., Highfield, W. E., Shepard, C. C., & Verdone, L. N. (2020). Strategic property buyouts to enhance flood resilience: a multi-criteria spatial approach for incorporating ecological values into the selection process. *Environmental Hazards*, *20*(3), 229-247. https://doi.org/10.1080/17477891.2020.1771251

Barker, L., Pace, M., Grubisich, M., Wright, G. T., Beaudoing, M., Guynes, T., Hanou, I., Cantor, J., & Saal, P. (2016). State of the Denton Urban Forest. https://www.itreetools.org/documents/295/2016_State_of_the_Denton_Urban_Forest_Preservation_Tree.pdf

Bendor, T. K., Salvesen, D., Kamrath, C., & Ganser, B. (2020). Floodplain Buyouts and Municipal Finance. *Natural Hazards Review*, *21*(3), 04020020. https://doi.org/10.1061/(asce)nh.1527-6996.0000380

Berland, A., Shiflett, S. A., Shuster, W. D., Garmestani, A. S., Goddard, H. C., Herrmann, D. L., & Hopton, M. E. (2017). The role of trees in urban stormwater management. *Landscape and Urban Planning*, *162*, 167-177.

Brody, S. D., & Highfield, W. E. (2013). Open space protection and flood mitigation: A national study. *Land Use Policy*, *32*, 89-95. https://doi.org/10.1016/j.landusepol.2012.10.017

Brody, S. D., Zahran, S., Highfield, W. E., Grover, H., & Vedlitz, A. (2008). Identifying the impact of the built environment on flood damage in Texas. *Disasters*, *32*(1), 1-18. https://doi.org/10.1111/j.1467-7717.2007.01024.x

Calil, J., Beck, M. W., Gleason, M., Merrifield, M., Klausmeyer, K., & Newkirk, S. (2015). Aligning Natural Resource Conservation and Flood Hazard Mitigation in California. *PLoS One*, *10*(7), e0132651. https://doi.org/10.1371/journal.pone.0132651

Conway, T. M. (2016). Tending their urban forest: Residents' motivations for tree planting and removal. *Urban Forestry & Urban Greening*, *17*, 23-32. https://doi.org/10.1016/j.ufug.2016.03.008

Curran-Groome, W., Haygood, H., Hino, M., BenDor, T. K., & Salvesen, D. (2021). Assessing the full costs of floodplain buyouts. *Clim Change*, *168*(1-2), 3. https://doi.org/10.1007/s10584-021-03178-x

Davies, Z. G., Edmondson, J. L., Heinemeyer, A., Leake, J. R., & Gaston, K. J. (2011). Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale. *Journal of Applied Ecology*, *48*(5), 1125-1134. https://doi.org/10.1111/j.1365-2664.2011.02021.x

Edmondson, J. L., Stott, I., Davies, Z. G., Gaston, K. J., & Leake, J. R. (2016). Soil surface temperatures reveal moderation of the urban heat island effect by trees and shrubs. *Sci Rep*, *6*, 33708. https://doi.org/10.1038/srep33708

Felipe-Lucia, M. R., Soliveres, S., Penone, C., Manning, P., van der Plas, F., Boch, S., Prati, D., Ammer, C., Schall, P., Gossner, M. M., Bauhus, J., Buscot, F., Blaser, S., Bluthgen, N., de Frutos, A., Ehbrecht, M., Frank, K., Goldmann, K., Hansel, F., . . . Allan, E. (2018). Multiple forest attributes underpin the supply of multiple ecosystem services. *Nat Commun*, *9*(1), 4839. https://doi.org/10.1038/s41467-018-07082-4

FEMA. (1998). Property Acquisition Handbook for Local Communities: A Summary for States.

