ABSTRACT

Title of Dissertation: LEVERAGING FINE-SCALE GEOSPATIAL DATA TO ADVANCE BIODIVERSITY SENSITIVE URBAN PLANNING, WILDLIFE MANAGEMENT, AND GREEN CORRIDOR DESIGN: APPLICATION TO THE DISTRICT OF COLUMBIA

Annette L. Spivy, Doctor of Philosophy, 2023

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Typically, urban wildlife communities are made up of generalist species that are adept at utilizing human resources. However, many wildlife species struggle in the face of extensive urbanization and would benefit from increased conservation of urban green space, increased urban landscape connectivity, and proactive wildlife population management strategies. Unfortunately, maintaining and/or increasing the availability of quality habitat for biodiversity conservation in urban areas can be challenging as these conservation efforts are often influenced by the decreasing availability of critical resources and the challenges in allocating those resources among competing socioeconomic and environmental needs. Therefore, to improve the management and conservation of urban wildlife, accurate measurements of potential trade-offs between the environmental, economic, and social goals and management actions of a city’s sustainable development plan are needed. Until now, much of the effort in wildlife habitat modeling and

biodiversity mapping has been across large geographic areas or broad spatial scales. Those efforts have provided valuable insights into overall biodiversity patterns, identifying key hotspots, and understanding large-scale ecological processes. However, in urban environments, the dynamics of wildlife, habitat availability, and ecosystem services operate differently than in natural or rural landscapes. As urbanization continues to expand, there is a growing need to focus on fine-scale factors to address specific conservation challenges in urban systems.

This research seeks to address some of these challenges and demonstrates how new and traditional species-relevant geospatial datasets can be leveraged in urban planning and design to drive local-scale conservation decisions that put biodiversity in the forefront. This work links long-term, multi-taxon, wildlife survey data and high-resolution land use and land cover datasets (1m) to determine where high quality, well-connected habitats exist, or could most easily be justified and acquired, within the District of Columbia. This work also evaluates the spatial patterns of ecosystem service provisions across the urban landscape to identify "win-win" areas for conservation or restoration that will benefit both biodiversity and human wellbeing. Finally, the work evaluates a local translocation effort of the vulnerable eastern box turtle (Terrapene carolina carolina) to inform mitigation strategies when a sudden loss of habitat in an urban environment is inevitable. This research is particularly relevant to wildlife managers and urban planners in highly urbanized areas, where large parcels of land with suitable habitat are minimal and municipal environmental departments are often under-resourced. Local policymakers interested in incentivizing conservation efforts to meet state or national goals can use this information for strategic urban conservation initiatives.

LEVERAGING FINE-SCALE GEOSPATIAL DATA TO ADVANCE BIODIVERSITY SENSITIVE URBAN PLANNING, WILDLIFE MANAGEMENT, AND GREEN CORRIDOR DESIGN: APPLICATION TO THE DISTRICT OF COLUMBIA

by

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Dissertation submitted to the Faculty of the Graduate School of the
University of Maryland, College Park, in partial fulfillment
Of the requirements for the degree of
Doctor of Philosophy
2023

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Dedication

To my parents, Peter B. Spivy (1947 – 2012) and Christine A.M. Spivy (1951 – 1998) who taught me to never give up on my goals; and to my brother, Berton E. Spivy IV who stepped in and helped me pursue those goals after the loss of our parents.

Acknowledgements

First, I would like to thank my committee members (Jennifer, George, Karen, Janet, and Joe) for your continuous support and patience during my PhD study.

Many thanks to Lindsay Rohrbaugh and Daniel Rauch with the Fisheries and Wildlife Division of the Department of Energy and Environment of Washington, D.C., for providing the species survey data.

Thank you to the US National Arboretum staff who helped support the box turtle translocation efforts.

Thank you to my friends for your endless encouragement, advice, love, and support. A special thank you to Beth Gaston and Rachel Lamb for your enduring support and much-appreciated advice throughout my dissertation.

I am grateful for the many University of Maryland student volunteers who assisted with various projects along the way.

A big thank you to my incredible ENSP colleagues: Joanna, Angela, Caroline, and Mark who cheered me on throughout this journey.

Lastly, I want to thank my brother for your unwavering love, for supporting my goals, and always having confidence in my abilities.

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List of Abbreviations

AIRES	Artificial Intelligence for Ecosystem Services
ANOVA	Analysis of variance
C.C	Chesapeake Conservancy
CBD	Convention on Biological Diversity
CC	Climate change mitigation
CDIC	Carbon Dioxide Information Analysis Center
CGIAR-CSI	Consortium for Spatial Information
CRIAC	Clean Rivers Impervious Area Charge
D.C.	District of Columbia
DOEE	District Department of Energy and Environment
DPR	Department of Parks and Recreation
ES	Ecosystem Services
EU	European Union
FAO	Food and Agriculture Organization
FRM	Flood risk mitigation
GHG	Greenhouse gas
GIS	Geographic Information System
GME	Geospatial Modelling Environment
GPS	Global Positioning System
HM	Heat mitigation
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	the Intergovernmental Panel on Climate Change
LAC	Low activity clay
LCZ	Local climate zone
LiDAR	Light Detection and Ranging
LSCV	Least-squares cross-validation
LUCI	Land Utilization and Capability Indicator
LULC	Land use and Land cover
MA	Millennium Ecosystem Assessment
MCP	Minimum convex polygons
MSPA	Morphological Spatial Pattern Analysis Tool
NCPC	National Capital Planning Commission
NCRN	National Capital Region Inventory and Monitoring Network
NOAA	National Oceanic and Atmospheric Administration
NPS	National Park Service

NRCS	Natural Resources Conservation Service
PCA	Principal Component Analysis
SGCN	Species of Greatest Conservation Geed
SSURGO	Soil Survey Geographic Database
SWAP	State Wildlife Action Plan
SWP	State Wildlife Grant Program
SWR	Stormwater retention
TEEB	The Economics of Ecosystems and Biodiversity
TMDL	Total maximum daily load
TN	Total nitrogen
TP	Total phosphorous
TSS	Total suspended solids
UES	Urban ecosystem services
UGS	Urban green spaces
USDA	United States Department of Agriculture
USNA	United States National Arboretum

List of Publications and Professional Conference Talks

Chapter 2

Spivy, A., Murrow, J., Rohrbaugh, L., Rauch, D., Ossi, D. (2019). Methods and Strategies for Effective Urban Wildlife Conservation and Management: Application to the District of Columbia. in American Fisheries Society \& The Wildlife Society 2019 Joint Annual Conference.

Spivy, A. (2023). Prioritizing Local-Scale Conservation and Restoration Areas in Washington D.C, USA based on Habitat Connectivity and Development Risk. The International Urban Wildlife Conference.

Chapter 3

Spivy, A. (2023). Using Ecosystem Service Valuation and Assessments to Advance Biodiversity Sensitive Urban Planning and Corridor Design. The International Urban Wildlife Conference.

Chapter 4

Spivy, A., J. L. Murrow, and L. Rohrbaugh (2017). The Effects of Repeated Relocations on Range Establishment and Movement Metrics of Eastern Box Turtles (*Terrapene carolina carolina)* in a Highly Urbanized Environment. Society for Conservation GIS (SCGIS) July 2017, Monterey, California.

Poor, E. E., **Spivy, A**., Rohrbaugh, L., & Mullinax, J. M. (2020). An Ad Hoc Translocation of Urban Eastern Box Turtles (Terrapene carolina carolina). *Northeastern Naturalist*, *27*(4), 631-640.

Chapter 1: Introduction

1.1 Motivation and Background

The consideration of human-built ecosystems in the context of larger conservation and restoration priorities is critical given the world's urgent biodiversity crisis (Simonson et al. 2021). Not only do these ecosystems account for approximately 40% of the ice-free land on Earth, but scientists have argued that the novel assemblages of species that compose these urban communities may play a vital role in providing ecosystem services in the future (Hobbs et al. 2009). Urban environments are, perhaps, the best example of unique ecological systems because they differ most in composition and structure from their historical counterparts, and they represent novel combinations of species, stress, disturbance, structure, and function within a system (Tratalos et al. 2007; Kowarik 2011; Pickett et al. 2011). These unique combinations may possess novel ecological functions and interactions that can contribute to the provision of ecosystem services (Ksiazek-Mikenas et al 2023). As environmental conditions change, certain species within these novel communities may prove to be especially adept at providing specific services, making them vital components for sustaining human well-being in the future (Kunz et al. 2011; Kabisch et al. 2016; Ziter 2016). In the face of climate change and other human-induced disturbances, ecosystems will experience altered conditions, such as species distribution shifts. In such scenarios, ecosystems that are flexible and capable of adapting to changing conditions will be more resilient. Urban communities, with their unique combinations of species, may possess a higher degree of adaptability, enabling them to persist and provide essential ecosystem services in a future that may be considerably different from the past. In this regard, idealistic restoration goals often aim for ecosystems in a pristine or unaltered state. While aiming for pristine conditions is valuable in certain cases, it may not always be feasible or desirable, especially in heavily human-modified landscapes or ecosystems that have undergone significant changes. Focusing solely on idealistic restoration may overlook the potential of these communities to adapt and provide valuable ecosystem services in a changing world (Choi et al. 2008). Conservation and restoration efforts thus need to evolve together with the changing landscape. Furthermore, conservation and restoration of novel ecosystems may prove to be vital in the safeguarding of biodiversity at the genetic, species and ecosystem level (Bridgewater et al. 2011).

Greater than half of the world's population, and >80% of the US population now lives in urban areas (Grimm et al. 2000, United States Census Bureau 2010). Urbanization and increasing human population growth has had a particularly negative impact on wildlife globally, mainly through the loss and fragmentation of natural areas by roads and development (Gibbons et al. 2000; McKinney 2002; McDonald et al. 2008; Laurance et al. 2014). We are now amid a global extinction crisis (Dirzo and Raven 2003) and as habitat loss and fragmentation continue, wildlife populations will continue to be divided and isolated in remaining natural areas, creating particularly unique challenges for dispersal, mating, and foraging in urbanized environments (Hamer, A. J., & McDonnell; Marzluff and Ewing 2008). While some species may survive and even thrive in these urban environments, others, such as those that have limited mobility to seek refugia, may need human intervention to persist. Because the majority of wildlife populations are negatively impacted by habitat loss and fragmentation, managers are forced to make difficult decisions regularly. Proactive management strategies will become necessary for wildlife populations which are small, ecologically threatened, or geographically restricted (Watson and Watson 2015).

Urban wildlife managers and population ecologists often invest a great deal of effort in devising methods to increase, conserve, and restore urban greenspace because greenspaces, with natural structures, can be important for maintaining high ecological diversity (Aronson et al. 2014; Beninde et al. 2015; Ives et al. 2016; Lepczyk et al. 2017). Furthermore, these green spaces are utilized by a variety of wildlife species as alternative habitats that provide food, water, shelter, and protection from the elements and predators. They also function as corridors that facilitate species dispersal (Bolger et al. 1997; McIntyre et al. 2001). When considered on a landscape scale, urban greenspaces can function as steppingstones that enable species migration and dispersal across a fragmented environment (Sandström et al. 2006). Moreover, several studies have indicated that natural corridors increase the movement rate of plants and animals into otherwise isolated fragments, thus offsetting local extinctions by more frequent colonization (Bennett 1999; Haddad and Baum 1999; McIntyre et al. 2001). Sufficient urban greenspaces can support target species of greatest conservation need (SGCN) and permit the presence of IUCN Red listed species (Mortberg and Wallentinus 2000). Accordingly, many cities have sustainable development plans that call for actions to provide or increase access to greenspaces; preserve wildlife, and landscapes; ensure the resilience of the coupled human-environment system; and encourage residents to value the benefits of a healthy relationship with natural resources and the environment. As such, the conservation and restoration of greenspaces within metropolitan areas is particularly urgent.

While there has been significant investment in sustainable land use planning and species conservation occurring in major metropolitan areas across the US, this planning does not readily consider habitat connectivity and structure (Hardy et al. 2022). The often-insufficient consideration of habitat connectivity and structure in the design of human-dominated landscapes can be contributed to the prioritization of human needs, fragmented decision-making, lack of ecological awareness and expertise among planners, as well as limited access to comprehensive ecological data and information on wildlife corridors, biodiversity hotspots, and habitat connectivity. Furthermore, maintaining and increasing the availability of high-quality habitat for wildlife conservation in urban areas can be difficult due to the limited supply and growing development demands on economically valuable land. The competition for available land becomes intense as the demand for housing, infrastructure, commercial spaces, and other development projects increases. The lack of suitable land can hinder efforts to create or maintain large and connected wildlife habitats. Moreover, these conservation efforts are often influenced by the availability of critical resources and the allocation of these resources among competing social, cultural, and environmental needs (Shwartz 2014; Aronson et al 2017). Subsequently, promoting biodiversity-inclusive cities requires successful and proactive land and species management strategies, and a commitment to integrating biodiversity considerations into urban planning and decision-making is vital for achieving sustainable, resilient, and thriving cities.

1.2 Research Framework and Objectives

To address these challenges, this dissertation seeks to answer the following research question: How can fine-scale species-relevant geospatial datasets be leveraged in urban planning and design to drive local-scale conservation decisions that prioritize biodiversity needs? This work leverages traditional wildlife surveys, existing landscape and network connectivity tools, ecosystem valuation models, as well as readily available spatial analysis tools to prioritize local-scale conservation and restoration areas based on habitat connectivity, development risk, and ecosystem service provision. The landscape and ecosystem service metrics, derived from using fine-grained spatial datasets that are more closely related to local management practices, will enhance, and inform, the science-policy interface of urban planning. Moreover, combining these datasets with long-term surveys of species presence can inform subsequent conservation and management decisions, allowing researchers to accurately identify areas that are more vulnerable to species loss and prioritize conservation and management efforts accordingly.

Figure 1-1 illustrates the conceptual framework for this research, which involved a three-step process to 1) assess urban species diversity and related habitat metrics and map potential conservation corridors using local species surveys and high resolution land use/land cover datasets, 2) quantify the biophysical value of multiple urban ecosystem services (UES) and attribute these services to potential conservation corridors to identify "win-win" areas for conservation and restoration that benefit wildlife and humans, and 3) examine the effectiveness of repeated relocation as a local wildlife management strategy when high quality habit for a vulnerable species is inevitably lost.

All components of this research employed spatially explicit analyses and models and used a Geographic Information System (GIS) as well as the open-source software R as the statistical engine (version 4.0.4; R Core Team 2021). Results from step 1 facilitated the completion of step 2 and assisted in determining where current high-functioning, well-connected habitats exist, or could most easily be justified and acquired, within the District of Columbia (D.C.). The local case study in step 3 resulted from efforts to mitigate the impending, sudden loss of habitat for a vulnerable species in an urban environment.

In Chapters 2-4, this conceptual framework is subsequently applied to answer three supporting and related research questions:

- Question 1: How can local-scale conservation and restoration efforts be improved in urban planning processes in Washington D.C. using a combination of high-resolution land cover data and local wildlife survey data?
- Question 2: Where are viable conservation areas that jointly maximize biodiversity protection and the supply of ecosystem services in Washington D.C.?
- Question 3: Can repeated relocation a vulnerable species be a feasible and successful mitigation strategy when the sudden loss of habitat in an urban environment is inevitable.

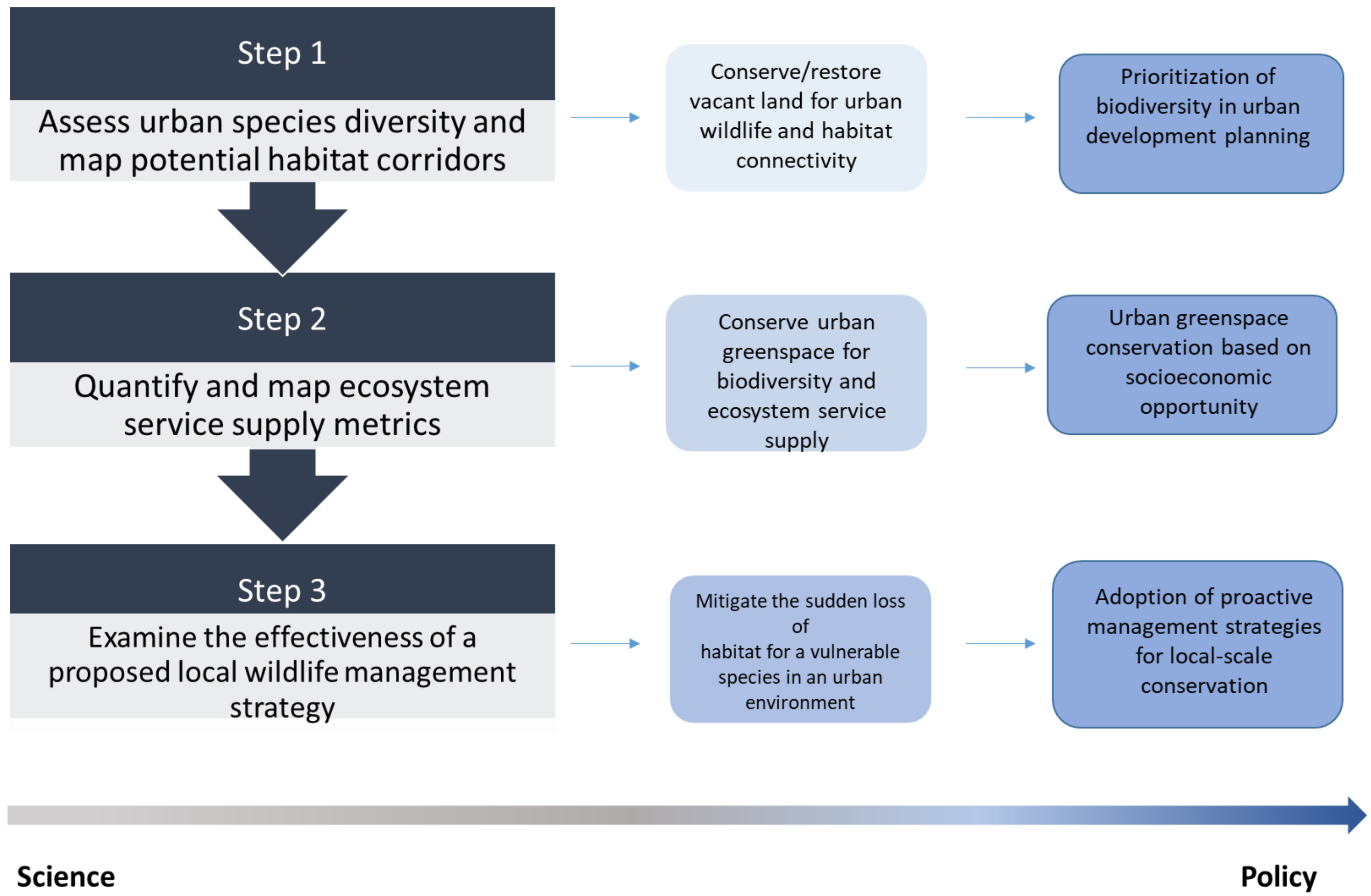

Figure 1-1: Conceptual framework illustrating the three-step process of scientific advancement to potential policy/management application.

An additional goal of this project is to demonstrate the policy-relevance of using new and traditional species-related geospatial datasets in urban biodiversity conservation research. While the methodology and results presented across the following three chapters will directly assist D.C. managers, planners, and decision-makers in managing and conserving the district's diverse wildlife populations and their habitats, these findings will contribute to applied conservation and management strategies in any highly urbanized area where large parcels of land with high-quality/ suitable habitat is minimal. Furthermore, this work can provide policymakers and planning officials with the necessary tools and information to make clear and informed decisions on how to allocate scarce dollars more efficiently. Lastly, local policymakers interested in incentivizing conservation efforts to meet broader state or national goals, may be able to use this information for strategic urban conservation initiatives, that account for both biodiversity and socioeconomic opportunity.

1.3 Dissertation Outline

This work is presented across three core chapters (2-4). Specifically, Chapter 2 explores the value of linking long-term, multi-taxon, wildlife survey data and high-resolution (1-meter) land use and land cover datasets in urban planning and corridor design. Chapter 3 investigates the economic tradeoffs of ecosystem services and wildlife protection by attributing socioeconomic value to the habitats identified in chapter 2 to strengthen consideration of conservation efforts in a time of competing demands for scarce public funding. Chapter 4 assesses a unique mitigation strategy when the sudden loss of quality habitat for a vulnerable species in an urban environment is impending. Chapter 5 concludes with a summary of the main findings, limitations, and areas of future research. Additional supporting analysis, figures, and data can be found in the Appendices.

Chapter 2: Linking Wildlife Data and High-Resolution Land Use/Land Cover Datasets to Identify Potential Urban Conservation and Restoration Areas.

Abstract

Proactive urban land-use planning that encourages both the conservation of greenspaces and the restoration of vacant lots enables effective biodiversity corridor design and early identification of development risk. Given the often-small property sizes in urban areas, fine-grained data that relates to local management practices are necessary for progressive urban planning. Here, I developed and applied a framework that puts wildlife biodiversity at the center of urban planning and corridor design. Using data from the Washington D.C. metro area, I quantified and mapped the spatial distribution of species of greatest conservation need (SGCN) across three taxa (birds, mammals, and reptiles/amphibians) from January 2009 to November 2019. Next, I calculated Hill numbers for SGCN richness (q0), Shannon diversity (q1), and Simpson diversity (q2) across the 27 public parks where SGCN were present and analyzed the associated habitat characteristics using high-resolution land use/land cover datasets ($1m^2$). Then, the habitat characteristics that were significantly correlated to SGCN biodiversity across all three taxa, principally core forest with openings, were used to identify and map potential habitat corridors between the 27 surveyed parks using the Linkage Mapper toolbox within ArcGIS®. Within these corridors, I identified 138 vacant lots considered at highest risk for gray infrastructure development in 20 years. The lots ranged in size from 48 m^2 to 165,645 m^2 with >78% of all lots located south of the Anacostia River (Ward 7 and 8). Of the 138 lots, only 28 were government owned, operated, or managed. My results demonstrated the value of

coupling site-specific species data and high-resolution land use/land cover data to inform biodiversity sensitive urban planning and corridor design. Identifying target restoration and conservation sites in an urbanized landscape can drive more effective wildlife conservation and management at local levels. This is especially relevant to cities with progressive re-greening policies like the District of Columbia's new Green Area Ratio requirement in many zoning areas.

2.1 Introduction

Biodiversity loss poses a significant threat to human well-being through the corresponding loss of critical ecosystem goods and services (Chapin et al. 2000; Duffy 2003; Hooper et al. 2005; Díaz et al. 2006; Haines-Young and Potschin 2005; Cardinale et al. 2012). To help maintain biodiversity and improve resilience to ongoing global environmental change, it is vital that natural systems are protected and restored. An array of global initiatives and policy frameworks, such as the Convention on Biological Diversity (CBD), have developed goals and targets aimed at halting the loss of biodiversity through national and sub-national level action. Most recently, the United States took executive action to commit to conserving at least 30% of U.S. lands and oceans by 2030 to "stem the extinction crisis, safeguard water and food supplies, absorb carbon pollution, and reduce the risks of future pandemics and other global health emergencies" (USDOI 2021, p.10). Under this new goal, called the 30x30 Goal, considerable attention has been given to expanding existing protected areas with large contiguous habitat features. However, emphasis on conservation and restoration efforts across smaller and highly fragmented spatial domains will be essential for species movement and species' adaptation to a rapidly changing environment (Lepczyk et al. 2017; Lindenmayer 2019).