FEMA. (2022). FEMA Flood Map Service Center. https://msc.fema.gov/portal/search?AddressQuery=clarence%20foster%20park#searchresultsanchor

Foundation, T. T. (2021). Dallas Urban Forest Master Plan. https://dallascityhall.com/projects/forestry/DCH%20Documents/City%20of%20Dallas%202021%20Urban%20Forest%20Master%20Plan.pdf

Gamfeldt, L., Snall, T., Bagchi, R., Jonsson, M., Gustafsson, L., Kjellander, P., Ruiz-Jaen, M. C., Froberg, M., Stendahl, J., Philipson, C. D., Mikusinski, G., Andersson, E., Westerlund, B., Andren, H., Moberg, F., Moen, J., & Bengtsson, J. (2013). Higher levels of multiple ecosystem services are found in forests with more tree species. *Nat Commun*, *4*, 1340. https://doi.org/10.1038/ncomms2328

Garrison, J. D. (2017). Seeing the park for the trees: New York's "Million Trees" campaign vs. the deep roots of environmental inequality. *Environment and Planning B: Urban Analytics and City Science*, *46*(5), 914-930. https://doi.org/10.1177/2399808317737071

Halff Associates Inc. (2022). *Tarrant County Aerial Imagery Classification*.

Hirokawa, K. H. (2011). Sustainability and the urban forest: an ecosystem services perspective. *Natural Resources Journal*, *51*(2), 233-259.

i-Tree. (2021a). i-Tree Eco Field Guide v.6.0. https://www.itreetools.org/documents/275/EcoV6_UsersManual.2021.09.22.pdf

i-Tree. (2021b). i-Tree Eco User's Manual v6.0. https://www.itreetools.org/documents/275/EcoV6_UsersManual.2021.09.22.pdf

Kim, G. (2016). Assessing Urban Forest Structure, Ecosystem Services, and Economic Benefits on Vacant Land. *Sustainability*, *8*(7). https://doi.org/10.3390/su8070679

Lavy, B. L., & Hagelman, R. R. (2017). Spatial and Temporal Patterns Associated with Permitted Tree Removal in Austin, Texas, 2002–2011. *The Professional Geographer*, *69*(4), 539-552. https://doi.org/10.1080/00330124.2016.1266953

Lee, R. J. (2021). Vacant land, flood exposure, and urbanization: Examining land cover change in the Dallas-Fort Worth metro area. *Landscape and Urban Planning*, *209*. https://doi.org/10.1016/j.landurbplan.2021.104047

Livesley, S. J., McPherson, E. G., & Calfapietra, C. (2016). The Urban Forest and Ecosystem Services: Impacts on Urban Water, Heat, and Pollution Cycles at the Tree, Street, and City Scale. *Journal of Environmental Quality*, *45*(1), 119-124. https://doi.org/10.2134/jeq2015.11.0567

Locke, D. H., Roy Chowdhury, R., Grove, J. M., Martin, D. G., Goldman, E., Rogan, J., & Groffman, P. (2018). Social Norms, Yard Care, and the Difference between Front and Back Yard Management: Examining the Landscape Mullets Concept on Urban Residential Lands. *Society & Natural Resources*, *31*(10), 1169-1188. https://doi.org/10.1080/08941920.2018.1481549

Loughran, K., & Elliott, J. R. (2019). Residential buyouts as environmental mobility: examining where homeowners move to illuminate social inequities in climate adaptation. *Population and Environment*, *41*(1), 52-70. https://doi.org/10.1007/s11111-019-00324-7
Lutz, J. A., Larson, A. J., Swanson, M. E., & Freund, J. A. (2012). Ecological Importance of Large-Diameter Trees in a Temperate Mixed-Conifer Forest. *PLoS One*, *7*(5), e36131. https://doi.org/10.1371/journal.pone.0036131
Mach, K. J. K., Caroline M.; Hino, Miyuki; Siders, A. R.; Johnston, Erica M.; Field, Christopher B. (2019). Managed retreat through voluntary buyouts of flood-prone properties. *Science Advances*, *5*(10), eaax8995. https://doi.org/10.1126/sciadv.aax8995
McPhearson, T., Kremer, P., & Hamstead, Z. A. (2013). Mapping ecosystem services in New York City: Applying a social–ecological approach in urban vacant land. *Ecosystem Services*, *5*, 11-26. https://doi.org/10.1016/j.ecoser.2013.06.005
McPherson, E. G. (2014). Monitoring Million Trees LA: Tree Performance During the Early Years and Future Benefits. *Arboriculture & Urban Forestry*, *40*(5), 285-300. https://www.fs.usda.gov/psw/publications/mcpherson/psw_2014_mcpherson003.pdf
McPherson, G. E. (1998). Structure and sustainability of Sacramento's urban forest. *Journal of Arboriculture*, *24*(4), 174-190.
McPherson, G. E. S., James R. (1999). Carbon dioxide reduction through urban forestry: guidelines for professional and volunteer tree planters. https://doi.org/https://doi.org/10.2737/PSW-GTR-171
Monteiro, M. V., Handley, P., & Doick, K. J. (2019). An insight to the current state and sustainability of urban forests across Great Britain based on i-Tree Eco surveys. *Forestry: An International Journal of Forest Research*. https://doi.org/10.1093/forestry/cpz054
Morgan, M., & Ries, P. D. (2022). Planting free trees on private property: understanding urban residents' motivations and hesitations. *Urban Forestry & Urban Greening*, *71*, 127557. https://doi.org/https://doi.org/10.1016/j.ufug.2022.127557
Morgenroth, J., Nowak, D., & Koeser, A. (2020). DBH Distributions in America's Urban Forests—An Overview of Structural Diversity. *Forests*, *11*(2). https://doi.org/10.3390/f11020135
Naturally Resilient Communities. (n.d.). *Rush Creek Property Acquisition Project, Arlington, Texas*.
NOAA. (n.d.). *Arlington Climatology*. Retrieved from https://www.weather.gov/fwd/arlingtonclimatology
Nowak, D. J., Hoehn, R., & Crane, D. E. (2007). Oxygen production by urban trees in the United States. *Arboriculture & Urban Forestry*, *33*(3), 220-226.
Nowak, D. J. C., Daniel E.; Stevens, Jack C.; Hoehn, Robert E.; Walton, Jeffrey T.; Bond, Jerry. (2008). A Ground-Based Method of Assessing Urban Forest Structure and Ecosystem Services. *Arboriculture & Urban Forestry*, *34*(6), 347-358.
Pace, M., & Kralik, A. (2014). Plano Urban Forest Ecosystem Analysis. *Preservation Tree Services*. https://www.itreetools.org/documents/340/Plano_Urban_Forest_Ecosystem_Analysis_2014.pdf