Species of greatest conservation need (SGCN) are important indicators of the health and diversity of a city's ecosystems. They are identified and listed in state wildlife action plans, regional biodiversity assessments, or conservation strategies to prioritize conservation efforts for species facing significant threats. SGCN often serve as indicator species, meaning their presence, absence, or abundance can provide valuable insights into the overall health and ecological condition of an area. Their decline or disappearance may indicate environmental degradation, habitat loss, or other threats affecting the broader biodiversity of the city. Identifying SGCN can also have legal and policy implications, leading to the development of regulations and protections to safeguard these species and their habitats. This can contribute to broader biodiversity conservation efforts within the city. Studying SGCN can provide valuable data for scientific research and monitoring programs. Monitoring changes in the abundance and distribution of these species over time can help assess the effectiveness of conservation actions and inform adaptive management strategies. By prioritizing the conservation of these species, cities can take proactive steps to protect their natural heritage, enhance ecological resilience, and foster a harmonious coexistence between urban development and biodiversity conservation.

The identification of structurally significant habitat corridors is typically based on focal species habitat use. While there has been an increase in urban biodiversity research over the past 30 years, there is a growing need to expand sampling efforts and broaden taxonomic representation to gain a more thorough understanding of urban biodiversity, especially as cities are increasingly recognized as biodiversity refugia (Hayward et al.

2015; Spotswood et al. 2021; Rega-Brodsky et al. 2022). While increasing the number of urban species sampled and increasing taxonomic representation would be ideal for future urban biodiversity studies, this can be an intensive endeavor in terms of cost and resources. In fact, financial and resource constraints are common in wildlife management programs (Hobbs & White 2012; Walls 2018) and can oftentimes result in unequal sampling effort in urban wildlife surveys. This occurs when different areas or habitats within a study area are being sampled with different levels of intensity which can result in a bias towards more heavily sampled areas and can lead to underestimation of species richness and biased diversity estimates (Brose et al. 2003; Reddy & Dávalos 2003).

In urban areas, the conservation of and restoration of small spaces and unconventional habitats is particularly urgent due to ongoing development pressures. Recent trends in urban land-use planning have centered on the potential benefits of conserving vacant lots for meeting critical social, environmental, and wildlife needs (Arendt 1996; Hostetler 2009; Rupprecht 2015; Villaseñor et al 2020). These benefits include equitable access to green space, improving the quality of local watersheds and airsheds, maximizing stormwater benefits, and increasing their value for wildlife when considering the urban-to-rural gradient (Pickett et al. 2011; Hostetler et al. 2009). Given the often-small property sizes in urban areas, fine-grained data that relate to local management practices are necessary for insightful and effective urban planning. Fortunately, the advent of high-resolution remote sensing data provides an opportunity for improved city-wide habitat connectivity mapping and, if appropriately coupled with species survey data, can uniquely inform local habitat restoration efforts.

Least-cost path analysis, as applied to species mobility, is a common Geographic Information System (GIS) technique used to identify the best potential networks of greenways that reduce conflict and increase co-benefits (Adriaensen et al. 2003; Lynch 2019; Balbi et al 2019; Balbi et al. 2021). In urban areas, high-resolution land cover data can greatly improve the fine-scale applications of the animal survey-corridor strategy (Boyd and Foody 2011; Boyle et al. 2014). The animal survey-corridor strategy is an approach used in wildlife conservation to assess and facilitate the movement of animals across landscapes. It involves the identification and creation of corridors or pathways that allow animals to move between fragmented habitats in an otherwise urbanized or human-dominated environment. These corridors help address the challenges of habitat fragmentation and provide connectivity for wildlife populations, promoting genetic diversity, species dispersal, and overall ecological health. By jointly leveraging these datasets and methodologies, city planners can make informed and evidence-based decisions about how to best conserve and manage their natural resources for the benefit of both wildlife and humans as well as better inform existing property owners about the ecological value of their property and its greenspace.

Identifying strategic areas for habitat connectivity within local comprehensive plans could trigger investment in critically important conservation and restoration initiatives (Sandström et al. 2006). In addition to protecting areas of existing high value for wildlife, proactive habitat corridor delineation as part of ongoing development planning has yielded higher levels of landscape connectivity and improved habitat utilization (Parker et al., 2008; Benedict and McMahon 2012; Bartuszevige et al. 2016; Brodie et al. 2016). Furthermore, giving biodiversity an ‘active’ role in the urban development/redevelopment planning processes is essential for enabling biodiversity inclusive cities (Hernandez-Santin, Bekessy and Desha 2023) and achieving national and global biodiversity targets. Although major US cities, including Washington, D.C., have made significant investments in species conservation and sustainable land use planning, there is still a growing need for urban land-use policies that encourage targeted conservation of small urban greenspace and restoration of vacant lots. As urbanization continues to expand and encroach upon natural areas, preserving and enhancing these smaller pockets of greenery within cities become crucial for biodiversity conservation, ecosystem services, and overall human well-being.

Small urban greenspaces, such as pocket parks, community gardens, and green corridors, can serve as refuges for native plant and animal species. These spaces may host a variety of wildlife, including birds, insects, and small mammals, contributing to urban biodiversity and ecological resilience. Even modest greenspaces can provide essential habitat for urban wildlife. By encouraging targeted conservation of these areas, urban land-use policies can support breeding sites, nesting locations, and feeding areas for various species, helping maintain wildlife populations in the urban environment. Just as pertinent, greenspaces and restored vacant lots can contribute to a city's climate resilience. Vegetated areas help moderate temperatures, reducing the impact of extreme heat events, and providing cooling effects in densely built urban areas. Additionally, access to green spaces has been linked to improved mental health, reduced stress levels, and increased physical activity for urban residents. Targeted conservation and restoration efforts in proximity to residential areas can create opportunities for recreation and leisure, enhancing the overall well-being of city dwellers. Urban land-use policies that encourage targeted conservation and restoration can address environmental inequities by ensuring that underserved neighborhoods have access to green spaces and the associated benefits they provide. Furthermore, implementing land-use policies that promote the conservation of and restoration of small urban greenspaces often involve community engagement and empowerment. Involving residents in decision-making processes fosters a sense of ownership and pride in these spaces, leading to better stewardship and long-term sustainability. Lastly, collaboration between public entities, private developers, and community organizations can lead to innovative approaches in conserving and restoring urban greenspaces. Public-private partnerships can leverage resources, expertise, and funding to implement successful conservation and restoration projects. For example, Washington, D.C. has implemented the Green Area Ratio (GAR) as part of its environmental regulations for new development projects. The Green Area Ratio is a unique and innovative approach to promote green spaces, vegetation, and ecological sustainability within the city's urban fabric. The GAR is a metric that quantifies the amount and quality of green space incorporated into a development site relative to its total area. It considers both the quantity and quality of vegetation and other green infrastructure elements, encouraging a comprehensive approach to sustainable and eco-friendly urban development.

The objective of this study was to determine how high-resolution spatial data, coupled with traditional species survey data, could be used to facilitate biodiversity sensitive urban planning, and inform more effective habitat conservation and management at a local scale. As such, I linked site-specific multi-taxa wildlife survey data with high-resolution land use and land cover (LULC) datasets (1m), to identify and map required structural habitat characteristics for urban biodiversity and target potential local-scale conservation and restoration areas that maximize habitat connectivity. My study followed four steps: (1) quantify the diversity of species of greatest conservation need (SGCN) within Washington D.C. public green spaces, using a longitudinal presence-only dataset of 274 terrestrial vertebrate species covering three taxonomic groups (birds, mammals, and reptiles/amphibians); (2) identify the structural habitat characteristics that are correlated with the most SGCN diverse parks using high-resolution geospatial data; (3) map least-cost corridor linkage pathways between these biodiverse core parks informed by significant habitat characteristics, and (4) identify and prioritize vacant lots for conservation or restoration. Lastly, I discuss the potential role of my candidate sites in supporting local environmental initiatives. This study's approach can be leveraged across major U.S. cities to help guide the integration of biodiversity considerations within urban planning and corridor design and better align ongoing city planning, tree canopy initiatives, and urban agriculture initiatives with specific biodiversity habitat metrics and goals.

2.2 Data and Methods

2.2.1 Study Area

This study was focused on the well-known metropolis, Washington, District of Columbia (D.C.): a 178.7 km² land area at the junction of two tidal rivers, the Anacostia and Potomac, which provide valuable habitat for many wildlife species (DOEE 2019). D.C. is also home to over 705,000 people (Ossi et al. 2015). The patterns of land use in D.C. are complex; with a 10.3 km² highly developed core surrounded by an inner ring of moderate to high-density residential and mixed-use neighborhoods. Beyond the inner ring is an outer ring of less dense development, characterized largely by single family housing and garden apartments (NCPC 2021).

The Trust for Public Land has identified D.C. as having the third highest-ranking park system in the country with more than 6,700 acres of land protected as U.S. National Parks and 900 additional acres of District-owned Park land (DOEE 2019). The D.C. government is committed to protecting the natural areas while also providing all residents convenient access to nature and green places. In 2016, Mayor Bowser signed the "Fisheries and Wildlife Omnibus Amendment Act of 2016," to help protect critical wildlife habitats and better manage invasive species, as well as the "Tree Canopy Protection Amendment Act" that discourages the removal of healthy, mature trees. Within the District government, the District Department of Energy and Environment (DOEE) is responsible for the conservation and management of all species of wildlife and their habitats. D.C.'s State Wildlife Action Plan (SWAP) was last updated in 2015 and serves as a comprehensive 10-year city-wide plan and framework aimed at conserving and protecting Washington, D. C.'s wildlife and their habitats, consisting of forests, waters, meadows, and wetlands. The city's protected areas currently provide habitat for approximately 240 species of birds, 78 fish species, 29 mammal species, 21 reptile species, 19 species of amphibians, and thousands of invertebrates (Ossi 2015; DOEE 2019).

2.2.2 Quantifying Biodiversity from Species Survey Data

The wildlife observation data used in this analysis resulted from fifteen years of research, inventory, and monitoring of the District's wildlife, as supported by D.C.'s SWAP. This work was funded by the State Wildlife Grant Program (SWP), which was created by the Department of the Interior (DOI) under the Appropriations Act of 2002, Title I, Public Law 107-63. I used site-specific survey data that was collected annually across 457 sites in 27 public parks (Figure 2-1) for 11 years (2009 – 2019) for 20 species of amphibians, 10 reptiles, 202 species of birds, and 18 species of mammals (Table 2-1). All data were collected by biologists within the DOEE. Species of greatest conservation need (SGCN) were identified based on information on the distribution and abundance of each species, including low and declining populations, as well as species that were considered indicative of the diversity and health of the area's wildlife (Ossi et al. 2015).

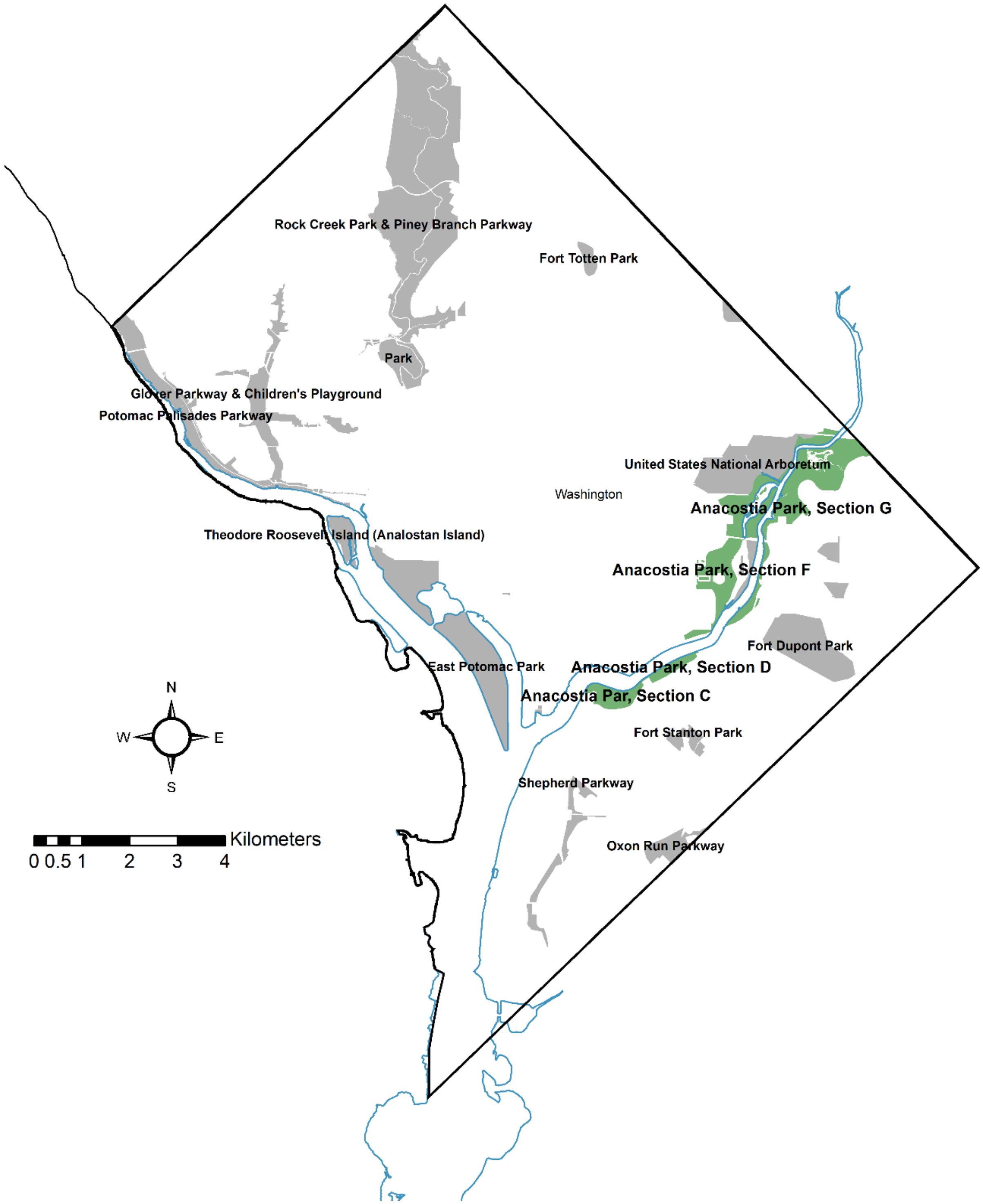

Figure 2-1: Map depicting the 27 public parks containing the 457 terrestrial vertebrate survey sites within the District of Columbia. Anacostia Park is separated into 4 management sections (C, D, F,G) designated by the National Pak Service (green).

Table 2-1: Number of species observed, number of survey locations, and species sightings for each taxon collected in the District of Columbia from 2009 to 2019.

Taxon	Years surveyed	Parks	Survey Sites	Species (Richness)	Sightings	SGCN Parks	SGCN Species	SGCN Sightings
Birds	2009-2019	28	290	202	92555	26	50	3115
Mammals	2009-2016, 2018	20	44	18	3773	17	9	684
Reptiles and Amphibians	2009-2019	16	123	30	3620	16	26	2569
Totals			457	250	99948		85	6368

Species were surveyed using a variety of inventory and monitoring techniques based on the taxon of interest. Bird protocols, except for waterbird surveys, were conducted by five-minute point counts at selected locations during which all birds seen or heard within an estimated 50m radius were recorded. All point counts occurred within four hours of sunrise, and when standard weather parameters were met. For waterbirds, distance was measured beyond the 50m radius, to include all birds viewable by a spotting scope, regardless of weather. Mammals, reptiles, and amphibians were surveyed using one of the following methods: live trapping techniques (pitfall and small mammal traps), mark/re-sight/recapture, telemetry, remote cameras, and scat analysis.

To derive habitat characteristics associated with core biodiverse areas, I utilized effective rapid assessment indicators, such as species richness and species diversity indices, to first quantify SGCN biodiversity within the 27 surveyed parks. To address any potential sample bias and to facilitate the comparison of biodiversity data between surveyed parks, I used the package *iNEXT* in program R (Hsieh et al. 2016; program R version 4.0.4; R Core Team 2021) to calculate the effective number of species including Hill numbers of species richness (q0), Shannon diversity (q1), and Simpson diversity (q2) (Hill 1973; Chao et al. 2014). Hill numbers are a mathematically unified method for measuring species diversity (Chao, Chiu & Jost 2014) that standardize samples on the basis of sample size or sample completeness by integrating curves based on sampling theory that smoothly link rarefaction (interpolation) and prediction (extrapolation) (Chao et al. 2014). They account for both the number of species and their relative abundance in a survey, based on the idea that some species may be more common or rarer than others in a given area. I calculated all diversity estimates and associated 95% confidence intervals via bootstrapping with 100 repetitions (Chao et al. 2014). These calculations were done for SGCN within each taxonomic group and for all SGCN together (regardless of taxon). Lastly, I graphed the Hill numbers (q0, q1, q2) for the SGCN groups using sample-size-based rarefaction and extrapolation (R/E) (Chao & Jost 2012). The R/E curves for q0, q2, and q3 were extrapolated to double the reference sample size, guided by an estimated asymptote (Hsieh et al. 2016). Analysis and results of Hill numbers for all species, including non-SGCN, across the three taxa can be found in Appendix A.

To visualize the relationship between habitat structure and all SGCN taxa, I identified a subset of parks (n= 9) where SGCN from all three taxa were present. To verify whether this subset of nine parks was an adequate representation of SGCN diversity, and thus SGCN habitat use, I graphed the Hill numbers (q0 and q2) and analyzed dataset sample completeness. I then performed a Kruskal Wallis (1952) Rank Sum Test (Hollander, Ma, and Wolfe 2013) using the *stats* package in program R (version 4.04) (R Core Team and contributors worldwide 2020) to identify whether significant differences existed in Shannon diversity (q1) of SGCN across the parks. To determine which of the nine parks produced statistically significant Shannon biodiversity metrics (q1), I performed a post-hoc Dunn's Test (Dunn 1961) using the Benjamini-

Hochberg (B-H) adjustment method (1995) which controls the expected proportion of falsely rejected hypotheses or the false discovery rate (FDR). This method adjusts the FDR which helps to control the fact that sometimes small p-values (less than 5%) happen by chance, which could lead to incorrectly rejecting the true null hypotheses that there is no statistical significance in biodiversity between the parks.

2.2.3 Identifying Significant Land Cover Characteristics

It is widely recognized that habitat and forest structure is a major predictor of both mammal and avian biodiversity (MacArthur and MacArthur 1961, Wiens 1974, Willson 1974, Tews et al. 2004; Pardini et al 2005) and landscape and local-scale habitat structure were determined to be the most significant for influencing reptile and mammal assemblages (Garden et al. 2010). Studies have also suggested an interaction between habitat type and alteration on the diversity and composition of amphibian assemblages (Ray, Lehmann, and Joly 2002; Jongsma et al. 2014). Therefore, I quantified a variety of habitat features using indicators of both the horizontal (composition and configuration) and the vertical (vegetation) aspects of the landscape (Culbert et al. 2013). First, I used the 2017 Chesapeake Conservancy's high-resolution ($1m^2$) land use and land cover dataset (CPBO 2022; Pallai & Wesson 2017), hereafter referred to as "CC", to create structural diversity metrics within the surveyed parks and across D.C. (Table 2-2). Then, to quantify horizontal forest structure in the 2017 CC dataset, I used the Morphological Spatial Pattern Analysis Tool (MSPA) (Soille and Vogt 2009) in GUIDOS (Graphical User Interface for the Description of image Objects and their Shapes) (Vogt and Riitters 2017) to generate landscape configuration classes representing forest pattern, forest connectivity, and area of openings enclosed by forest. Unlike many patch-based metrics, MSPA can account for the spatial configuration of patches in the landscape, which is an essential aspect in habitat mapping and extremely important to urban planners and policymakers (Soille & Vogt 2009). To classify basic forest structure elements, I considered forest as “foreground”, low vegetation, shrub, barren, and wetlands as “background”, and all impervious surfaces and structures, water, and tree canopy over impervious surfaces were considered “missing and ignored.” The first step in MSPA processing is to identify the core, which is done using the connection rule for neighbors and the value for edge width (Soille and Vogt, 2009). The number of pixels designated as core and the minimum size of core are both impacted by edge width. Increasing the width of the edge will increase the non-core area at the expense of the core-area. Due to CC's 98% accuracy in characterizing tree canopy in the high-resolution LULC dataset, I reduced edge width to avoid reducing regions designated as forest (Pallai and Wesson, 2017). Therefore, I used eight-neighbor connectivity (8 cell) and set the edge width value to 1 cell and the transition zone value to 1 cell (Vogt 2022). The resulting forest structure elements identified by MSPA were total forest area, integral forest area (forest + openings within forest), core forests [(large$\geq$4600 m^2, medium $\geq$1000m^2 and < 4600m^2, and small <1000m^2), openings in core forests, perforation (the transition zone between core forest and opening in core forest), and edge (transition zone between core forest and core non-forest) (Table 2-2). In addition to the MSPA generated features, I calculated two additional metrics: 1) percent low vegetation/shrub within each park in ArcMap® (Version 10.8) and 2) percent wetlands within each park using a circular neighborhood radius of three cells. Because the level of diversity of land use types within an area is an important aspect of urban and spatial planning (Song, Martin & Rodriguez 2013; Zhang & Zhao 2017; Mavoa et al. 2018), I calculated an entropy index as

$$Entropy = -\left(\left[\sum_{j}^{k} = 1^{p^{j}\ln p^{j}}\right]\right) / \ln k$$

where p^j (calculated in terms of area) is the percentage of each land use type, j the area, and k is the total number of land use types (Song et. al., 2013; Voukenas 2021). The entropy index was calculated using the raster calculator in ArcMap®, using the 2017 CC dataset.

Table 2-2: Dataset, and associated geospatial tools used for calculating habitat characteristics. A subset of these metrics was then used to drive corridor modeling efforts.