Palmer, E. J. (1948). Hybrid Oaks of North America. *Journal of the Arnold Arboretum*, *29*, 1-48.
Pearce, L., Kirkpatrick, J., & Davison, A. (2013). Using Size Class Distributions of Species to Deduce the Dynamics of the Private Urban Forest. *Arboriculture & Urban Forestry*, *39*(2). https://doi.org/10.48044/jauf.2013.011
Reyes-Riveros, R., Altamirano, A., De La Barrera, F., Rozas-Vásquez, D., Vieli, L., & Meli, P. (2021). Linking public urban green spaces and human well-being: A systematic review. *Urban Forestry & Urban Greening*, *61*. https://doi.org/10.1016/j.ufug.2021.127105
Riondato, E., Pilla, F., Sarkar Basu, A., & Basu, B. (2020). Investigating the effect of trees on urban quality in Dublin by combining air monitoring with i-Tree Eco model. *Sustainable Cities and Society*, *61*. https://doi.org/10.1016/j.scs.2020.102356
Santamour, F. S. (1990). Trees for urban planting: diversity, uniformity, and common sense. *Proceedings of the Seventh Conference of the Metropolitan Tree Improvement Alliance (METRIA)*, 57-65.
Selmi, W., Weber, C., Rivière, E., Blond, N., Mehdi, L., & Nowak, D. (2016). Air pollution removal by trees in public green spaces in Strasbourg city, France. *Urban Forestry & Urban Greening*, *17*, 192-201. https://doi.org/10.1016/j.ufug.2016.04.010
Sheppard, L. B., Katharine; Appelbaum, Alec. (2021). *On safer ground: floodplain buyouts and community resilience*
Sonti, N. F., Campbell, L. K., Svendsen, E. S., Johnson, M. L., & Novem Auyeung, D. S. (2020). Fear and fascination: Use and perceptions of New York City's forests, wetlands, and landscaped park areas. *Urban Forestry & Urban Greening*, *49*. https://doi.org/10.1016/j.ufug.2020.126601
Sousa-Silva, R., Duflos, M., Ordóñez Barona, C., & Paquette, A. (2023). Keys to better planning and integrating urban tree planting initiatives. *Landscape and Urban Planning*, *231*. https://doi.org/10.1016/j.landurbplan.2022.104649
Wu, J., Wang, Y., Qiu, S., & Peng, J. (2019). Using the modified i-Tree Eco model to quantify air pollution removal by urban vegetation. *Sci Total Environ*, *688*, 673-683. https://doi.org/10.1016/j.scitotenv.2019.05.437
Zavar, E. (2015). Residential Perspectives: The Value of Floodplain- Buyout Open Space. *Geographical Review*, *105*(1), 78-95. https://doi.org/10.1111/j.1931-0846.2014.12047.x
Zavar, E. (2016). The role of magnetic agents in shaping post-disaster land use. *Land Use Policy*, *56*, 38-46. https://doi.org/10.1016/j.landusepol.2016.04.037
Zavar, E., & Hagelman, R. R. (2016). Land use change on US floodplain buyout sites, 1990-2000. *Disaster Prevention and Management*, *25*(3), 360-374. https://doi.org/10.1108/Dpm-01-2016-0021
Zhang, Y., Van den Berg, A. E., Van Dijk, T., & Weitkamp, G. (2017). Quality over Quantity: Contribution of Urban Green Space to Neighborhood Satisfaction. *Int J Environ Res Public Health*, *14*(5). https://doi.org/10.3390/ijerph14050535

Vita

Peter Joseph Fahey Jr. was born on October 23, 1997, in Farmingville on Long Island, New York. He is the son of Peter Fahey Sr. and Donnamarie Favuzza. A 2015 graduate of Sachem High School East in Farmingville, New York. He went on to receive his Bachelor of Science in environmental science and biological sciences with a minor in ecology and evolutionary biology from the University of Connecticut in 2019. Throughout his education, he completed internships with the United States Fish and Wildlife Service at the Wertheim National Wildlife Refuge in Shirley, New York and the United States National Park Service at Devil's Postpile National Monument in Mammoth Lakes California. He also taught outdoor environmental education to grade school students before pursuing his Master of Science degree. In August of 2021, Peter enrolled in graduate study at Texas Christian University in pursuit of a Master of Science degree in environmental science and sustainability sciences.

Abstract

USING i-Tree Eco™ TO EVALUATE ECOSYSTEM SERVICES FOLLOWING FLOODPLAIN BUYOUTS

By PETER FAHEY JR., M.S., 2024
College of Science and Engineering
Texas Christian University

Thesis Advisor: Dr. Brendan L. Lavy, Assistant Professor of Environmental and Sustainability Sciences

Committee Members: Dr. Michael C. Slattery, Professor of Environmental and Sustainability Sciences

Dr. Esayas Gebremichael, Assistant Professor of Geological Sciences

Dr. Elyse Zavar, Associate Professor of Emergency Management and Disaster Science, University of North Texas

Municipalities are increasingly participating in floodplain buyout programs to mitigate the risk and severity of flood damage. However, these buyout properties often remain vacant and underutilized. Planting trees in these areas could optimize ecosystem services, including carbon storage and sequestration, stormwater attenuation, air pollution removal, oxygen production, and urban heat island mitigation. The purpose of this research is to determine if planting trees within buyout sites would maximize ecosystem services. I measured 359 trees across a buyout landscape in Arlington, Texas, and calculated their ecosystem services using i-Tree Eco^{TM}. A geographic information system was created to

assess available planting space and model four tree planting scenarios. Scenario modeling indicated that planting at least 75% of the available space (504 trees) would optimize ecosystem services. The results of this research suggest that floodplain buyouts are not just effective for mitigating flood hazards; they also provide opportunities to maximize ecosystem services for local communities and municipalities.