Habitat Metric	**Dataset**	**Tool**
Total Forest Area	Chesapeake Conservancy 1m Land Use (CC)	Morphological Spatial Pattern Analysis
Integral Forest Area (forest + openings within forest)		
Large Core Forest (%) (<1000m)		
Medium Core Forest (%) (>1000<4600m)		
Small Core (Forest (%) (>4600m)		
Islet (% of Forest Area)		
Edge (% or forest)		
Number of Perforations		
Perforation (% of Forest Area)		
Openings in core forest		
Forest Integrity		

Canopy Height (mean)	LiDAR	ArcGIS 10.8
Percent Impervious (mean)	CC	
Percent Low Vegetation/Barren/Shrub/Wetlands	CC	
Slope (mean)	LiDAR	
Wetland (% of Park)	CC/Wetland Inventory	
Land Use Entropy	CC	

Lastly, I calculated mean canopy height from the 2018 LiDAR Classified LAS Point Cloud dataset in ArcMap (version 10.8). These lidar data were processed, classified LAS Specification version 1.4, Quality Level 2 covering the District of Columbia (ASPRS 2019). I used the *stats* package in program R (version 4.04) to conduct a Principal Component Analysis (PCA) on the habitat metrics calculated for nine parks where all three SGCN taxon groups were present. The PCA condensed my calculated habitat metrics and identified which habitat characteristics were most related to presence of all three taxonomic groups of SGCN in one location. Next, to determine which of those habitat metrics were significantly correlated to the number of bird SGCN, the number of mammal SGCN, the number of reptile/amphibian SGCN, and total number of SGCN regardless of taxa, I performed a pairwise correlation test using the *PerformanceAnalytics* package in program R. I used Pearson's correlation coefficient (r) where a higher value indicates stronger correlations and significance was considered as $p \leq 0.05$. The habitat variables that most supported all SGCN taxa and were significantly correlated to SGCN biodiversity were used to develop the cost surface in the corridor model.

2.2.4 Mapping Least Cost Corridors

I used the Linkage Pathways tool in the Linkage Mapper 3.0.0 toolbox in ArcGIS 10.8 (McRae and Kavanagh 2011, McRae 2012, Gallo and Greene 2018) to map the best movement routes (linkages) between the surveyed parks. First, I created a 1-meter resolution cost-resistance map across the D.C.-wide LULC layer, with a 100-meter buffer to account for potential greenspace outside of D.C.'s boundary. Feature classes included forests, low vegetation, wetlands, tree canopy over impervious, and impervious surfaces.

To account for potential corridors that may cross the jurisdictional boundary between D.C. and Maryland, I applied a 100-meter buffer when mapping the least cost paths between the surveyed parks. We assigned equally increasing resistance values for each LULC based on their representative value to SGCN. Forests were all allocated the lowest resistance because it supported all SGCN taxa and because of its significant correlation with SGCN diversity within all three taxonomic groups. Low vegetation (including shrub/scrub) was given the next lowest cost because it was an important habitat feature to all SGCN taxa. Tree canopy over impervious and impervious surface and structures were assigned the highest resistance values (Table 2-3). Next, I used the Linkage Pathways tool to connect each of the 27 surveyed parks to their four nearest neighbors using cost-weighted distance to identify the lowest cost corridors. After nearest neighbors were identified, additional links were added to connect the nearest pairs of otherwise isolated groups of core parks.

Table 2-3: Assigned Resistance Scores used to create the cost surface input layer for the linkage pathways model. Scores were prioritized based on the significance of habitat structure to SGCN parks and SGCN diversity in D.C

Resistance Score	**Landscape Characteristic**
1	Total Forest Area (Small, Medium, and Large Core Forest with Openings)
2	Low Vegetation (including shrub)
3	Wetlands
4	Tree Canopy over impervious
5	Impervious Surface and Structures

Finally, to create linkage maps focused on zones most relevant for future conservation and restoration, I truncated the normalized corridors. In contrast to the functionality of corridors, narrow linkages suggest a greater ability to identify exact locations in the landscape that are of greatest importance for connection or restoration. This is particularly true when passing through a high-resistance landscape, as is typically seen in urban areas (McRae 2012; WHCWG 2010). As such, I chose values from zero to 100 cost distance units. The resulting corridors were summarized by total land area and land cover class (forest, shrub, low vegetation, barren, impervious/ tree canopy over impervious surface).

Most connectivity analyses have focused on conserving areas that facilitate animal movement, but few studies have focused on identifying landscape features that impede movement between ecologically important areas (McRae 2012). Thus, to complement my least-cost corridor analysis, I used the Barrier Mapper tool, which is part of the Linkage Mapper toolbox to locate barriers that strongly reduced the structural connectivity of my surveyed parks within my established linkage zones. I used a 10-meter (10 pixel) search diameter to detect restoration opportunity areas equal to or less than 10m wide (e.g., roads). These additional restoration areas were candidates for potential implementation of wildlife-friendly overpasses or underpasses between parks with high biodiversity.

2.2.5 Prioritizing Conservation and Restoration Sites within Corridors

To identify targeted areas for conservation and potential restoration within my mapped linkage pathways, I used the D.C. generalized dataset of existing land use (OCTO 2022) to locate all parcels of land with an existing designation of 'vacant'. A vacant lot is considered abandoned, vacant, empty, or a neglected parcel of property with no buildings. Then, I assigned each vacant lot a future land use designation based on D.C.'s 2021 Future Land Use dataset which depicts intended land use designations approximately 20 years in the future. This dataset represents the revised Comprehensive Plan Generalized and Future Land Use Maps from D.C. Law 24-20 (Comprehensive Plan Amendment Act 2021), which became effective on August 21, 2021. Next, I calculated the percentage of each vacant lot that was forested using the 2017 CC land use dataset. Specifically, forest in the 2017 CC dataset was defined by Claggett et al. (2017) as "all contiguous patches of trees ≥1 acre in extent with a patch width ≥240-ft somewhere in the patch: the 240-ft girth references potential altered microclimate conditions extending inwards up to 120-ft from the patch edge; the forest understory is assumed to be undisturbed/unmanaged; and forests that are also wetlands are included in this class." Then, I designated all currently vacant lots that are ≥ 90% forested as potential conservation sites and designated all currently vacant lots that are < 90% forested as potential restoration sites. I utilized the 90% threshold because the parks with the highest SGCN richness and/or diversity had greater than 90% forest cover. Finally, I assessed the risk of future gray infrastructure development for all potential conservation and restoration sites by attributing projected land use designations to the current vacant lots through a spatial join in ArcMap. These future land use designations were based on D.C.'s projected land use map which estimates land use in approximately 20 years (NCPC 2021). All identified sites that were not designated as parks (i.e.: with a future designation of residential, commercial, industrial, or mixed use) were considered priority conservation or restoration sites, respectively.

2.3 Results

2.3.1 Biodiversity Indicators

The highest species richness of all SGCN richness was in Section G of Anacostia Park ($q0 = 63$). The highest observed Shannon Diversity value for SGCN was in the U.S National Arboretum (USNA) ($q1 = 23.60$), followed by Section G of Anacostia ($q1 = 16.5$) and Fort Dupont ($q1 = 13.61$) (Table 2-4; Figure 2-2D). The highest Shannon Diversity and Inverse Simpson values for bird SGCN were found in the U.S Arboretum (USNA), where 32 different avian SGCN species were detected ($q1 = 16.52$; $q2 = 10.76$) (Table 2-4; Figure 2-2A). However, the highest richness of birds was found in section G of Anacostia Park ($q2 = 43$). Section C of Anacostia had the highest value of Shannon diversity for mammal SGCN ($q1=3.16$), where 5 different species were detected (Table 2-4; Figure 2-2B). When measuring the Inverse Simpson index for mammals, Rock Creek was the most diverse ($q2 = 2.78$). Reptile/Amphibian SGCN diversity was highest in Fort Dupont Park ($q1= 8.28$ and $q2 = 8$) where 11 different species were found (Table 4; Figure 2C). However, Section G of Anacostia Park has the greatest number of reptiles/Amphibians ($q0 = 18$).

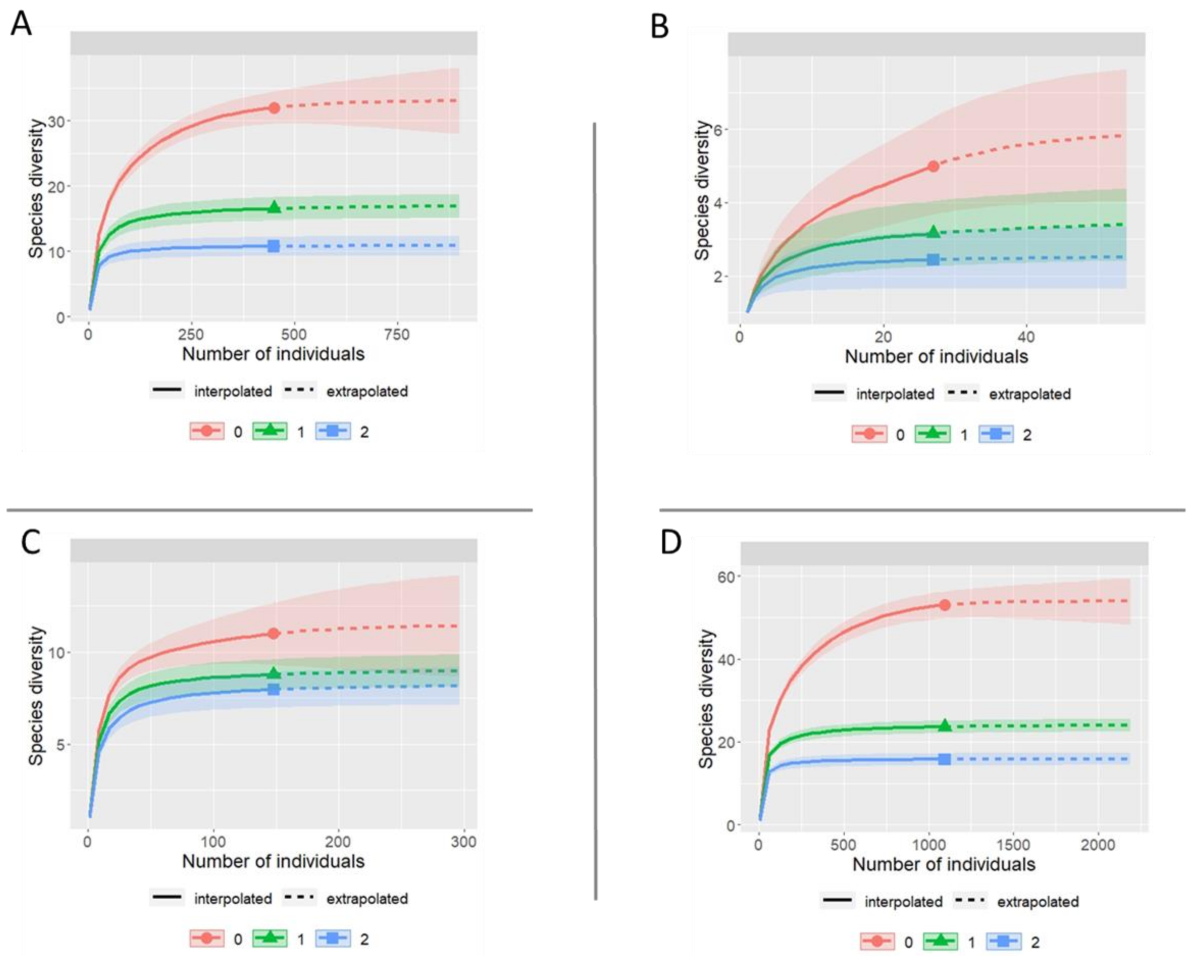

Figure 2-2: Rarefaction/extrapolation biodiversity curves for parks with the highest SGCN diversity for each taxon: Birds; US National Arboretum (A), Mammals; Anacostia Park- Section C (B), Reptiles/Amphibians; Fort Dupont (C), and for all SGCN; US National Arboretum (D); based on Hill's numbers with q0 (species richness – red line), q1 (Shannon index- green line), and q2 (Inverse Simpson index- blue line). The solid line is the rarefaction curve, and the dotted line is the extrapolation curve, which goes up to double the number of individuals.

Table 2-4: Number of individual observations (n) and Hill numbers of species richness (q0), Shannon diversity (q1), and Simpson diversity (q2) for SGCN within each taxonomic group and for all SGCN for the 27 surveyed parks in the District of Columbia. The bolded numbers represent those sites with the highest species richness and/or diversity for each taxon and for all Species of Greatest Conservation Need (SGCN).

	SGCN BIRDS				**SGCN MAMMALS**				**SGCN REPTILES & AMPHIBIANS**				**All SGCN**			
	(n)	**q0**	**q1**	**q2**	**(n)**	**q0**	**q1**	**q2**	**(n)**	**q0**	**q1**	**q2**	**(n)**	**q0**	**q1**	**q2**
Anacostia Park, Section C	17	8	5.8	4.3	27	**5**	3.2	2.4	123.0	9.0	5.9	4.9	167	22	11.3	8.0
Anacostia Park, Section D					1	1	1.0	1.0	14.0	4.0	2.4	1.8	15	5	2.9	2.1
Anacostia Park, Section F	12	5	4.4	4.0	35	1	1.0	1.0	22.0	5.0	4.1	3.4	69	11	5.6	3.4
Anacostia Park, Section G	1542	**43**	8.9	4.5	80	2	1.1	1.0	778.0	**18.0**	7.0	3.9	2400	**63**	16.5	8.3
C&O Canal									16.0	7.0	5.3	4.3	16	7	5.3	4.3
Archbold Parkway	13	7	5.4	4.3									13	7	5.4	4.3
Barnard Hill	11	2	1.4	1.2	2	1	1.0	1.0					13	3	2.0	1.6
Battery Ricketts	1	1	1.0	1.0									1	1	1.0	1.0
Dumbarton Oaks Park	144	20	11.4	8.0	36	1	1.0	1.0					180	21	11.5	8.3
East Potomac Park	1	1	1.0	1.0									1	1	1.0	1.0
Fort Chaplin Park	17	2	1.3	1.1	2	1	1.0	1.0					19	3	1.7	1.4
Fort Dupont Park	118	17	9.5	7.2	121	1	1.0	1.0	148.0	11.0	**8.8**	**8.0**	387	29	13.6	7.7
Fort Mahan Park	69	6	3.0	2.1	8	1	1.0	1.0					77	7	3.7	2.5
Fort Stanton Park	8	5	4.8	4.6									8	5	4.8	4.6
Fort Totten					103	3	1.5	1.2	6.0	2.0	1.6	1.4	109	5	1.8	1.4

continued

Table 2-4 continued

Oxon Run Parkway	17	6	4.0	3.1	7	1	1.0	1.0	11.0	3.0	2.5	2.3	35	10	7.4	6.3
Glover Parkway & Children's Playground	37	7	4.7	3.8	12	1	1.0	1.0	4.0	3.0	2.8	2.7	53	11	6.9	5.5
Heritage Island	186	22	11.9	7.6					12	5.0	3.9	3.1	198	27	14.0	8.5
Kingman Island	285	25	10.4	5.1					76	10	6.4	4.9	361	35	15.6	7.6
Park	10	2	1.4	1.2									10	2	1.4	1.2
Potomac Palisades Parkway	9	5	4.3	3.9	35	1	1.0	1.0	392	16	4.6	2.5	436	22	5.9	3.1
Rock Creek Park & Piney Branch Parkway	83	20	12.1	7.8	13	3	2.9	2.8	374	14	8.3	6.9	470	37	15.5	10.5
Shepherd Parkway	2	2	2.0	2.0									2	2	2.0	2.0
Theodore Roosevelt Island	34	10	4.3	2.5					23.0	4.0	3.3	3.0	57	14	7.7	5.1
United States National Arboretum	450	32	**16.5**	**10.8**	91	4	1.7	1.3	554	17	7.9	6.0	1095	53	**23.6**	**15.8**
Whitehaven Parkway	16	5	2.8	2.0	2	1	1.0	1.0					18	2	3.6	2.5
West Potomac Park	2	2	2.0	2.0									2	6	2.0	2.0

2.3.2 Sample Completeness and Differences in Diversity Across the Subset of Parks Where All SGCN Taxa Were Present.

When comparing diversity across the subset of nine parks where SGCN for each taxonomic group were present, the same general pattern of rarefaction/extrapolation occurred for the Shannon index (q1) (strongly representing the number of recurrent species), however, a change in the order of richness did occur with USNA and section G of Anacostia (Figure 2-3). All parks were characterized with high levels of dataset completeness suggesting that they were an adequate representative of SGCN diversity, and thus SGCN habitat use (Table 2-5). Both Anacostia-Section G and USNA were characterized by the highest levels of dataset completeness; 99.7% and 99.6% respectively, which resulted in a small increase in species richness (q0) when the sampling effort increased above 750 incidences (comparing with the measured species richness based on reference sample of 2,264 individual incidences). All but one of the nine SGCN parks had greater than 92% sample completeness with Oxon Run parkway at 85% sample completeness.

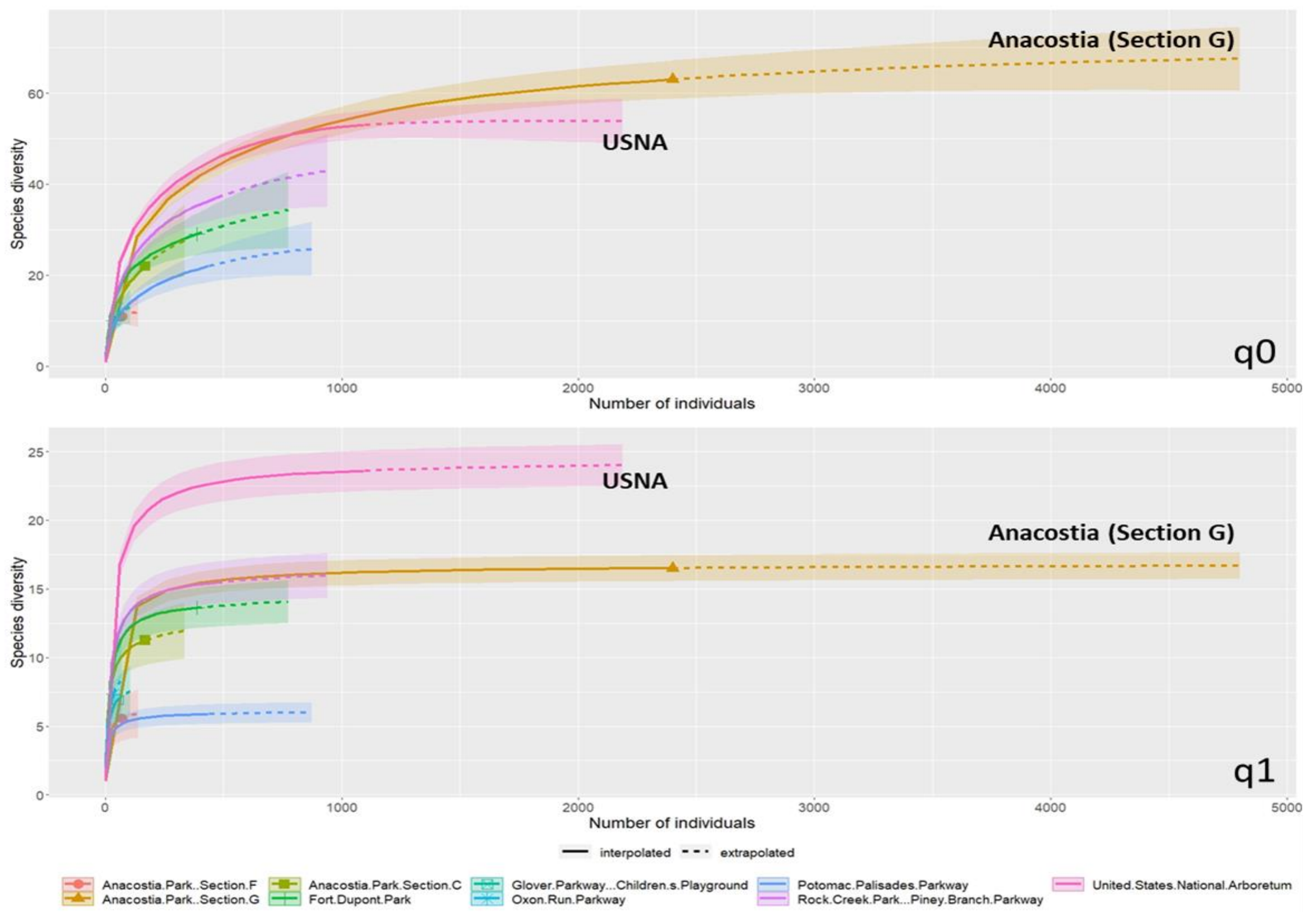

Figure 2-3: Integrated curves of rarefaction/extrapolation for q0 (species richness) and q1 (Shannon index), subset of nine parks where all SGCN were present – presented as a function of the number of individuals (n). Extrapolation goes up to double the size of the reference sample; n = 4528. The shaded area represents 95% confidence intervals obtained using the bootstrap method based on 100 repetitions

Table 2-5**:** Number of Species Observed (S-obs) and Sample Completeness (SC) of surveyed parks for the subset of nine parks where SGCN for every taxonomic group were found. Completeness inferences were made at a significance level of $\alpha = 0.05$ (95% confidence interval).

Site	**SGCN Species Group**	
	S-obs	SC
Anacostia Park (Section C)	22	95.20%
Anacostia Park (Section F)	11	97.20%
Anacostia Park (Section G)	63	99.70%
Fort Dupont Park	29	98.20%
Glover Parkway	11	92.70%
Oxon Run Parkway	10	85.90%
Potomac Palisades Parkway	22	98.60%
Rock Creek Park	37	98.10%
United States National Arboretum	53	99.60%

Results of the Kruskal–Wallis one-way analysis of variance indicated that there were significant differences in Shannon diversity (q1) (H=292.78 df=8, p = 0) across the subset of nine parks (Table 2-6). This is further illustrated in the box plots in Figure 2-4. Pairwise comparisons using Dunn's test indicated that the difference in Shannon diversity (q1) in Section C of Anacostia Park and Fort Dupont were not statistically significant (p= 0.08) (Table 2-6). Additionally, there was no statistical difference in Shannon diversity between Potomac Palisades Parkway and Section F of Anacostia Park or Glover Parkway (p = 0.93 and 0.05 respectively). Nor was there statistical significance in diversity between Rock Creek Park and Section G of Anacostia Park or Fort Dupont (p= 0.40 and 0.09 respectively). All other differences in diversity across the parks were statistically significant (p<0.05).

Table 2-6: Kruskal Wallis test statistic (H) and Dunn's pairwise comparison (using the Benjamini-Hochberg adjustment method) of Shannon Diversity (q1) between the nine parks where SGCN for all three taxa were present. Dunn's test compares the Z-test which is comparison of the means when the variances are known. Significance is taken at $p < 0.05$.

Kruskal-Wallis		
chi-squared (H) = 292.7849	df = 8	p-value = 0

Dunn Test (Benjamini-Hochberg)			
Comparison	**Z**	**P (unadjusted)**	**P (BH)**
Anacostia_C - Anacostia_F	5.15	0.00	0.00
Anacostia_C - Anacostia_G	-4.38	0.00	0.00
Anacostia_F - Anacostia_G	-9.53	0.00	0.00
Anacostia_C - Fort Dupont	**-1.79**	**0.07**	**0.08**
Anacostia_F - Fort Dupont	-6.94	0.00	0.00
Anacostia, G - Fort Dupont	2.59	0.01	0.01
Anacostia, C - Glover Pkwy	3.11	0.00	0.00
Anacostia, F - Glover Pkwy	-2.04	0.04	0.05
Anacostia, G - Glover Pkwy	7.50	0.00	0.00
Fort Dupont - Glover Pkwy	4.90	0.00	0.00
Anacostia, C - Oxon Run	2.56	0.01	0.01
Anacostia, F - Oxon Run	-2.59	0.01	0.01
Anacostia, G - Oxon Run	6.94	0.00	0.00
Fort Dupont - Oxon Run	4.35	0.00	0.00
Glover Pkwy- Oxon Run	-0.55	0.58	0.60
Anacostia_C - Pot. Palisades	5.07	0.00	0.00
Anacostia, F - Pot. Palisades	**-0.08**	**0.93**	**0.93**
Anacostia, G - Pot. Palisades	9.45	0.00	0.00
Fort Dupont - Pot. Palisades	6.85	0.00	0.00
Glover_PKWY - Pot. Palisades	**1.95**	**0.05**	**0.06**
Oxon Run - Pot. Palisades	2.51	0.01	0.02
Anacostia, C - Rock Creek Park	-3.51	0.00	0.00
Anacostia, F - Rock Creek Park	-8.66	0.00	0.00
Anacostia, G - Rock Creek Park	**0.88**	**0.38**	**0.40**
Fort Dupont - Rock Creek Park	**-1.72**	**0.09**	**0.09**
Glover Pkwy - Rock Creek Park	-6.62	0.00	0.00
Oxon Run - Rock Creek Park	-6.07	0.00	0.00
Pot. Palisades - Rock Creek Park	-8.57	0.00	0.00
Anacostia, C - USNA	-6.58	0.00	0.00
Anacostia, F - USNA	-11.73	0.00	0.00
Anacostia, G - USNA	-2.20	0.03	0.03
Fort Dupont - USNA	-4.79	0.00	0.00
Fort Dupont - USNA	-9.69	0.00	0.00
Oxon Run - USNA	-9.14	0.00	0.00
Pot. Palisades - USNA	-11.64	0.00	0.00
Rock Creek Park - USNA	-3.07	0.00	0.00

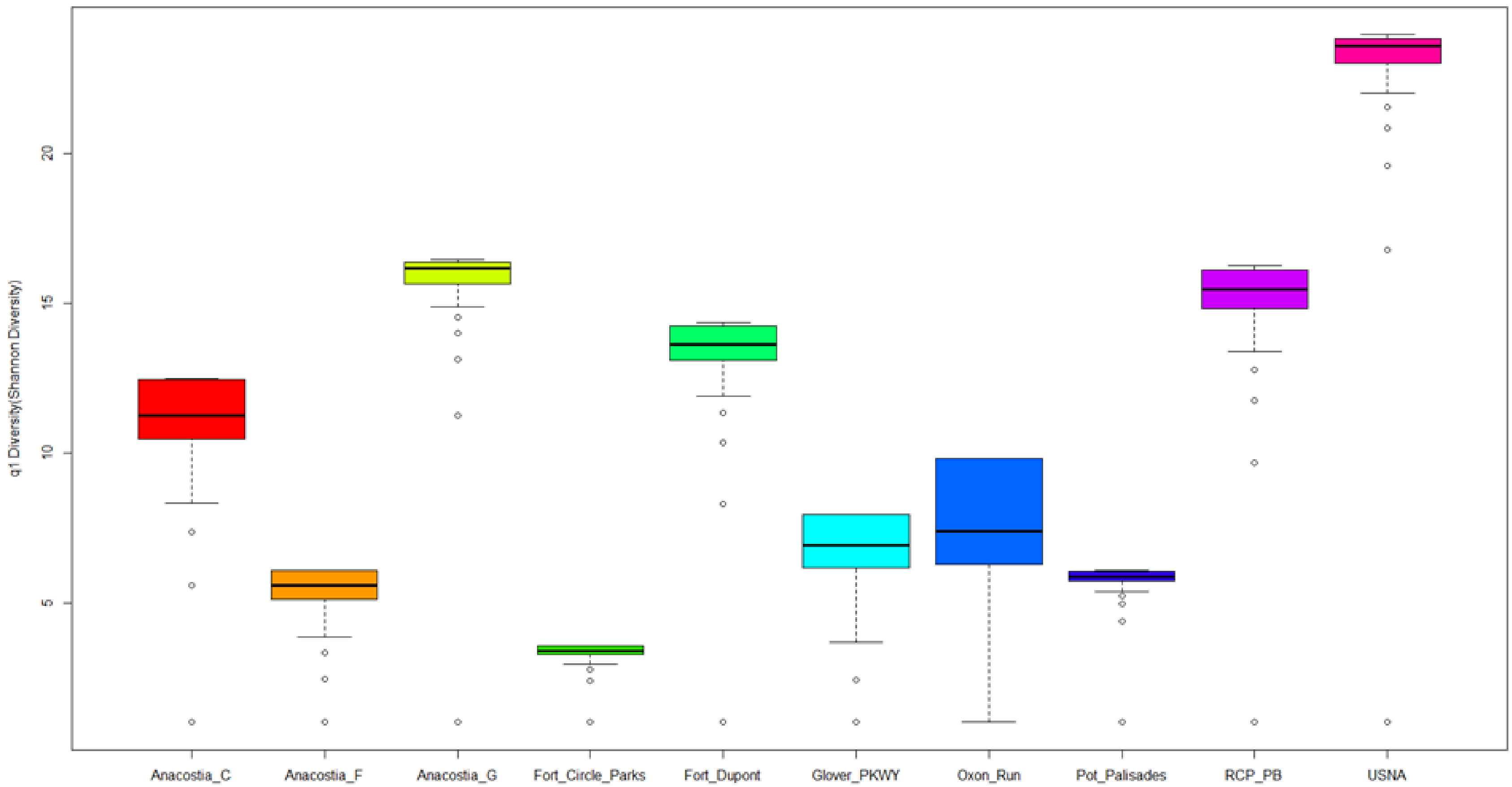

Figure 2-4: Box and whisker plots comparing Shannon diversity (q1) q1 across the subset of nine D.C parks where SGCN for each taxon were present. The box indicates the shape of the data distribution, the bold center line represents the central value (median), and the whiskers represent variability in the data.

2.3.3 Habitat Characteristics

Results of the PCA on the habitat metrics calculated for the nine parks where SGCN in all three taxonomic groups were present showed that the first three principal components have eigenvalues greater than 1 (Table 2-7). These three components explained 82 % of the variation in habitat variables. The first principal component explained 52% of the total variance and had large positive associations with forest cover and low vegetation (Figure 2-5). The second component explained 18% of total variance and had a positive association with forest openings. The third component explained 12% of total variance with a large negative association with small and medium core forests and canopy height. The proportion of large core forests and total forest edge made the highest contribution to PC1 with values of 0.37 and -0.37 respectively, followed by the percent of low vegetation, and the proportions of medium and small core forests. Openings in core forest and total forest area made the highest contributions to PC2 with values of 0.63 and 0.47, respectively (Figure 2-5). The number of wetlands made the highest contribution to PC3 with a value of 0.63.

Table 2-7. Results of the Principal Component Analysis (PCA) used to determine which habitat characteristics were retained for the correlation test and subsequent model.

Importance of components:									
	PC1	**PC2**	**PC3**	**PC4**	**PC5**	**PC6**	**PC7**	**PC8**	**PC9**
Standard deviation	2.40	1.4	1.15	1.0	0..60	0.59	0.36	0.30	0
Proportion of Variance	0.5	0.18	0.12	0.09	0.03	0.03	0.01	0	0
Cumulative Proportion	0.5	0.7	0.83	0.91	0.95	0.98	0.99	1	1

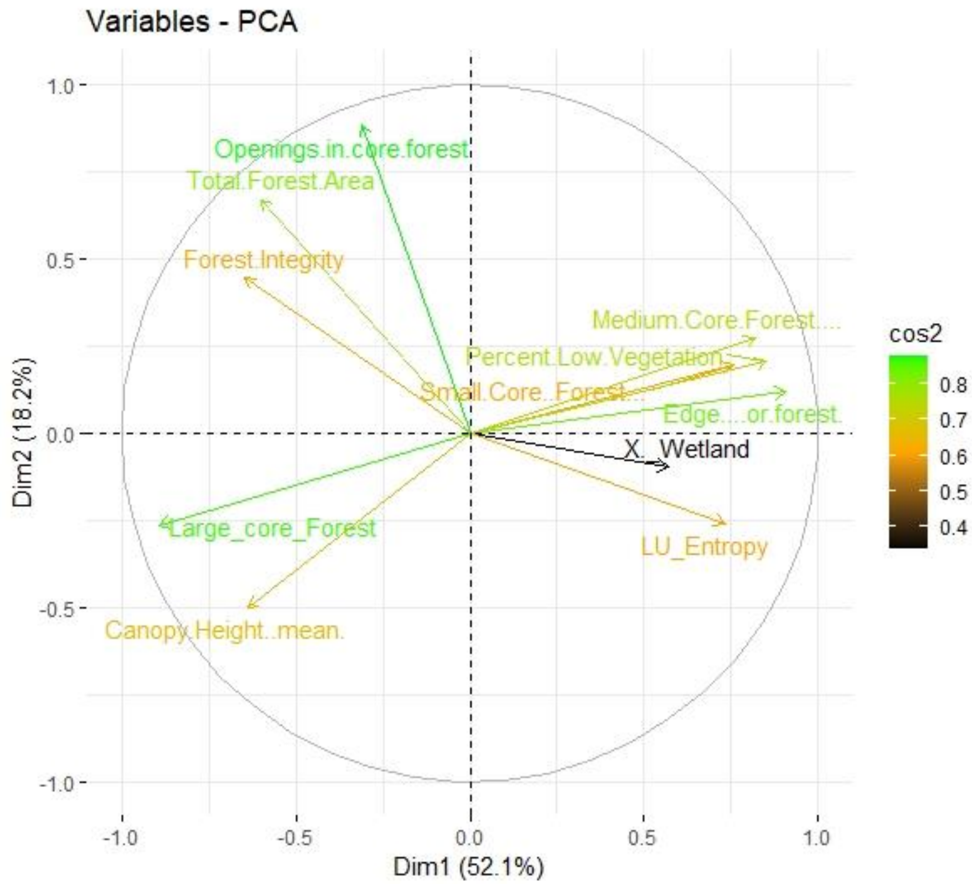

Figure 2-5: Variable correlation plot showing the relationships between the calculated habitat variables; mapped onto the first two principal components of PCA (cumulative variance=70%). Habitat variables are colored by their contributions to PC1 and PC2 where green has the largest contribution and black has the least. Strong predictors have longer arrows than weak predictors and the direction of the vector suggests correlation between variables. Smaller angles between two vectors represent larger correlation

Total forest area was significantly correlated with Shannon diversity (q1) for SGCN in all three taxonomic groups, and for all SGCN regardless of taxon, with p-values of 0.05 (Table 2-8). As total forest area increased, Shannon diversity (q1) significantly increased for SGCN bird, mammals, and reptiles/amphibians. Openings within core forests were also highly significant to SGCN diversity within and across taxa. Medium core forest showed a significant correlation to SGCN mammal diversity (r = 0.47, p-value =0.01) (Table 2-8).

Table 2-8: Pearson's correlation coefficients and significance ((*p < 0.05), (**p < 0.01) and (***p < 0.001)) between Shannon Diversity (q1) and the three land cover classes that were significant for each taxon and for all SGCN.

	BIRDS		**MAMMALS**		**HERPS**		**SGCN**	
	r	P Value	r	P Value	r	P Value	r	P Value
Total Forest Area	0.41	*0.05	0.56	*0.05	0.54	*0.05	0.40	*0.05
Medium Core Forest (%)	-0.24	0.2	0.47	*0.05	0.23	0.2	-0.21	0.2
Openings in core forest	0.57	**0.01	0.41	0.1	0.77	***0.001	0.68	***0.001

2.3.4 Conservation and Restoration Sites within Corridors

I identified 47 least cost paths connecting all 27 surveyed parks across D.C. (Figure 2-6). The distribution of land cover within my established linkage corridors was 31% forest cover (2.74 km²) and 18% impervious surfaces and structures (1.5 km²; Figure 2-7).

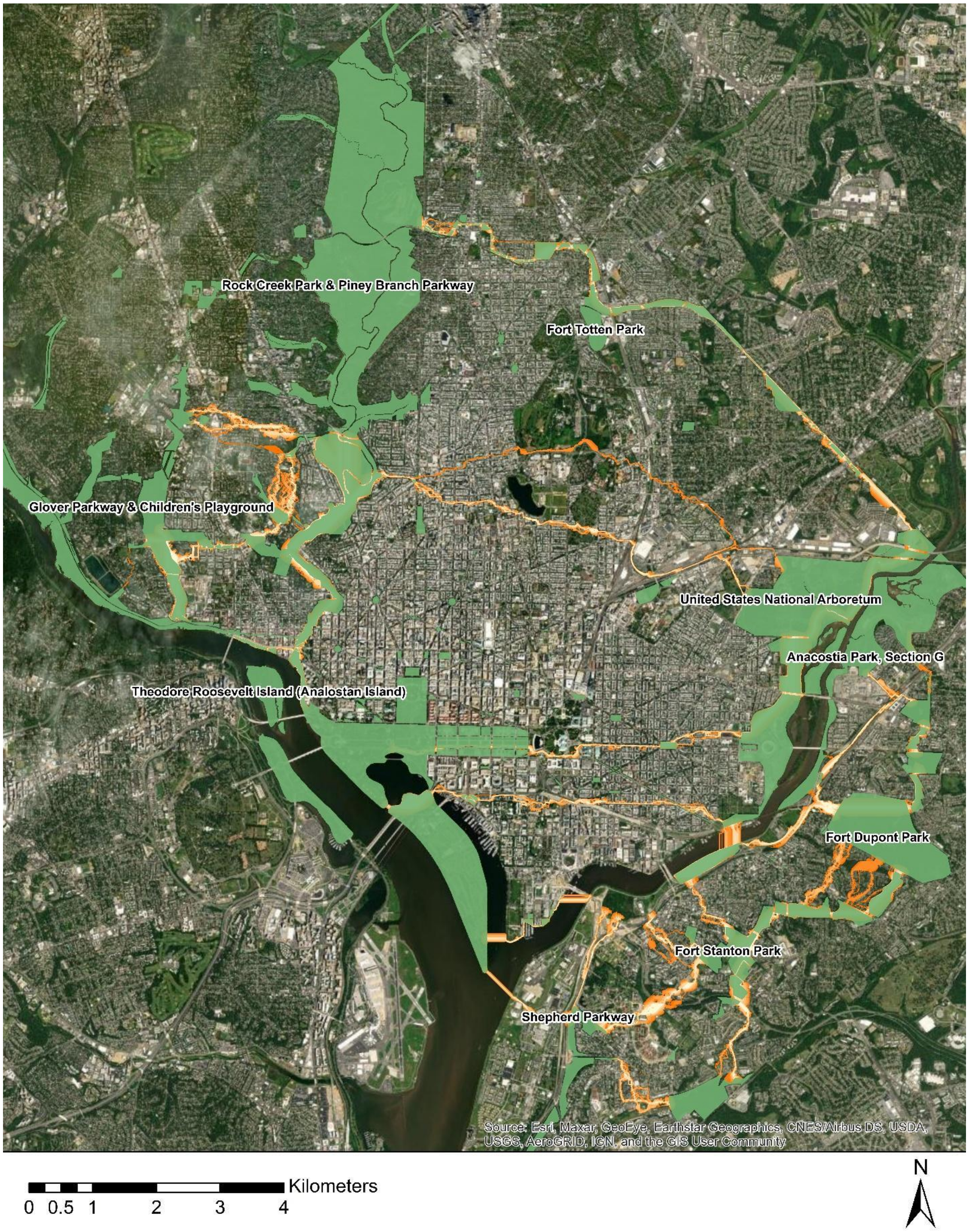

Figure 2-6: Least cost corridor linkage pathways (orange) connecting the 27 surveyed parks (green) in Washington D.C. The 47 least cost paths were modeled using the Linkage Pathways tool in the Linkage Mapper 3.0.0 toolbox in ArcGIS 10.8 to connect each core park to its four nearest neighbors using cost-weighted distance to identify the lowest cost corridors. After nearest neighbors were identified, additional links were re-added to connect the nearest pairs of otherwise isolated groups of core parks.

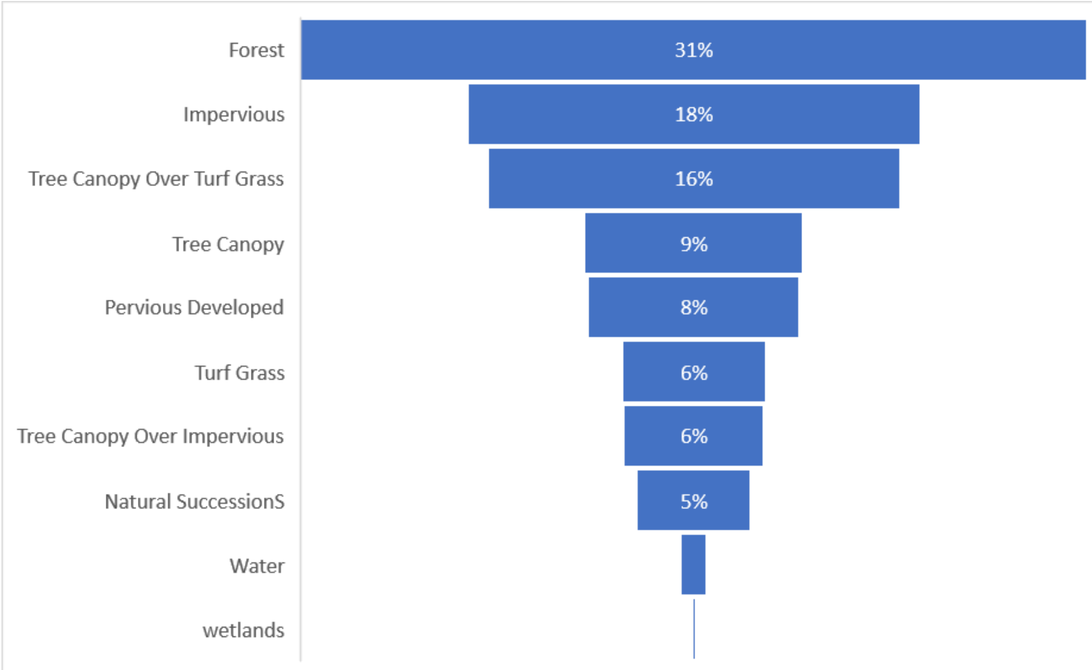

Figure 2-7: Distribution of land cover classes within the truncated linkage pathway corridors in D.C.

I identified several barriers equal to or less than 10m wide. Restoration of any of several barriers identified in Southeast D.C. could improve connectivity between four of the most biodiverse parks (Figure 2-8). The locations with highest improvement scores were 1) Ridge Road SE between Eli Pl SE and Texas Ave SE; 2) C Street SE between Burns Street and Burbank St; 3) Massachusetts Ave. SE just north of O Street, Pennsylvania Ave between Fort Davis Drive and 38th Street; 4) Branch Ave just South of 33rd Street SE; 5) Good Hope Road between Fort Davis Park and Altamonte Pl SE, and 6) the section of Suitland Parkway just Southeast of Gainesville Street (Figure 2-9, light pink/tan). These areas are important restoration opportunities for maintaining network connectivity and may serve as ideal locations to place wildlife crossing structures.

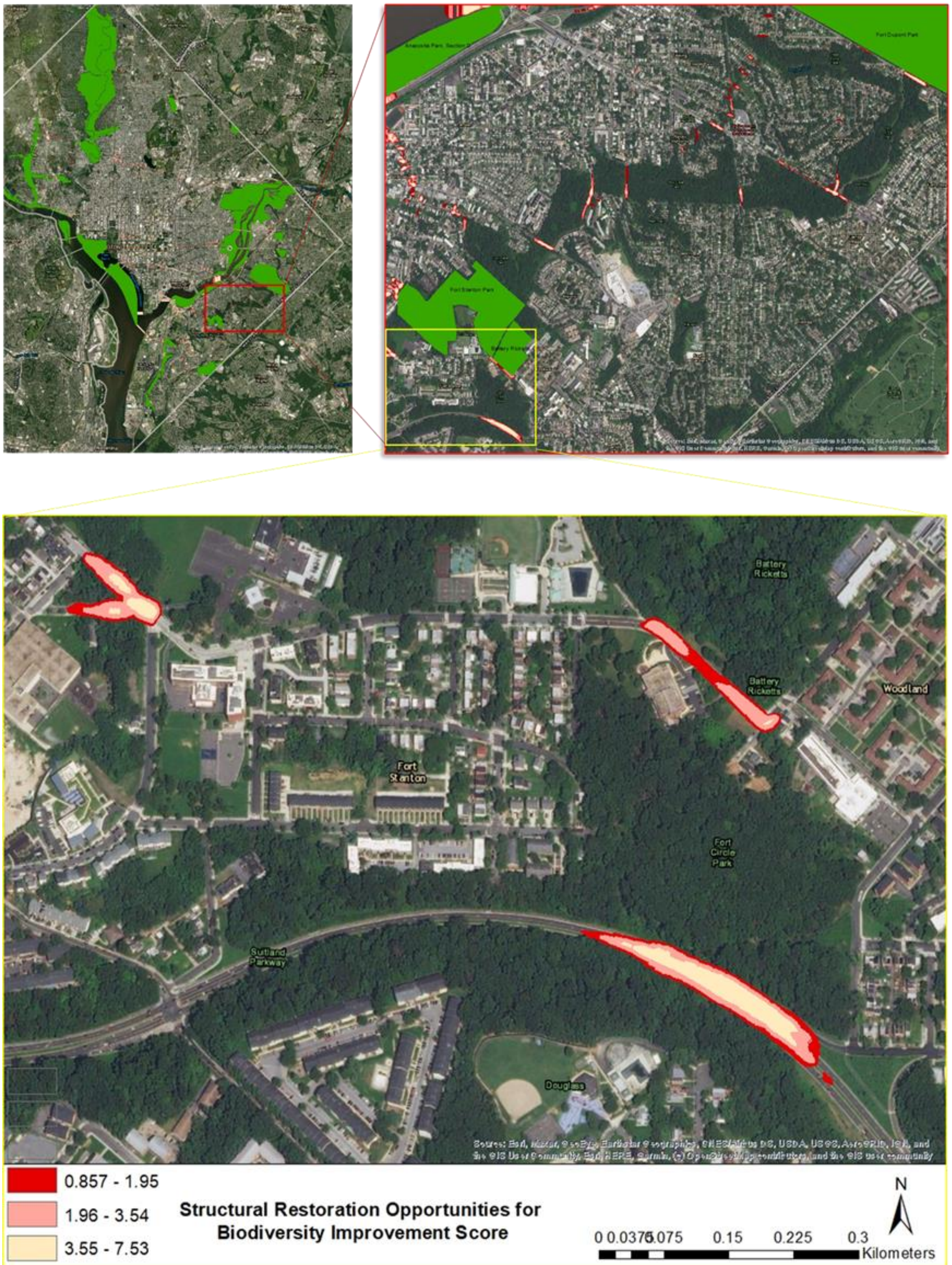

Figure 2-8: Inset maps showing the results of the barrier analysis with original patch pair at 10 m search diameter, to detect structural restoration opportunities equal to or less than 10 m across (e.g., roads) between the 4 most biodiverse parks in Washington D.C, USA. Areas with the highest biodiversity improvement indicate that these areas are important restoration opportunities for maintaining network connectivity.

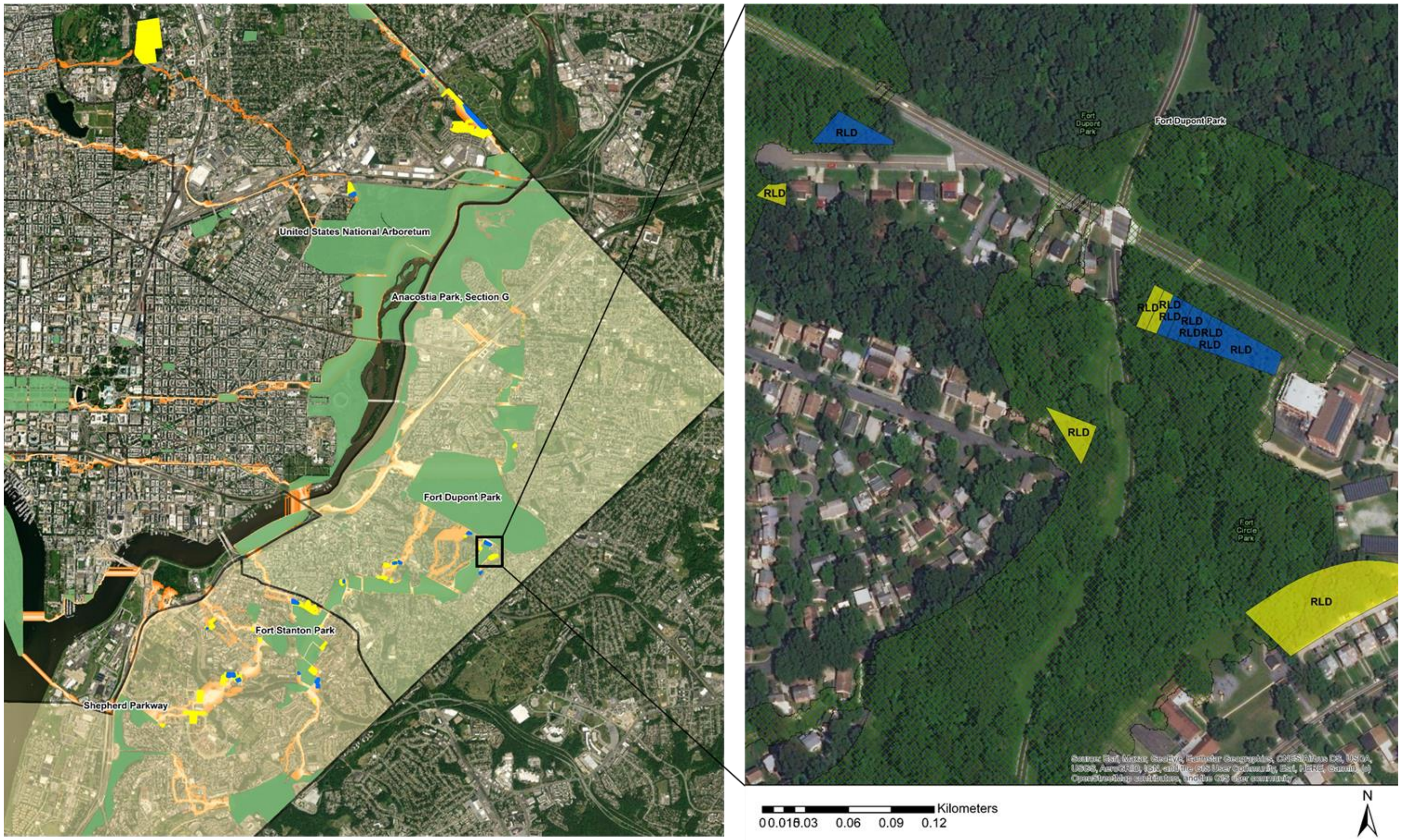

Figure 2-9: The priority conservation sites (blue) and candidate restoration sites (yellow) that have the potential to increase connectivity between the biodiverse core parks (green) in Washington D.C.'s Southeast quadrant. Approximately 80% (70/89) of the restoration sites and 90% (44/49) of the conservation sites are in the Far Northeast and Southeast and Far Southeast and Southwest planning zones (left). The figure to the right highlights a portion of those sites that are within a section of the Fort circle of parks corridor just south of Fort Dupont.

There was a total of 5,475 land parcels, with nine different primary land use designations, intersecting the truncated linkage corridors (Table 2-9); 75.75% of these parcels were zoned for residential use, 152 parcels (2.78%) were zoned for commercial use, with the remaining 21.55% zoned for industrial use, institutional use, mixed use, religious use, or were vacant lots.

Table 2-9: Distribution of current land uses intersecting the truncated linkage pathways corridor in Washing D.C. Of the 979 vacant sites, 671 sites are described as 'vacant-true' meaning that the lot is not improved with any structure. 155 vacant lots have zoning limits, 74 lots have permits, 64 are considered vacant-false abutting meaning the lot is assigned no real estate improvement value but has part of a structure whose value is assigned to another lot. The remaining 10 vacant lots are designated for commercial use.

Primary Land Use	**Number of Lots**	**Percent of Corridor**
Commercial	152	2.78%
Industrial	16	0.29%
Institutional	90	1.65%
Mixed	15	0.27%
Other	38	0.70%
Religious	35	0.64%
Residential	4138	75.75%
Vacant - True	671	18%
Vacant – Zoning Limits	155	
Vacant – with Permit	74	
Vacant – False Abutting	64	
Vacant - Commercial	10	
Vacant – Unimproved Parking	1	
Total	5475	100.00%

Lands designated as vacant were 18% (987 parcels) of the total identified land parcels that fell within or intersected the established corridors (Table 2-10; Figure 2-10). Of the 987 vacant lots, 15% (149 lots) had greater than 90% forest cover, making them candidate conservation sites. Forty-nine (33%) of those 149 candidate conservation sites were designated for either residential or mixed-use development: flagging them as priority conservation sites (Table 2-10). Of the remaining 614 vacant lots, 224 lots (22%) have between zero and 90% cover making them candidate restoration sites. Eighty-nine (40%) of those candidate restoration sites were designated for residential, commercial, or mixed-use development, thus identified for priority restoration efforts.

Table 2-10: Planned land use associated with candidate conservation and restoration sites that intersect the truncated linkage pathway corridor in Washington D.C. Conservation sites have ≥ 90% forest cover and restoration sites have 0% > 90% forest cover. The planned land use classes are derived from the 2021 Comprehensive Plan Future Land use dataset which depicts intended land use designations approximately 20 years in the future. Forest cover is derived from the 2017 Chesapeake Conservancy's High- resolution 1m landcover dataset and is defined as "all contiguous patches of trees ≥1 acre in extent with a patch width ≥240-ft somewhere in the patch. The 240-ft girth references potential altered microclimate conditions extending inwards up to 120-ft from the patch edge. The forest understory is assumed to be undisturbed/unmanaged. Forests that are also wetlands are included in this class".

	Planned Land Use	**Number of Sites**	**Percent of Sites**
Conservation	Mixed Use	2	0.20%
	Parks, Recreation, Open Space	100	10.13%
	Residential	47	4.76%
Restoration	Mixed Use	6	0.61%
	Other	7	0.71%
	Parks, Recreation, Open Space	135	13.68%
	Residential	76	7.70%
Total		987	100.00%

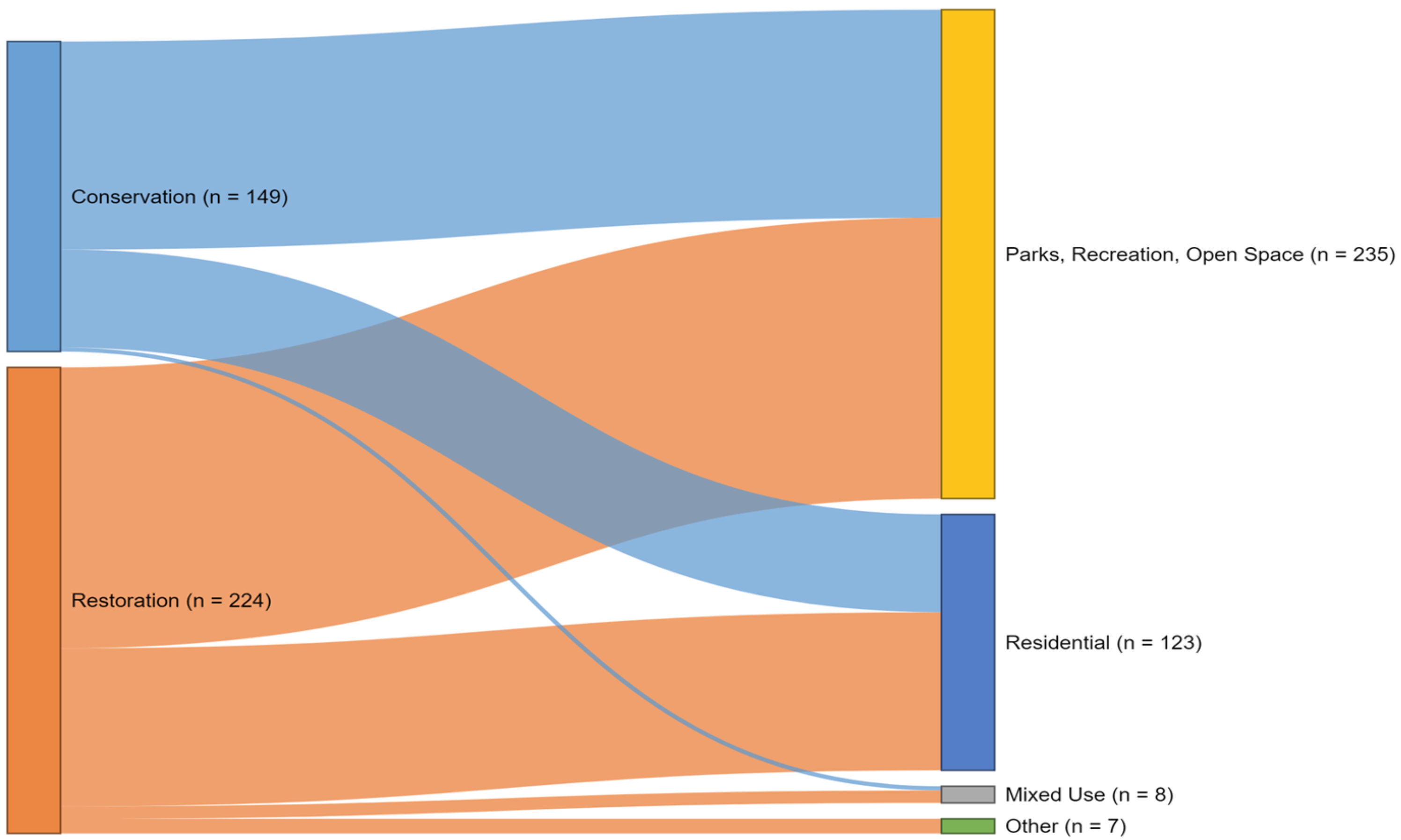

Figure 2-10: Sanky diagram representing the number of candidate conservation and restoration sites that fell within or intersected the linkage pathway corridors connecting the surveyed parks, that are designated as Parks, Recreation or Open Space (PROS), Residential, Mixed Use, or Other (Industrial, Commercial, Federal) in the District of Columbia's 2021 Comprehensive Plan.

2.4 Discussion

A primary goal of urban wildlife conservation is to conserve, connect, or restore habitat for species that would otherwise be displaced by past and on-going anthropogenic land use changes. In recent years, urban planners have been encouraged to, reactively or proactively, incorporate the need for biodiversity conservation within local comprehensive plans. This study provides a framework for prioritizing biodiversity in urban land use planning by combining traditional species surveys with new high-resolution land use and land cover data to identify high impact conservation or restoration areas within a broader green corridor network. Like many cities, D.C. has a state wildlife action plan (SWAP), with a focus on SGCN. To support planning goals, local agencies around the country have invested in wildlife data collection to assess biodiversity metrics and overall habitat quality for their SGCN. This study identified vacant lots and small greenspaces within city-wide habitat corridors that can directly assist in the continued conservation of SGCN.

Here, I utilized long-term multi-species survey data, covering three taxonomic groups and was able to derive more accurate biodiversity metrics for SGCN, including the number of species present, the richness and diversity of species, and the relative abundance of species. Section G of Anacostia park was identified as the most species-rich; with 2400 individual SGCN identified over the past 10 years. To maintain the integrity of this biodiverse park, city planners can ensure compliance with existing environmental regulations, such as the Anacostia River cleanup, and enforce protective measures. The use of Hill diversity also allowed me to capture a more nuanced and comprehensive understanding of the spatial variation in SGCN diversity. For example, I was able to differentiate between parks that have high species richness (i.e., a high number of species) but are dominated by a few highly abundant species such as Section F of Anacostia Park, versus parks that have high species richness but are more evenly distributed among species such as the U.S National Arboretum. This information is critical for making informed conservation and planning decisions, as it allows D.C. wildlife managers to accurately identify areas that are more vulnerable to species loss and prioritize conservation efforts accordingly. Many cities, including Chicago, New York City, and Portland, that also do long-term trend-type surveys of species presence can benefit from the framework provided in this study. It is important to note, however, that while Hill biodiversity metrics can account for unequal sampling effort to an extent, they are not a complete solution for this problem. Moving forward, I recommend more spatially and temporally consistent data collection to refine current biodiversity metrics and allow for better tracking of changes over time.

Washington D.C is well known for its high tree canopy cover and intentionality in tree planting in public areas (DOEE 2021; Leets et al. 2022). D.C. also actively encourages numerous tree planting initiatives on private lands (Leets et al. 2022). Furthermore, D.C. is generally well connected by greenspace, thanks to its extensive network of parks, recreational areas, and green corridors. The city is known for its numerous public parks, including the National Mall, Rock Creek Park, and many neighborhood parks and community green spaces. However, this study identified several areas in D.C. where connections between greenspaces need to be strengthened to enhance ecological connectivity and improve the overall greenspace network. These areas include connections between Rock Creek Park and Fort Totten in the Northeast region, and connections between Fort Dupont and Fort Stanton and Shepherds Parkway, Southeast of the Anacostia River. Notably, the corridors identified in this study closely align with the network of Fort Circle Parks. The Fort Circle Parks are a collection of Civil War-era fortifications and associated parklands that encircle the city, providing recreational opportunities and historical significance to residents and visitors. Developing urban greenways and trails that connect the individual Fort Circle Parks, particularly those high in biodiversity, such as Fort Dupont, Fort Mahan, and Fort Totten, will be a fundamental step in creating a continuous greenway that will improve species movement between the larger parks. In Addition to enhancing ecological connectivity, ensuring that underserved neighborhoods have equitable access to greenspaces is crucial. Establishing small pocket parks in areas with limited greenery along the mapped corridors, such as in the neighborhoods of Howard University, LeDroit Park, and Cordazo/Shaw can improve urban biodiversity and offer residents closer connections to nature. Lastly, establishing partnerships between government agencies, non-profit organizations, and community groups can help leverage resources and expertise to advance greenway connectivity initiatives. For example, Casey Trees (https://caseytrees.org); a nonprofit organization based in Washington, D.C., with a mission to restore, enhance, and protect the tree canopy of the nation's capital is dedicated to the preservation and improvement of the city's tree canopy. Working in collaboration with the Urban Forestry Division (UFD) and the Department of Energy and Environment (DOEE), one of their key initiatives is to increase the presence of healthy native trees on private residential land. The vacant lots found within the proposed wildlife corridors present potential for conservation easements or land acquisitions to protect these critical natural areas and encouraging private landowners to enter into these conservation easements will help ensure the conservation of biodiversity in the city.

Within the proposed wildlife corridors, I identified 138 vacant lots considered the highest risk for future gray infrastructure based on future land use designations from D.C.'s Comprehensive Plan Amendment Act of 2021. Of the 138-priority conservation and restoration sites identified, I found that a total of 22 priority conservation and restoration sites already lie within national parks. This suggests that proactive conservation planning even within protected areas is important for ensuring ongoing habitat requirements for existing SGCN.

By considering how the restoration of unconventional habitats such as vacant lots might promote local biodiversity, urban wildlife managers can accomplish conservation gains in areas that might otherwise be overlooked due to their high development value (Rupprecht et al. 2015; Soanes et al. 2019). Furthermore, cities can turn high-valued, underutilized land into ecological assets that advance biodiversity needs while contributing to the overall quality of life in their communities. Here, I identified over 900 vacant lots within my proposed corridors, with over 130 lots having existing conservation or restoration potential for SCGN. Given the rising interest in reclaiming these lots and putting them to productive use (McClintock 2010; Metcalf 2011; Neilson & Rickards 2017; Pettygrove & Ghose 2018), proactive land-use policies that encourage strategic targeting of vacant lots would assist planners in better identifying and managing potential development risks that may counter biodiversity goals. While it is not feasible to ecologically restore every vacant lot, targeting vacant lots with biodiversity corridor potential may have the greatest impact. Furthermore, the identification of these strategic areas for habitat connectivity within communities' comprehensive plans can trigger investment in critically important habitat conservation and restoration initiatives. Greater than 78% of the vacant lots I flagged as high-risk for future gray infrastructure were located south of the Anacostia River (Ward 7 and 8). Because of decades-long institutional and structural bias, the neighborhoods and communities in this area have received fewer resources for recovery from catastrophic climate events (Ranganathan and Bratman 2021; King et al. 2022). Conserving and/or restoring these lots within these proposed corridor would provide the greatest benefit per unit cost and could increase resilience to climate change at the neighborhood level, particularly in these vulnerable riverfront communities where the impacts from catastrophic events, like floods, will be exacerbated. By making these co-benefits explicit, local policymakers can better communicate the broader societal benefits of conservation efforts and engage a wider range of stakeholders in conservation initiatives.

For example, city planners could work with companies planning residential, commercial or industrial projects to include trade-offs that protect – or even improve upon – currently vacant lots in the wildlife corridor areas. For instance, a residential builder might be granted a variance allowing higher-density building in lots that are not crucial for wildlife in return for creating pocket parks along the critical corridors connecting the fort circle parks. Or the city could give special consideration to an industrial project that also includes – as a tradeoff -- the company creating greenspace in the South East, which would result in both wildlife and social benefits in the Anacostia area.

Additionally, since approximately one-third of the land in DC is owned by the Federal government, any analyses under the National Environmental Policy Act should include mitigations. Planners could urge consideration of trade-offs to include preservation of critical greenspace, wildlife corridors, and wildlife crossings in sensitive areas to mitigate the effects of any planned land management actions.

Ultimately, city planners could encourage developers, by offering consideration or special variances, to consider environmental mitigations that dedicate currently vacant lots in wildlife corridors to green space. Areas to focus on would be: Ridge Road, SE; C St. SE; Mass. Ave. SE; Branch Ave.; Good Hope Road and the Suitland Parkway.

My study demonstrates that the use of high-resolution data provides an opportunity for improved city-wide habitat connectivity mapping and, when coupled with existing species surveys, can inform local habitat restoration efforts or new urban comprehensive plans. The movement barrier analysis also allowed me to distinguish key areas where habitat restoration may lead to the greatest improvement in habitat connectivity. It is essential to emphasize that although the high-resolution corridors were initially mapped using habitat characteristics of shared interest to all three SGCN taxa, further mapping specific to individual species might be necessary to establish causation rather than relying solely on high correlations. Nevertheless, this level of analysis allowed me to identify unconventional areas that are particularly important for overall biodiversity conservation by examining structural habitat characteristics not only within current protected areas but across a range of land-use/property ownership classes, which might otherwise be inaccessible to species survey efforts.

Given the often-small property sizes throughout urban areas like D.C., high spatial resolution analyses, such as this, can play a vital role in D.C., and other urban areas, where small property sizes and high population densities present unique challenges for conservation and sustainable development. By leveraging fine-scale data, urban planners and conservationists can make informed decisions, prioritize conservation efforts, and enhance the integration of greenspaces within the urban landscape, ultimately contributing to the well-being of both residents and wildlife. For example, like many past studies, I found that large intact forests patches were highly associated with SGCN from each taxon in my study (Jim and Lie 2001; Colding et al. 2003; Alvey 2006; Beninde, Veith, and Hochkirch 2015), and small property areas along the corridors (>=20m²) highlighted the need to consider strategic local restoration options. These small urban habitats can be essential to the persistence of local populations and can also enhance regional diversity (Soanes et al. 2019). Moreover, recognizing the high conservation and restoration value of these areas for regional biodiversity goals, can drive ongoing scientific study to ensure appropriate restoration design and emphasize the need for broader integration of biodiversity conservation with other social, planning, and policy goals (Spotswood et al. 2021).

Because biodiversity underpins ecological functioning (Constanza et al. 1997; Balvanera et al. 2006; Mace et al. 2012; Tilman et al. 2014; Oliver et al. 2015; Maes et al. 2016), policymakers interested in investing in nature-based solutions to promote urban resilience can refer to this work to design strategic urban conservation and restoration initiatives that maximize biodiversity conservation, while also accounting for economic opportunity, and the needs of vulnerable communities.

Chapter 3: Multiple Ecosystem Service Valuations and Assessments Strengthen Biodiversity Sensitive Urban Design and Green Corridors.

Abstract

Multiple ecosystem service assessments can be used to identify areas or actions that could benefit urban biodiversity while simultaneously supporting the economic and social goals of a city's sustainable development plan. Optimally targeting wildlife conservation and restoration areas that have the greatest benefits per unit cost assists in allocating scarce dollars more efficiently. Here, I utilized the suite of open-source models within InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) software to quantify the biophysical values of seven ecosystem services that are of major importance in most urban areas – climate change mitigation, heat mitigation, storm-water retention, extreme flood risk mitigation, and retention of three common pollutants. I then aggregated the biophysical output value of each service to 33 viable biodiversity corridors, and their associated planning area, in D.C., to identify potential conservation areas with high ecological and socioeconomic value. Results identified 11 high-impact corridors with above average ecosystem service scores. If preserved or restored, these corridors would maximize biodiversity and ecosystem service supply, thus delivering the greatest return on investment. The results also revealed variations in the type and magnitude of each service within and across each corridor due to the differences in land uses and their contribution to each service. By quantifying and mapping ecosystem services at a very high spatial resolution ($1m^2$), patterns were revealed in the distribution of

ecosystem service supply that might not be captured at larger scales, allowing for highly targeted conservation and restoration initiatives. Local policymakers interested in incentivizing habitat conservation to meet state or national sustainable development goals, can use this information for strategic urban conservation and restoration initiatives that account for both biodiversity and socioeconomic opportunity.

3.1 Introduction

The rising demand for renewable and non-renewable resources will lead to human-induced changes, putting additional stress on ecosystems. These pressures will, in turn, affect the functioning of ecosystems and the vital services they provide to human populations. (Costanza et al 1997; Daily 1997; Guo et al., 2010; Burkhard et al. 2012). Urban green spaces (UGS) are important to maintain ecological functioning and studies have indicated that even the smallest of green spaces can provide both ecological and social benefits in the form of high impact ecosystem services (ES) such as air and water purification, carbon storage, runoff retention, cooling, and increased property value (e.g., Dwyer et al. 1991; Bolund and Hunhammer 1999; Tzoulas et al. 2007; Bowler et al. 2010; Gómez-Baggethun et al. 2013). Additionally, these green spaces support numerous native species (Aronson et al. 2014; Ives et al. 2016; Lepczyk et al. 2017), due to their functioning as potential dispersal corridors and providing potential linkages between larger habitat patches (Bennett 1999; Beier and Noss 1998; Rudd et al. 2002). Nevertheless, maintaining and increasing available habitat for biodiversity conservation in urban areas is challenging because conservation efforts are often influenced by the availability of scarce financial resources and the difficult allocation of those resources among competing social, cultural, and environmental needs (Aronson et al 2017). Accounting for a combination of biophysical, socioeconomic, and cultural factors that underlie policy decisions in urban conservation and restoration plans, can result in more defensible and strategic funding requests (Nelson et al. 2009; Turo & Gardener 2020).

The potential of UGS to store and annually sequester carbon is significant (Nowak and Crane 2002), prompting increased UGS conservation and restoration efforts and local tree planting initiatives. Consequently, quantifying carbon storage in UGS can lead to a better understanding of the relationship between urban trees in global carbon accounting, improved urban planning and management, and improved human and environmental health (Meong et al. 2006; Nowak and Dwyer 2007). For example, urban forests serve as a carbon sink (Nowak and Crane 2002; McKinley et al. 2011) by providing shade to residential areas (McPhearson et al. 1997). They provide ecological stability by promoting biodiversity, preserving soil structure, and providing habitat for local wildlife (Dwyer et al. 1992). Urban forests also play a vital role in urban hydrology by reducing the rate and volume of stormwater runoff (Kuehler, Hathaway, and Tirpak 2017). As such, increasing urban forest cover may not only increase biodiversity but also complement many other valuable ecosystem functions (Miller, Hauer, and Werner 2015).

It has been noted that, by the end of this century, large areas of the Earth will experience unprecedented climatic changes (Williams, Jackson & Kutzbach 2007). Global mean temperatures are projected to increase 1.5–2° C by the end of the 21^{st}

Century (IPCC 2019). Studying urban ecosystems can assist conservationists in anticipating and mitigating climate change effects because urban ecosystems are already experiencing higher temperatures due to the urban heat island effect (Dearborn and Kark 2009). The urban heat island effect is caused by large amounts of heat being generated by urban structures absorbing and re-radiating solar radiation. Planting of vegetation is the most widely applied mitigation measure for such heat (Rizwan, Dennis & Chunho 2008), and biophysical effects of urban vegetation on local climate can be significant. Shade effects and evaporative cooling through transpiration can reduce local temperatures – and subsequently reduce energy consumption and concomitant greenhouse gas (GHG) emissions, providing further climate change mitigation (Akbari 2002).

As in many highly urbanized cities, D.C. experiences increased stormwater runoff because of development. When not managed properly, this runoff stresses sewer systems and degrades aquatic ecosystems (DOEE and CWP 2013). Approximately 43% of D.C.'s natural groundcover has been replaced with impervious surfaces, which accumulate pollutants, and during storm events those pollutants rapidly wash into downstream waterways (DOEE and CWP 2013). Unmanaged stormwater overloads streams and storm sewers (Beecham and Chowdhury 2012) and increases combined sewer overflow events and adverse downstream impacts such as flash flooding, channel erosion, surface and groundwater pollution, and habitat degradation (Semadeni-Davies et al 2008; Willems and Olssen 2012; DOEE and CWP 2013). In the context of stormwater management and water quality, TN, TP, and TSS are important parameters used to assess and regulate the levels of nitrogen, phosphorus, and total suspended solids in a water body. These parameters are central to the Stormwater Consolidated Total Maximum Daily Load (TMDL) Implementation Plan in the District of Columbia (D.C.). In urban areas, increased retention of runoff by green spaces can reduce flooding, thus lowering risks to human life and infrastructure. By combining rainfall data, land use characteristics, soil type, and slope information, stormwater engineers and planners can predict the potential impacts of stormwater runoff on drainage systems, water bodies, and the surrounding environment. This information is essential for designing and managing effective stormwater infrastructure to minimize flooding and protect water quality.

Given these highly valuable, yet often unaccounted for services of such systems, numerous global and regional efforts have started to incorporate ecosystem service (ES) science to advance biodiversity objectives. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) was established to "strengthen the science-policy interface for biodiversity and ecosystem services for the conservation and sustainable use of biodiversity, long-term human well-being and sustainable development" (IPBES 2012) and the EU Biodiversity Strategy to 2030 uses ES mapping and valuation to bolster biodiversity conservation approaches (Negre 2020). Major initiatives, such as the Millennium Ecosystem Assessment (Reid 2005) and The Economics of Ecosystems and Biodiversity (TEEB 2010) established a significant need to link urban ecosystem services (UES) to human wellbeing, leading to increased attention on urban greenspace policy.

Recently, in response to need, several authors have developed spatially explicit tools [e.g., Integrated Valuation of Ecosystem Services and Tradeoffs

(InVEST) (National Capital Project 2022), Artificial Intelligence for Ecosystem Services (AIRES) (Villa et al. 2009), Land Utilization and Capability Indicator (LUCI) (Jackson et al. 2012)] to quantify, value, and map ecosystem services at broad spatial scales (e.g. Boyd & Banzhaf 2007; Willemen et al. 2008; Johnston & Russell 2011; Gomez-Baggethun & Barton 2012). There are numerous studies that have quantified and mapped ecosystem services in cities around the world. However, few studies have explicitly quantified multiple UES and assessed or integrated their synergies and tradeoffs in local land-use decisions and planning (Davies et al. 2011; Gomez-Baggethun & Barton 2013; Kronenberg and Hubacek 2013; Kremer et al. 2016; Lourdes et al 2022). For example, Davies et al. (2011) conducted a city-wide quantification of above-ground carbon storage in the heavily urbanized European city of Leicester, U.K. to show the potential benefits of accounting for, mapping, and effectively managing aboveground vegetation carbon storage. Goldstein et al. (2012) quantified ecosystem service and economic implications to help design a land-use development plan that balances multiple private and public values on North Shore land holdings on the Island of O'ahu, Hi. Baró et al (2014) quantified urban forest regulating services to determine their contribution to air quality and climate change mitigation policies in Barcelona, Spain. While many of the earlier UES assessments were often restricted to one or two services and/or a coarser resolution of analysis, newer studies are assessing multiple UES and making use of high spatial resolution datasets. For example, Kremer et al. (2013) assessed multiple UES across New York City and discussed how different planning priorities affected the distribution of ecosystem service values. Derkzen et al. (2015) used high resolution data to quantify and map the supply of multiple UES on urban greenspace (UGS) types in Rotterdam, the Netherlands and demonstrated that the amount, the composition, and configuration of UGS plays a major role in ecosystem service provision. A very recent study by Lourdes et al. (2022) combined biophysical ecosystem service models with multicriteria analysis to characterize the spatial distribution of multiple UES in a rapidly developing, peri-urban catchment in Greater Kuala Lumpur, Malaysia, and identify areas where green infrastructure should be targeted.

While these studies have demonstrated the value of multiple UES assessments for socioeconomic policies, few, if any, have discussed how this methodology can be directly applied to strengthen biodiversity sensitive urban planning and corridor design by justifying the conservation of high development value lands as shown here. Yet, quantifying, valuing, and mapping UES at a high-resolution would allow urban planners to identify high-impact areas for conservation and restoration that would provide value to both local biodiversity and community well-being (Daily and Matson 2008; Burkhard 2012). Additionally, valuing multiple UES allows for the mapping of several weighting scenarios that can support various policy preferences, such as strong investments in stormwater mitigation (Kremer et al. 2015).

The objective of this study was to develop a framework to facilitate the integration of multiple UES valuation into biodiversity sensitive urban planning and corridor design. I quantified the biophysical values of seven ecosystem services of major importance in urban areas – climate change mitigation, heat mitigation, storm-water retention, extreme flood risk mitigation, and retention of three common pollutants– to create the first high-resolution multiple UES assessment of all green infrastructure in Washington, D.C. I then aggregated the biophysical output value of each individual service to 46 distinct planning areas and to 33 viable biodiversity corridors, and assessed and compared the types, magnitude of supply, and proportion of each UES within and across each corridor and planning area. Lastly, I computed the amount and proportion of each significant land use and land cover (LULC) class within each proposed corridor and identified which land cover types were the most significant predictors of individual and combined UES to inform future restoration efforts.

3.2 Methods

3.2.1 Study Area

I focused this analysis on Washington, D.C., the capital city and federal district of the United States of America. The city is located at the junction of two tidal rivers, the Anacostia and Potomac, between the Appalachian Piedmont and Atlantic Coastal Plain, providing a diversity of habitat types for subsequent wildlife species. Nearly one quarter of the city's land area (30.82km^2) is devoted to park and open space resources with 52,205m^2 of park per 1,000 residents (one of the highest per capita ratios of any city in the United States) (Urban Parks and Programs (n.d)). The federal government currently owns approximately 24% of the land in Washington, D.C. and nearly 90% of all parkland (6700 acres) is under National Park Service (NPS) jurisdiction (Urban Parks and Programs (n.d)). The Potomac and Anacostia Riverbanks are home to several low-lying tidal wetlands that are part of the NPS' National Capital Region Inventory and Monitoring Network (NCRN) parks. The wetland ecosystems of these parks and the neighboring coastal populations are threatened by accelerating sea level rise, and given its location is between 0 and 125m above sea level D.C. is likely to see record flooding events as early as 2040 (NPS (n.d); Strauss et al. 2014).

The Department of Parks and Recreation (DPR) oversees much of the non-federal park space, principally located in neighborhood parks. DPR relies on multiple agencies and park partners to plan, build, maintain, and program these public spaces. The D.C. Government is committed to protecting natural areas while also providing all residents with convenient access to nature and greenspace. However, like many highly urbanized areas, complex jurisdictional responsibilities and limited resources lead to challenges in balancing the needs of a variety of park users while protecting sensitive, valuable resources and wildlife within the parks (NPS (n.d)).

3.2.2 InVEST Models

InVEST is a suite of open-source, spatially explicit software models, developed by the Natural Capital Project that allow decision-makers to assess and map the delivery of ecosystem services and the associated tradeoffs in their decision-making processes (Hamel et al. 2020). InVEST models use LULC patterns to estimate the provision, value, and tradeoffs of ecosystem services. The results can be explained in either biophysical or economic terms. Here, I used a subset of four models (Table 3-1) within the new Urban InVEST modeling platform (version 3.11.0) (Hamel et al. 2020) to quantify and map the biophysical values of seven UES that are of major relevance to human wellbeing: climate change mitigation (CC), heat mitigation (HM), storm-water retention (SWR), flood risk mitigation (FRM), and pollution/nutrient retention (total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP)). I used a 2017/18 Chesapeake Conservancy's high-resolution (1m²) land cover dataset (CBPO 2022) as the reference LULC layer for all UES models along with additional GIS datasets including a soil type map from the United States Department of Agriculture's National Resources Conservation Service's Soil Survey Geographic Database (Soil Survey Staff) and a map of average annual precipitation from the U.S. Climate Normals (NOAA 2021). Then, I mapped the per-pixel (1m²) estimated amount of biophysical output produced by each UES, across 15 general land use categories in D.C. (supplemental information on classification of each LULC class is provided in Appendix B). Further details on each of the four InVEST UES model parameters and sources are provided below and in Appendix B.

Table 3-1: The subset of InVEST urban models used in the study and the name of the service and supply metrics.

InVEST Model	Ecosystem Service	Supply Metric
Carbon Sequestration and Storage	Climate Mitigation (CC)	Total current carbon stored and sequestered (Mg)
Urban Cooling	Urban Heat Mitigation (HM)	Reduction in air temperature by vegetation (°Celsius)
Urban Flood Risk Mitigation	Extreme Weather runoff retained/Natural Hazard Protection (FRM)	Extreme weather runoff volume retained (m^3)
Urban Storm Water Retention	Stormwater flow regulation (SWR)	Annual stormwater runoff retained (mm/yr)
	Pollution/nutrient retention - Suspended Solids (TSS)	Avoided nutrient load (kg/yr)
	Pollution/nutrient retention – Nitrogen (TN)	Avoided nutrient load (kg/yr)
	Pollution/nutrient retention – Phosphorous (TP)	Avoided nutrient load (kg/yr)

3.2.2.1 Carbon Sequestration and Storage

The InVEST Carbon Storage and Sequestration model aggregates the amount of carbon stored in four carbon pools (aboveground biomass, belowground biomass, soil carbon, and dead organic matter) to estimate the total amount of carbon (Mg) currently stored in each LULC class (Sharp et al. 2020; Table 3-1). The carbon model is exceptionally straightforward, relying solely on those four carbon pools and a land cover map. While it offers a convenient means of mapping carbon pool values to LULC, it deliberately omits any biophysical complexities or dynamics, such as tree growth, evolving soil chemistry, or accounting for the impacts of changing temperature and precipitation (Sharp et al. 2020).

I used the Carbon Dioxide Information Analysis Center's (CDIC) global biomass carbon lookup tables (Ruesch & Gibbs 2008) to determine the carbon stock (metric tons of Carbon) in aboveground and belowground biomass. I referenced the broadleaf forest class within the sub-tropical moist deciduous forest Food and Agriculture Organization (FAO) Ecofloristic zone, for North America (Eggleston et al. 2006). These average carbon pool values were calculated following the Intergovernmental Panel on Climate Change's (IPCC) 2006 Tier-1 methodology for estimating vegetation carbon stocks using the globally consistent default values provided for aboveground biomass (Eggleston et al. 2006; Ruesch & Gibbs 2008). I used the IPCC (2006) default reference carbon stock values for leaf litter in a broadleaf deciduous forest as well as default estimates of carbon stocks in soils with low activity clay (LAC) and soils with restricted drainage (wetland soils; Aalde et al. 2006).

3.2.2.2 Urban Cooling

The InVEST urban cooling model estimates a temperature reduction by vegetation (degrees Celsius) (Table 3-1), based on shade (the proportion of area in each land use class that is covered by tree canopy at least 2m high), evapotranspiration (amount of water that vaporizes from land into the air over a given period of time), albedo (the proportion of solar radiation that is directly reflected by each LULC class), and distance from cooling islands (e.g. parks; Sharp et al., 2020). I used the Consortium for Spatial Information's (CGIAR-CSI) global map of annual potential evapotranspiration (ET0_v3) based on WorldClim climate data (Fick and Hijmans 2017), as the reference evapotranspiration layer (Zomer and Trabucco 2022). I assigned surface albedo values for each LULC class based on the "local climate zone" (LCZ) classification system from Stewart and Oke (2012).

3.2.2.3 Urban Storm Water Retention

The InVEST urban stormwater retention model quantifies two aspects of runoff retention: annual stormwater runoff retained (mm/yr) and avoided nutrient load (kg/year) in response to annual precipitation (Sharp et al. 2020; Table 3-1). I used runoff retention and avoided pollution values based on the average annual precipitation (1062 mm/year) in Washington, D.C. (derived from NOAA Climate Prediction Center). I assigned runoff coefficient values for each LULC class and A/B/C/D soil hydrologic groups based on the rational method for a 2-year return period (McCuen 2016). In stormwater models, runoff coefficients play a crucial role in estimating the amount of runoff generated from rainfall or precipitation events. Runoff coefficients are dimensionless values that represent the proportion of rainfall that becomes surface runoff. They are used to convert the total rainfall volume into the volume of water that will flow over the land surface and eventually enter stormwater systems, streams, or other drainage facilities. The rational method is one of the most used procedures for calculating peak flows from small drainages typical of urban areas. I assigned local land use-based event mean concentrations (the total mass of pollutant discharged, divided by the total runoff volume) by pollutant type (TN, TP, and TSS), which were derived from D.C.'s Stormwater Consolidated TMDL Implementation Plan (DOEE 2016).

3.2.2.4 Urban Flood Risk Mitigation

While related to the urban stormwater retention model, the urban flood risk mitigation model assesses runoff retention in response to a single large ("extreme") storm event (Sharp et al. 2020). The model calculates the amount of runoff retained (m^3) per pixel relative to the storm volume (Table 3-1). I used a hypothetical 100-year storm (211.33mm of rain in 24 hours) to calculate runoff retention values, because this is indicative of an extreme storm event in D.C. (NOAA 2006). I obtained rainstorm precipitation values from the NOAA Atlas 14-point precipitation frequency estimates for Washington, D.C., which provide point precipitation frequency estimates for 5-minute to 60-day durations at average recurrence intervals of 1-year to 1,000-year (NOAA 2006). I assigned a runoff curve number to each LULC class based on combinations of hydrological soil group and land cover according to the widely used techniques and methodologies developed in USDA TR-55 (USDA, 1986) and USGS 6-A31 (Westenbroek et al. 2010). I derived Hydrologic Soil Groups for D.C. from the USDA's Natural Resources Conservation Service (NRCS) Soil Survey Geographic Database (SSURGO) (Soil Survey Staff 2022).

3.2.3 Planning for Conservation and Restoration Corridors

UES valuation provides urban planners and policymakers with a better understanding of the flows between supply and demand. Furthermore, many UES are spatially dependent. For example, stormwater control and flood protection services are particularly important to reduce risks in flood-prone locations (Andersson et al. 2015; Kremer, Hamstead, & McPhearson 2016). Therefore, to help target areas where environmental policies need to be implemented and monitored, it is important that ES valuations are quantified and mapped at planning-relevant scales. To delineate areas that are of high value to biodiversity, I utilized proposed biodiversity corridors from chapter 2. The distribution of land cover within these proposed corridors included 31% forest cover (2.75 km^2), 18% impervious surfaces and structures (1.5 km^2), 5% natural succession (0.3 km^2), 6% turf grass (0.4 km^2), 9% tree canopy (0.7 km^2), 6% tree canopy over impervious (0.4 km^2), and 16% tree canopy over turf grass (1.4 km^2).

To define planning-relevant areas for conservation and restoration, I divided the proposed corridor network into 33 distinct corridors based on neighborhood clusters within D.C. (Figure 3-1). The district does not have official neighborhood boundaries, but uses neighborhood clusters (henceforth, 'planning areas'), set in the early 2000s based on the expert opinion of the Office of Planning staff. These boundaries have been maintained to allow for comparable comparisons across time and continue to be used by various local agencies and independent analysts.

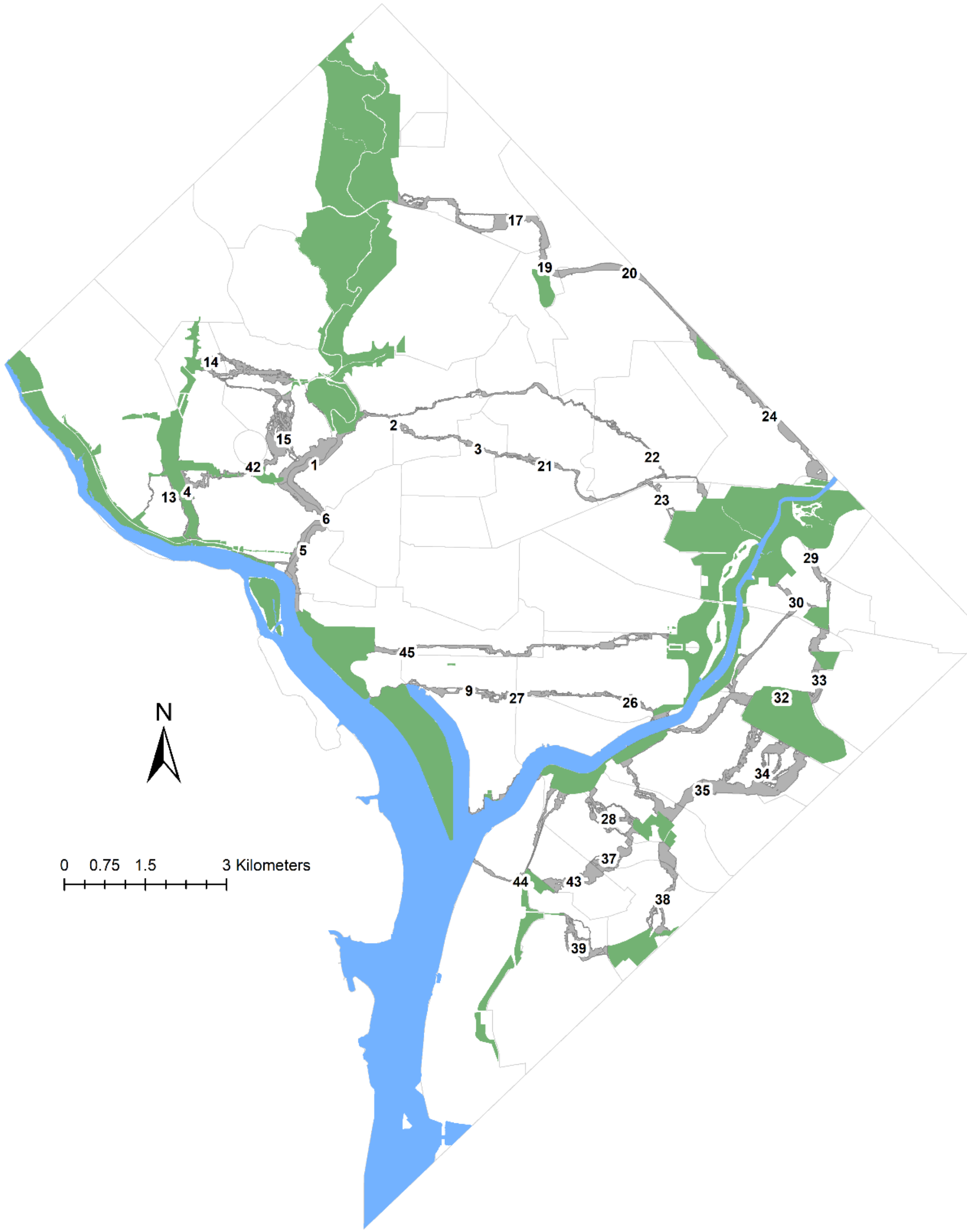

Figure 3-1: Proposed biodiversity corridors (n=31) (gray), connecting the major parks (green), that were used to delineate the planning-relevant areas for conservation and restoration, mapped onto the 46 distinct planning areas within Washington, D.C.

3.2.4 Mapping and Analysis

3.2.4.1 District-wide Comparison

To allow for comparison of UES across D.C., I normalized the UES model outputs to values between 0 and 1, using the following equation:

$$UES_{norm} = (UES - \min(UES)) / (\max(UES) - \min(UES)),$$

where UES_{norm} represents the normalized model output value of the UES at a given pixel, and min(UES) and max(UES) represent the minimum and maximum value respectively across that data layer. Then, I summed the individual normalized UES raster datasets to visualize locations with overlapping services and illustrate the concurrence between two or more services.

3.2.4.2 Planning Areas Comparison and Multicriteria Corridor Analysis

I aggregated each individual UES model output to the 46 distinct planning areas within D.C. and normalized the values by each planning unit. This allowed for comparisons of the spatial pattern of each service relative to the planning area size. To identify corridors that maximized biodiversity as well as the supply of services, I aggregated each individual UES model output to the 33 viable biodiversity corridors, normalized the values between 0 and 1, and summed the normalized values of each service to determine a total ES supply score for each corridor. Each viable corridor would have high or low scores, constrained to a minimum of zero and maximum of seven, depending on the number and magnitude of services they provide. Using the *raster* (Hijmans and Van Etten (2012), *ggplot2* (Gómez-Rubio 2017), *factoextra* (Kassambara and Mundt 2020), *ggfortify* (Horikoshi and Tang 2018), and *plyr* packages in R (version 4.04;R Core Team and contributors worldwide 2020) and

ArcMap (version 10.8), I quantified, mapped, and assessed the type, magnitude, proportion, and spatial distribution of each UES within and across planning areas and corridors.

To inform future restoration efforts aimed toward improving biodiversity and the provision of UES within the proposed corridors, I calculated the amount and proportion of each LULC class within each proposed biodiversity corridor. Then, I performed a multiple linear regression analysis, using the *stats* package in R, to determine which land use type was the most important predictor of overall UES supply. To account for varying sizes of the corridors, I used total UES output (normalized by area) for each corridor as the response variable and proportion of coverage of each land cover class as the predictor variables in the regression model.

3.2.4.3 Interactions Among Ecosystem Services

Interactions among ES occur when multiple services respond to the same driver of change (Bennet, Peterson, and Gordon 2009). Tradeoffs occur when one service is enhanced at the expense of another service being reduced, whereas ecosystem service synergies occur when more than one service is enhanced simultaneously. Synergies can be managed to improve the multifunctionality of the landscape and overall well-being of people (Sachs and Reid 2006). To investigate interactions between pairs of UES within the corridors, I performed a pairwise correlation test using the *PerformanceAnalytics* (Peterson et al. 2018) package in R (version 4.04); R Core Team and contributors worldwide 2020). I used Spearman's correlation coefficient (p) where a higher magnitude value indicated stronger correlations, the significance cutoff was assumed to be $p < 0.05$, and the direction of correlation, positive and negative, indicated a synergy or tradeoff between each UES respectively.

3.3 Results

3.3.1 District-Wide Distribution of UES Supply

I estimated that current total vegetation in D.C. provided ~1 teragram (Tg) of carbon and an overall average temperature reduction by vegetation of 0.6° Celsius. The vegetation in D.C. also helped to retain over 7 million m^3 of stormwater per 100-year storm event, a total of 69,385 meters of annual stormwater, and over 4 million kg of TSS (Table 3-2). I found low values for each UES within 10 km^2 of highly developed core areas of D.C. These values increased moving from the center of D.C. toward less dense development; characterized largely by single family housing and open park space (Figure 3-2).

Table 3-2: Annual total production of seven urban ecosystem services (UES) and proportion of the district wide total for all viable biodiversity corridors within Washington, D.C.

Ecosystem Service	Supply Metric	District Wide Total	Corridor Total	Proportion
Climate Mitigation (CC)	Total Current Carbon Stored and Sequestered	1,201,697 (Mg)	166,123(Mg)	13.8%
Urban Heat Mitigation (HM)	Average Reduction in air temperature by vegetation	0.6° (Celsius)	0.7° (Celsius)	
Extreme Weather runoff retained/Natural Hazard Protection (FRM)	Extreme weather runoff volume retained (m3)	6,164,735 (m3)	545,719(m3)	8.9%
Stormwater flow regulation (SWR)	Annual stormwater runoff retained	69,385,168 (mm/yr)	5,107,132(mm/yr)	7.4%
Pollution/nutrient retention - Suspended Solids (TSS)	Avoided Total Suspended Solids load (kg/yr)	4,282,683 (kg/yr)	275,048(kg/yr)	6.4%
Pollution/nutrient retention – Nitrogen (TN)	Avoided Nitrogen load (kg/yr)	130,922 (kg/yr)	9,325(kg/yr)	7.1%
Pollution/nutrient retention – Phosphorous (TP)	Avoided Phosphorous load (kg/yr)	11,851 (kg/yr)	765(kg/yr)	6.5%

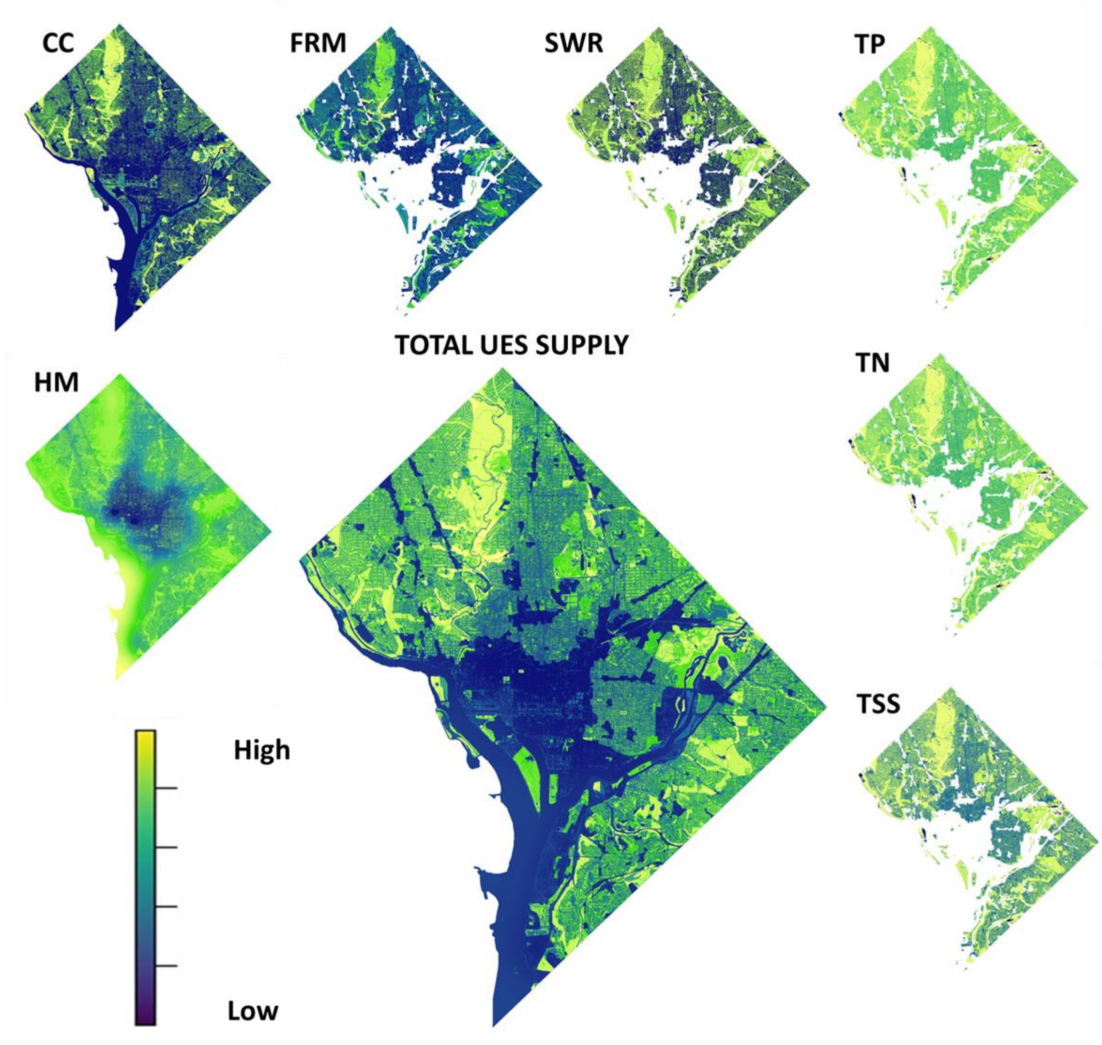

Figure 3-2: Figure 2. Spatial distribution of the seven urban ecosystem services UES: storm-water retention (SWR), pollution/nutrient retention [total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP)], carbon storage (CC), flood risk mitigation (FRM), and heat mitigation (HM) after normalizing (values between 0 and 1) and the overlap of the seven UES (by summing the seven individual urban ecosystem service maps), in Washington, D.C.

3.3.2 UES Distribution Across Neighborhood Planning Areas

At the planning area scale, I identified variations in UES due to differences in the amount of land cover type and corresponding supply of each service. The planning areas with the lowest biophysical output relative to their size were in downtown D.C. in the northwest quadrant (Figure 3-3). That region was characterized by high amounts of impervious surface, and UES supply was consistently low. Because central neighborhoods were the most densely populated and the least green, UES increased with increasing distance from the city center (Figure 3-3). The upper north-west region of D.C., which includes the neighborhoods of Rock Creek Park, Forest Hills, Van Ness, and Chevy Chase has the highest value for SWR, pollution retention (TN, TP, and TSS), and FRM because these services are primarily being supplied by the same types of green land cover. Whereas the central downtown region and surrounding neighborhoods such as Union station, Howard University, LeDroit Park, Cordazo/Shaw, and Navy Yard (planning areas 3 and 27) had the lowest overall biophysical output of UES. Historic Anacostia, in the Southeast Quadrant and southeast of the Anacostia River had high output for all UES relative to its size. Notably, the spatial distribution of CC and HM differ from the other services and each other due to the variation in the amount of land cover that supplied those specific services.

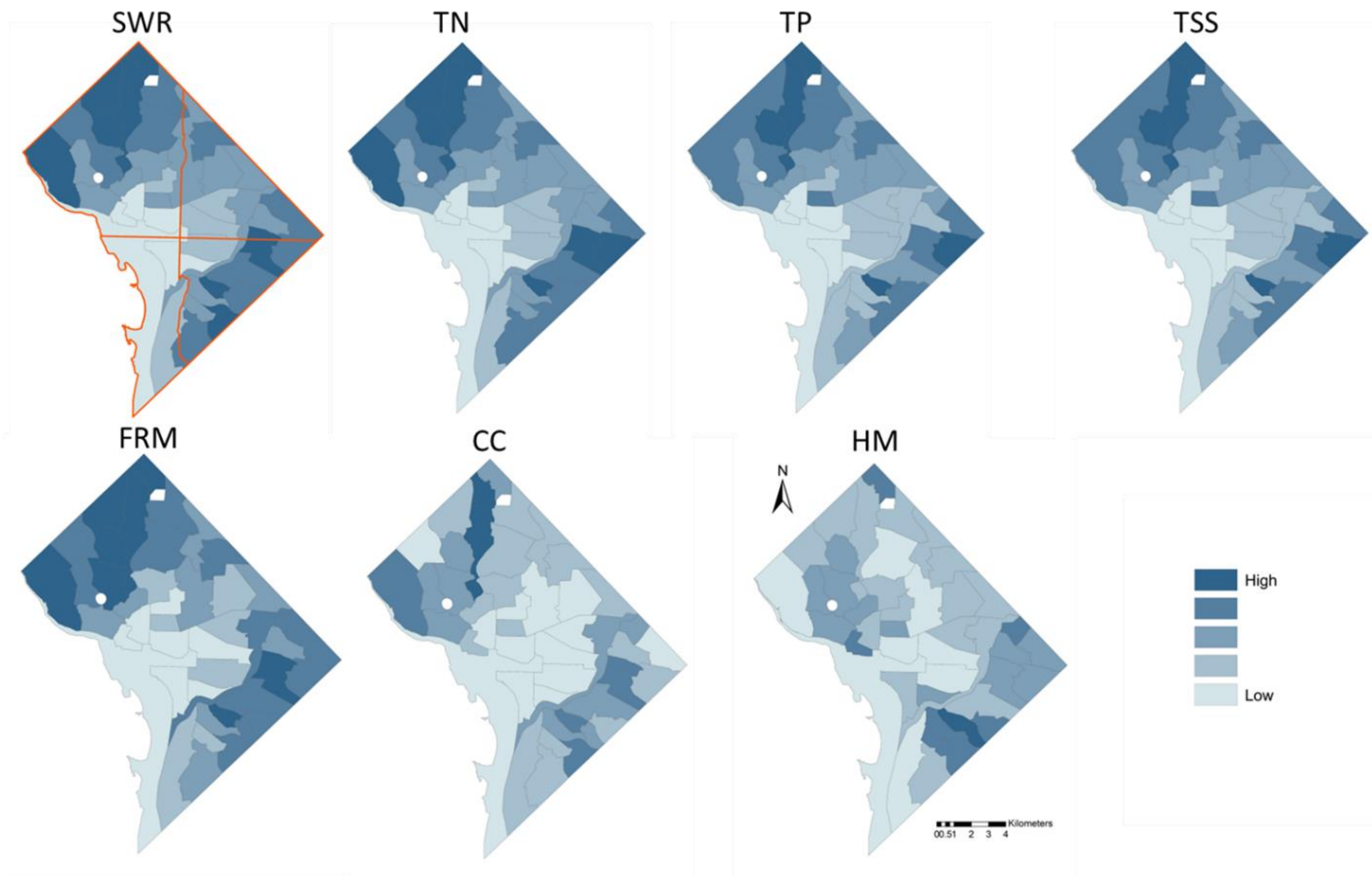

Figure 3-3: Distribution of the seven urban ecosystem services (normalized by area): storm-water retention (SWR), pollution/nutrient retention [total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP)], carbon storage (CC), flood risk mitigation (FRM), and heat mitigation (HM) across planning areas. D.C.'s four quadrants are delineated in orange on the SWR map.

3.3.3 Synergies and Tradeoffs between UES

Carbon storage (CC), SWR, pollution retention (TN, TP, and TSS), and FRM demonstrated significant positive correlations ($p \geq 0.93$; p-value = 0.001) (Figure 3-4). There was moderate correlation between HM and CC and FRM ($p = 0.46$; p-value = 0.01 and p = 0.39; p-value - 0.01 respectively). Heat mitigation (HM) was not significantly correlated to SWR, TSS, TP, or TN.

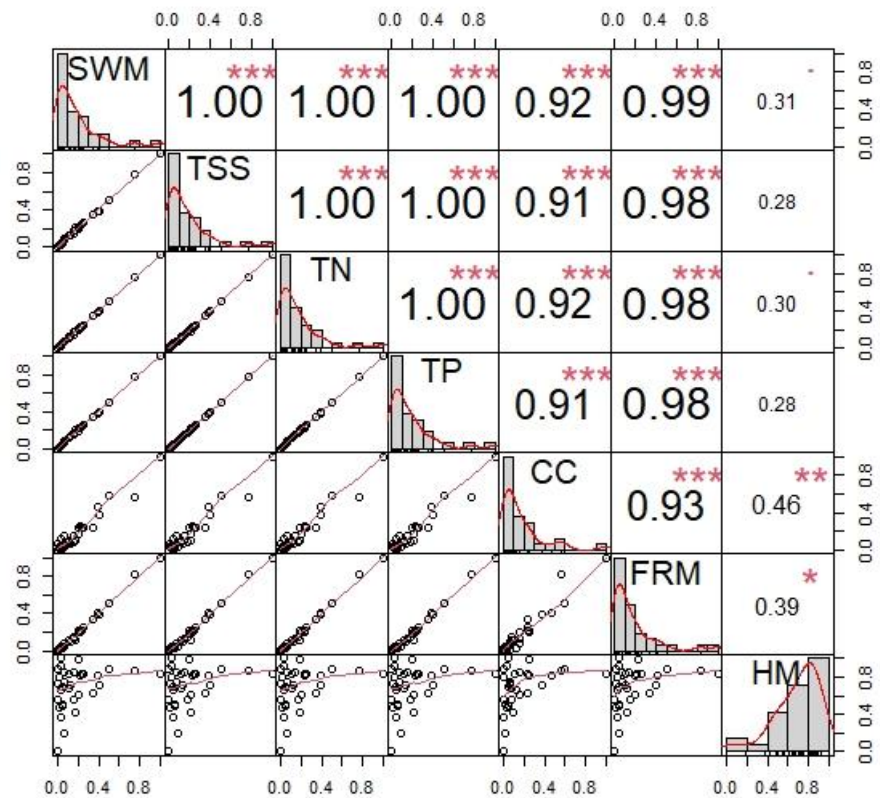

Figure 3-4: Spearman's pairwise correlation between seven normalized urban ecosystem services (UES) supplied by each planned corridor - storm-water retention (SWR), pollution/nutrient retention [total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP)], carbon storage (CC), flood risk mitigation (FRM), and heat mitigation (HM). The asterisks indicate level of significance where p-values (0.0, 0.001, 0.01, 0.05, 0.1) <=> symbols ("***", "**", "*", ".") All correlations were significant at $p \leq 0.05$, except for the correlation between HM and TP, TSS, TN, and SWM ($p \geq 0.1$).

3.3.4 Corridor Analysis

I estimated that current total vegetation in the 33 proposed corridors provides ~166,123 Mg of carbon (13% of the district-wide total) and an overall average temperature reduction of 0.7° Celsius. The corridors were estimated to retain over 545,719 m^3 of stormwater per 100-year storm event, a total of 5,107 meters of annual stormwater, and over 285,138 kg/yr of total pollutants (Table 3-2).

Eleven proposed corridors had above-average total ES supply scores (Figure 3-5). Corridor 34, which is part of the Fort Circle Parks connecting Fort Dupont and Fort Stanton in the S.E quadrant of the D.C., accounted for 26% of the land area of its corresponding planning area and received the highest total ES supply score (6.84) indicating a high supply rate of all UES (Table 3-3). Corridor 15, which contained a section of Normanstone parkway in the Northeast quadrant of the D.C., had the second highest ES supply score (5.33), and accounted for 22% of the land area of its corresponding planning area. Additionally, both corridors accounted for the largest relative proportion of carbon storage, 60% and 40% respectively (Table 3-3) and both corridors accounted for >25% supply of all other UES within their corresponding planning areas.

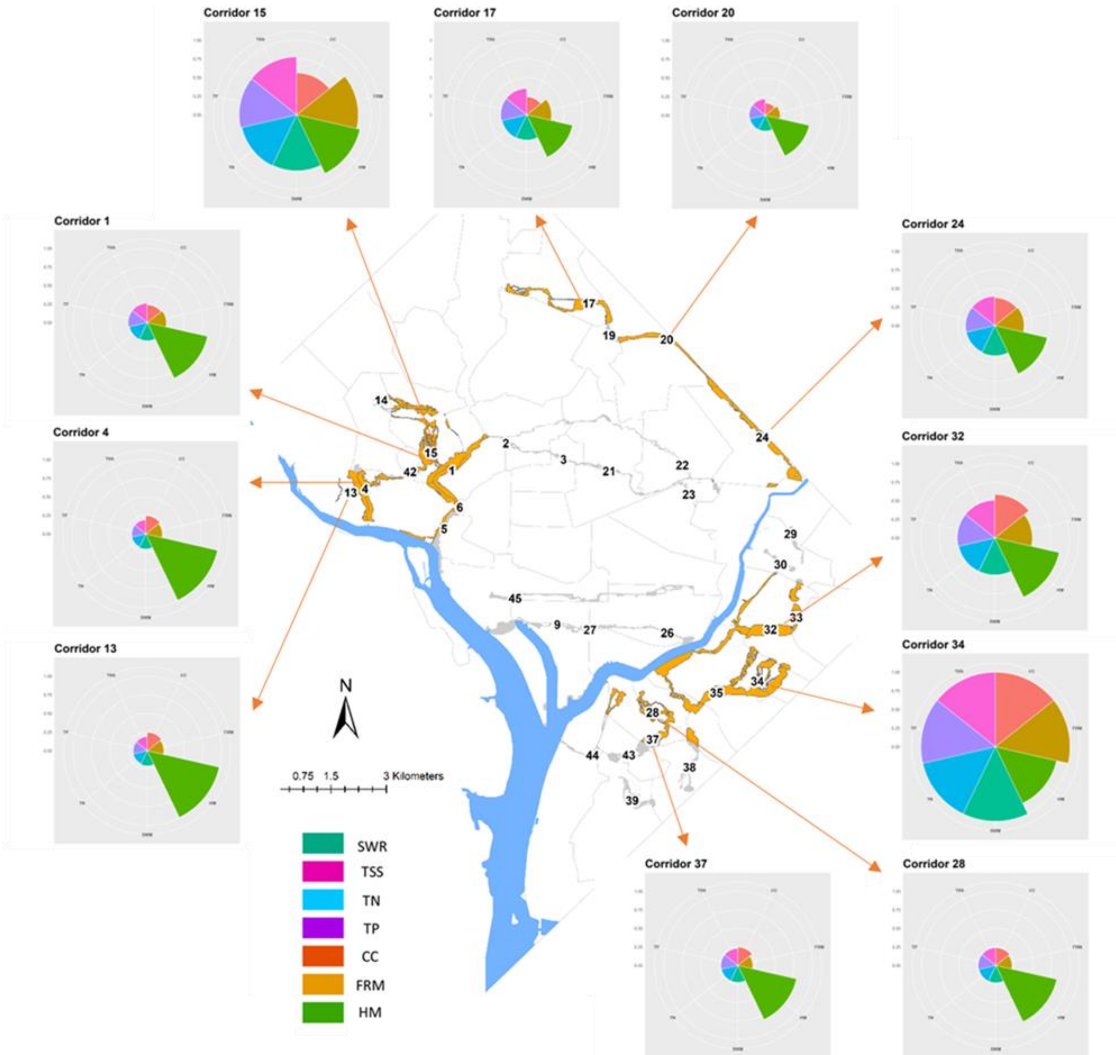

Figure 3-5: Rose plots showing the distribution and magnitude (petal length) of each urban ecosystem service (UES): storm-water retention (SWR), pollution/nutrient retention [total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP)], carbon storage (CC), flood risk mitigation (FRM), and heat mitigation (HM), for the 11 planned corridors that had above average ES supply ($\mu = 1.76$, $\sigma = 1.4$).

Table 3-3: The supply (S) of seven urban ecosystem services (UES) - storm-water retention (SWR), pollution/nutrient retention [total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP)], carbon storage (CC), flood risk mitigation (FRM), and heat mitigation (HM) - provided by each biodiversity corridor (C) (ordered by total supply score) and the proportion (Prop) of supply relative to their corresponding planning area in Washington, D.C. The total ES supply score (Score) is the sum of the normalized value for each UES. The top 11 rows are those corridors with the highest average supply for all seven UES.

C	**Area (km2)**		**SWR (mm/yr)**		**TSS (kg/yr)**		**TN (kg/yr)**		**TP (kg/yr)**		**CC (Mg)**		**HM (° Celcius)**		**FRM (m3)**		**Score**
	Tot	Prop	Supply	Prop	Supply	Prop	Supply	Prop	Supply	Prop	Supply	Prop	Supply	Prop	Supply	Prop	
34	1.26	0.26	865543.83	0.36	45394.81	0.30	1577.04	0.34	126.68	0.31	29271.41	0.60	0.74	0.28	94439.15	0.46	6.84
15	0.89	0.22	650066.90	0.29	35298.65	0.25	1195.28	0.28	98.28	0.26	16464.30	0.40	0.75	0.25	78200.76	0.34	5.33
32	0.64	0.15	432069.44	0.18	23078.04	0.17	790.16	0.18	64.32	0.17	17159.49	0.21	0.76	0.16	47660.53	0.20	3.99
4	0.55	0.16	339361.37	0.28	17403.95	0.22	605.96	0.26	48.58	0.23	13584.72	0.48	0.73	0.19	38433.70	0.36	3.22
24	0.44	0.12	351761.12	0.20	17914.63	0.16	624.67	0.19	50.01	0.16	11023.98	0.43	0.69	0.13	37053.00	0.27	3.08
17	0.39	0.09	292734.32	0.15	16046.78	0.12	539.61	0.14	44.65	0.12	7111.86	0.30	0.65	0.10	31433.35	0.20	2.59
1	0.31	0.16	212764.40	0.29	11828.64	0.20	393.05	0.26	32.88	0.21	7020.69	0.51	0.74	0.25	24169.28	0.41	2.34
13	0.21	0.02	176689.34	0.03	8623.01	0.03	306.38	0.03	24.11	0.03	7242.18	0.06	0.81	0.02	21035.56	0.04	2.24
28	0.31	0.23	202081.51	0.32	10911.68	0.28	371.02	0.31	30.39	0.29	7025.16	0.49	0.74	0.27	20461.43	0.36	2.24
37	0.40	0.20	197549.99	0.28	10486.16	0.23	361.07	0.27	29.24	0.24	7275.33	0.40	0.73	0.23	18885.08	0.32	2.18
20	0.26	0.06	182115.31	0.09	9942.20	0.08	334.59	0.09	27.67	0.08	4817.01	0.22	0.64	0.07	18680.66	0.12	1.82
21	0.23	0.04	144227.41	0.06	8070.78	0.05	267.34	0.06	22.43	0.05	2565.29	0.11	0.61	0.05	13953.62	0.07	1.48
38	0.21	0.09	135899.16	0.11	6928.48	0.10	235.46	0.11	19.30	0.10	4593.07	0.15	0.75	0.10	12965.90	0.12	1.76
26	0.35	0.08	147321.61	0.12	9783.49	0.10	286.94	0.11	26.91	0.10	2969.72	0.15	0.56	0.10	11241.48	0.15	1.42
30	0.18	0.07	98266.15	0.09	5456.55	0.08	181.77	0.08	15.17	0.08	2311.21	0.11	0.67	0.08	10581.80	0.10	1.33
14	0.15	0.05	101159.37	0.08	5490.97	0.06	185.98	0.07	15.29	0.06	3226.07	0.11	0.74	0.06	10378.09	0.09	1.52
39	0.19	0.04	112501.25	0.05	6153.90	0.04	206.43	0.05	17.12	0.04	1895.15	0.08	0.68	0.04	9778.47	0.06	1.40
19	0.07	0.02	59254.59	0.03	3154.44	0.03	108.38	0.03	8.79	0.03	1962.93	0.09	0.70	0.03	7048.42	0.05	1.17

Continued

Table 3-3 continued

43	0.19	0.12	62454.11	0.09	3311.82	0.08	114.12	0.09	9.24	0.08	4539.27	0.35	0.74	0.15	6806.56	0.12	1.36
23	0.07	0.02	42729.90	0.04	2179.58	0.03	77.30	0.03	6.09	0.03	1311.67	0.13	0.68	0.03	4354.60	0.06	0.98
5	0.15	0.11	46562.48	0.87	2464.74	0.86	85.04	0.87	6.87	0.86	1476.63	0.30	0.58	0.17	4247.79	0.88	0.79
33	0.04	0.01	24682.07	0.01	1270.96	0.01	44.76	0.01	3.55	0.01	1450.81	0.09	0.81	0.01	3698.01	0.03	1.20
2	0.14	0.04	52805.54	0.05	4010.71	0.05	107.38	0.05	10.96	0.05	738.61	0.05	0.45	0.05	3681.19	0.05	0.56
22	0.13	0.03	45858.35	0.02	2610.55	0.02	85.40	0.02	7.25	0.02	619.70	0.04	0.59	0.03	3506.63	0.03	0.79
35	0.04	0.02	27647.49	0.03	1472.57	0.02	50.58	0.03	4.11	0.02	1164.70	0.08	0.78	0.02	3246.74	0.04	1.13
45	0.55	0.03	33200.95	0.02	1781.44	0.02	59.82	0.02	4.96	0.02	3554.50	0.08	0.66	0.02	2980.29	0.02	0.96
29	0.02	0.01	18933.69	0.02	1009.48	0.02	34.65	0.02	2.81	0.02	285.61	0.01	0.72	0.01	2120.19	0.02	0.92
3	0.05	0.03	26480.07	0.07	1654.04	0.06	50.63	0.07	4.57	0.06	196.51	0.05	0.53	0.04	2077.26	0.08	0.54
9	0.19	0.05	14104.04	0.06	787.33	0.06	26.11	0.06	2.19	0.06	978.97	0.09	0.58	0.07	1193.85	0.07	0.59
44	0.13	0.02	8513.37	0.00	426.51	0.00	15.33	0.00	1.19	0.00	1937.00	0.03	0.69	0.02	1086.70	0.01	0.84
42	0.01	0.02	1789.39	0.01	101.48	0.01	3.33	0.01	0.28	0.01	185.04	0.05	0.76	0.02	319.37	0.01	0.90
27	0.01	0.00	3.98	0.00	0.60	0.00	0.01	0.00	0.00	0.00	31.60	0.01	0.37	0.00	0.18	0.00	0.00
6	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	133.78	0.02	0.62	0.01	0.00	0.00	0.56
Mean			154761.59		8334.82		282.59		23.21		5034.06		0.67		16536.96		1.76

Corridor 1, which comprised sections of Rock Creek and Potomac Parkway, and corridor 28 which connected Fort Stanton with Anacostia Park, also had high UES supply relative to their planning areas, predominantly CC and FRM (Table 3-3). Corridor 1 accounted for 51% of the planning area's CC and 41% FRM, while corridor 28 accounted for 49% CC and 36% FRM. In addition to corridors 34, 15, 1, and 28, corridors 4, 13, 17, 20, 24, 32, and 37 all supplied greater than average amounts of all UES and accounted for > 20% supply of one or more UES relative to their corresponding planning areas (Table 3-3; Figure 3-5). Notably, while corridor 5 only accounted for 11% of its related planning area's land, it contained 45% of its greenspace and accounted for the highest relative proportion of SWR (87%), avoided TSS load (86%), avoided TN load (87%), avoided TP load (86%), and FRM (88%; Table 3-3).

3.3.4.1. Land cover within Corridors

The coefficients and results of the land use type regression model that predicted UES supply are shown in Table 3-4. The results of the regression indicated that the proportion of turf grass, proportion of forest cover, proportion of natural succession, proportion of tree canopy, and proportion of tree canopy over turf grass explained 81% of the variance ($R^2 = 0.81$, $F(5, 27) = 28.4$, $p = < 5.919e-10$). Forest cover significantly predicted UES supply ($\beta = 0.89$, $p = 2.09e-11$). The relative proportion of natural succession, turf grass, and tree canopy did not significantly predict UES supply within the corridors.

Table 3-4: Table of multiple regression analysis showing the relationships of different urban greenspace to total urban ecosystem supply within viable biodiversity corridors in Washington D.C. Where NS = Natural Succession and TCTG = tree canopy over turf grass.

Residuals:				
Min	1Q	Median	3Q	Max
-0.6453	-0.4135	-0.2268	0.3541	1.8448

Coefficients:				
	Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	1.76035	0.10807	16.288	1.72e-15 ***
NS	-0.13144	0.12106	-1.086	0.2872
TCTG	-0.23371	0.12013	-1.945	0.0622
Forest	1.25199	0.11468	10.917	2.09e-11 ***
Tree_Canopy	0.03929	0.11713	0.335	0.7399
Turf_Grass	-0.06906 4	0.11304	-0.611	0.546

Signif. codes: 0.0 ‘*’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1**				

Residual standard error: 0.6208 on 27 degrees of freedom

Multiple R-squared: 0.8402, Adjusted R-squared: 0.8106

F-statistic: 28.4 on 5 and 27 DF, p-value: 5.919e-10

Forest cover across the corridors ranged from zero to 555,242 m² with only 6 corridors containing ≥ 50% coverage. One third of the corridors (n=11) contained less than 10% cover and 8 of those 11 contained no forests making them viable candidates for restoration efforts (Figure 3-6).

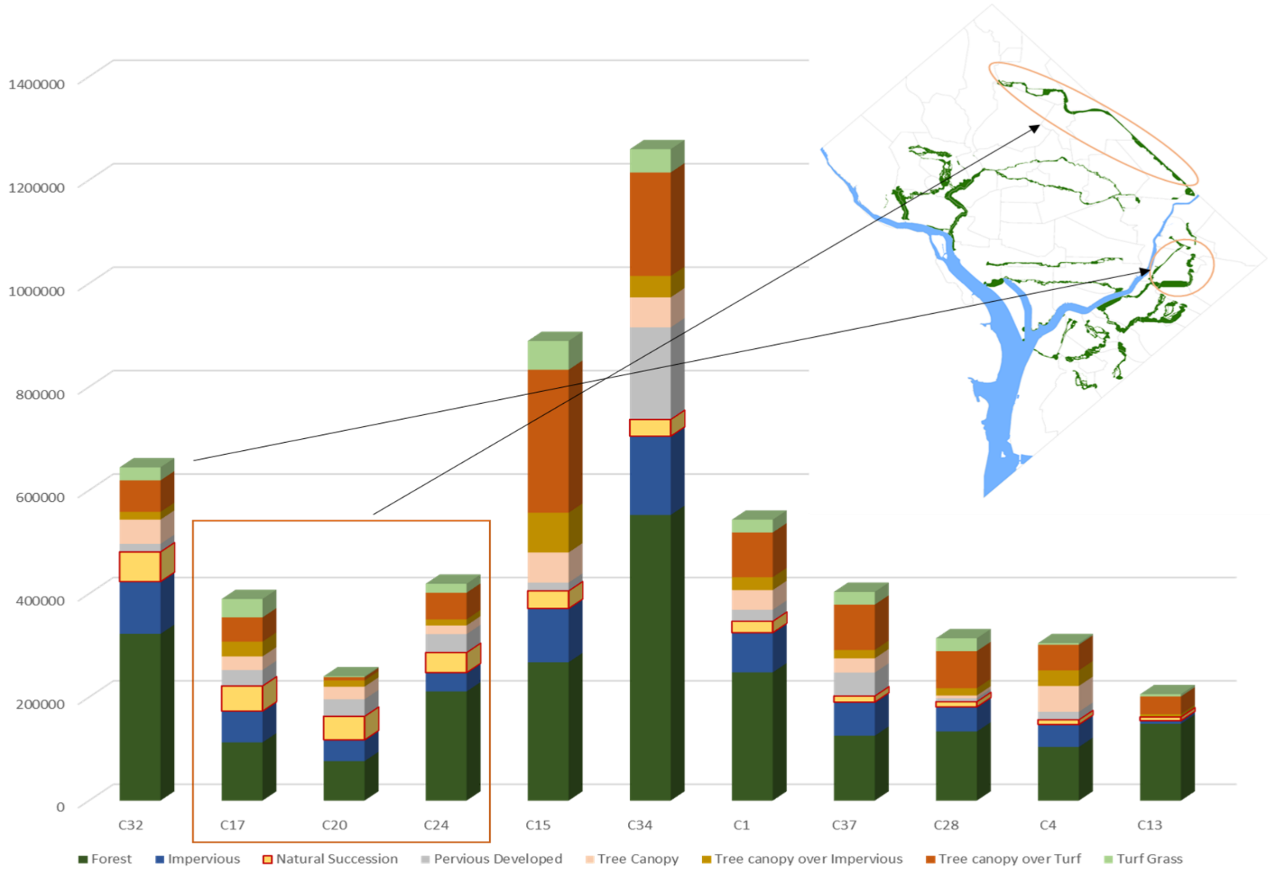

Figure 3-6: Distribution and amount (m²) of 8 land use/land cover (LULC) classes within the 11 viable biodiversity corridors with above average ecosystem service (ES) ordered by decreasing amount of natural succession land District of Columbia (D.C.).

Of the 11 corridors with above-average ES supply scores, corridor 34 and corridor 32, both located southeast of the Anacostia River (connecting Fort Dupont and Fort Stanton and connecting Fort Mahan and Fort Chaplin), contained the highest forest cover, followed by corridors 15 and 1 which were in the Northeast Quadrant of D.C. and connected the bottom section of Rock Creek Park to Glover Parkway and Dumbarton Oaks Park. The corridors with the greatest amount of barren, herbaceous, or scrub-shrub lands that were presumed to be undergoing either natural or managed succession were corridor 32 and 3 corridors (17, 20, and 24) along the northeast D.C. Boundary. Natural succession land cover is described as "barren, herbaceous, or scrub-shrub lands that are not classed as cropland, pasture, turf grass, or pervious developed. These are areas that are presumed to be undergoing either natural or managed succession and will eventually become forested although this process may take years to decades to complete. Abandoned mine lands are included in this class" (CPBO 2022).

3.4 Discussion

Here I developed a practical geospatial framework that employed multiple ecosystem service valuation and assessment to advance biodiversity-sensitive urban planning and corridor design. By quantifying and mapping ecosystem services at a very high spatial resolution ($1m^2$), patterns were revealed in the distribution of ecosystem service supply that might not have been captured at larger scales, allowing for highly targeted conservation and restoration initiatives. I identified several potential restoration corridors that contained high-value, underutilized land for restoration as an ecological asset that advances biodiversity needs while contributing to the overall quality of life in urban communities. Such knowledge is crucial to natural resource managers looking to achieve conservation gains and justify natural land uses in areas that are typically overlooked due to their high development value.

In addition to contributing to the emerging literature that attempts to combine ecosystem service assessments and multicriteria analysis to inform urban planning, the framework of analysis provided here is unique in that it addressed the trade-offs between ecosystem services and wildlife protection by assigning biophysical value to viable wildlife corridors to strategically identify areas with the greatest benefit per unit cost for conservation or restoration. By quantifying and mapping UES at a high-resolution, I was able to reveal patterns in the distribution of services within these corridors that were not apparent at larger scales, allowing for a more informative evaluation of the spatial relationship between UES and those local communities that benefit from these services (Naidoo et al. 2008). For example, corridor 5, a section of Rock Creek Parkway located in the highly urban center of D.C., had a low overall UES supply rate and was relatively small, yet it accounted for more than 85% of its corresponding planning area’s annual stormwater and pollution runoff, 30% of carbon storage, and retains 88% of runoff from extreme weather events, making this likely unrecognized greenspace both environmentally and economically highly valuable for the associated neighborhoods.

3.4.1 Application to Urban Policy and Planning

Urban areas have diverse needs and priorities, and cities and local governments face challenges in balancing those needs with limited resources. Not surprisingly local policymakers are now beginning to recognize the importance of multiple UES valuation in local decision-making processes (McKenzie et al. 2014) and urban planning officials are looking to quantitative assessments of UES to inform strategic planning (Tallis and Polasky, 2009; Burkhard et al. 2012; Gómez-Baggethun et al. 2013; Haase 2014).

In Washington D.C., stormwater management is a critical concern. This work identified locations where ecological restoration can be leveraged for stormwater management and flood mitigation. For example, corridors 34 and 32, which include portions of the neighborhoods of Greenway, Dupont Park, and Twinning are highly permeable and have the potential to absorb and retain water, reducing the burden on existing stormwater infrastructure. By targeting these areas for restoration, D.C. planners can improve its resilience to extreme weather events while enhancing biodiversity.

One of the challenges faced by natural resource managers in urban environments is justifying the conservation and restoration of natural lands in competition with high development value. By providing robust quantification of ecosystem services, this framework can help present a clear economic case for protecting and restoring targeted areas. For instance, conserving the valuable wetland portion of corridor 24, just north of the Baltimore Washington Parkway, may provide cost-effective water purification services for D.C. that would otherwise require expensive infrastructure investments.

This quantification of ecosystem services at a fine scale also highlighted areas within the city that need green infrastructure for enhancing the overall quality of life for urban communities. For instance, I identified several corridors connecting

Anacostia Park with Rock Creek Park and crossing residential areas (corridors 2,3,21,22, and 23) with limited access to green spaces and a high demand for recreational amenities. By strategically restoring and conserving ecological assets within these areas, city planning officials can address biodiversity needs while also providing residents with opportunities for recreation and improved mental well-being.

How money is allocated for the preservation and restoration of urban biodiversity is influenced by a wide range of interconnected social, cultural, and economic factors; including governance, economics, social networks, stakeholder preference, and societal constraints (Aronson et al. 2017). To address conservation funding challenges, some cities have established dedicated funding mechanisms for conservation initiatives, such as stormwater fees or park and open space levies. Stormwater fees are typically based on the amount of impervious surface area on a property, such as the area covered by buildings, driveways, and parking lots. In the District, if landowners in heavily urbanized biodiversity corridors expand green infrastructure, they may be eligible to apply for incentive programs like RiverSmart Rewards and Clean Rivers Impervious Area Charge (CRIAC), which would result in reductions of up to 55% off DOEE Stormwater Fees. Interestingly, 16 of the 33 proposed biodiversity corridors intersect all 10 of D.C.'s resilience focus areas. These areas were designated under the 2018 Resilient D.C Strategy as areas most vulnerable to acute disasters, storms, floods, and heatwaves. Six of these resilience focus areas lie along the Anacostia River and are under increased risk of flooding caused by climate change, erosion, and land subsidence (DOEE 2016). By providing incentives to residents, companies, and property owners to conserve or restore areas within these corridors, it may be possible to buffer these vulnerable communities from the negative impacts of ongoing environmental change. This unique framework allowed for the fine-scale identification of areas within D.C.'s resilience focus areas that would best serve socially, economically, and environmentally.

This analysis identified the network of corridors around Rock Creek Park as crucial. As mentioned above, corridor 5 has a relatively small UES rate, but a huge relative dividend in stormwater and pollution runoff, carbon sequestration and mitigating extreme weather events for the entire neighborhood. That corridor, in conjunction with corridors 14, 15, 1 and 6 collectively help with the critical issue of storm water management D.C. already faces. Residents in this affluent area could be provided with education and incentives such as tax breaks or rebates to replace impermeable surfaces with those that better absorb and retain water.

Having high-resolution mapping and quantitative analyses available can also help city planners introduce equity into budget and planning conversations. Historically, affluent communities have had a voice in governmental discussions due to access, available time, education, and a variety of other advantages. The representations of relative value presented here can help the public, stakeholders, NGOs, and policymakers understanding and advocate based off trade-offs inherent in any decisions around development, preservation, and restoration. By making the ecological value explicit, analyses such as these can lead not only to a better understanding of the overall value of green space, but a more transparent and equitable decision-making process during urban planning and restoration initiatives.

3.4.2 Broader Policy Relevance

Local policymakers interested in incentivizing conservation or restoration to meet state or national sustainable development goals, can use the information and methodology provided in this study for strategic urban conservation and restoration initiatives, which account for both biodiversity and socio-economic opportunity. For example, as part of his administration's efforts to combat climate change and transition to a clean energy economy, in 2021, President Joe Biden announced a federal goal to reach net zero carbon emissions by 2050. The goal was in line with the Paris Agreement, which aimed to limit global temperature rise to 1.5° Celsius above pre-industrial levels. Achieving net zero carbon emissions by 2050 would require significant reductions in emissions from all sectors of the economy to a level where any remaining emissions were offset by removal methods, such as reforestation. Because forest cover had significant positive correlation with biodiversity in D.C. (see Chapter 2) and was the strongest predictor of total UES supply, this study suggests reforesting as the most impactful restoration strategy within the biodiversity corridors.

There was a total of 395,613 m^2 of barren, herbaceous, or scrub-shrub lands, likely experiencing succession, within the viable biodiversity corridors used in this analysis. These areas would be ideal for targeted reforestation. The amount of potential reforestation land within each biodiversity corridor ranged from 0 to 57,369 m^2, with 5 corridors containing >32,000 m^2 of reforestation potential. Notably, the four biodiversity corridors with the highest reforestation potential lie within the Fort Circle Parks, managed by NPS. Furthermore, three of the priority conservation sites and nine restoration sites, identified in Chapter 2, intersect these four corridors. Focusing reforestation efforts within biodiversity corridors will not only aid in the preservation of biodiversity but will also aid in reducing the impact of the urban heat island effect and the energy needed for air conditioning. Reforestation will also stabilize soil to decrease runoff and sedimentation and increase nutrient cycling, lessening the frequency of flooding and droughts and improving water quality. Moreover, a recent analysis in Maryland suggested that by 2050, reforestation on currently non-forested areas in the neighboring Prince George's County would yield an average forest aboveground biomass of 3 kg C per m^2 (Hurtt et al. 2019, Ma et al. 2022). If that average projection was applied to mapped corridors in this study, D.C. could expect an increase in their forest carbon stock of at least 96 Mg of carbon by 2050. It is worth noting that reforestation initiatives would benefit from a simulated analysis of projected UES potential should the land be actively reforested versus being allowed to remain in natural succession.

3.5 Limitations and Uncertainties

Because the Urban InVEST models rely on biophysical estimate values related to LULC classes, a fundamental limitation of this study was the lack of availability of biophysical data related to the uniquely urban LULC classes. For example, carbon storage clearly differs among LULC types within urban areas, however, for the InVEST carbon storage model, I had to rely on carbon pool values derived from course scale studies that used generalized LULC classes over broad regions such as "tropical moist forest" for all areas described as "forest" or "tree canopy - other". Because the results of the model are only as reliable as the LULC classification used and carbon pool values supplied, this limitation could potentially lead to inaccuracies and biases in total carbon storage reporting. Nevertheless, because I scaled UES supply metrics to values between 0 and 1, this limitation should not have impacted comparisons across the normalized ES supply scores and thus the results of the multicriteria corridor analysis. Moving forward, UES assessments would benefit from a database of high spatial resolution biophysical data representative of urban areas, and further research is needed to document reliable urban metrics. Additional limitations related to the simplification of the Urban InVEST models which can introduce uncertainties. For example, the carbon storage and sequestration model and urban stormwater models do not fully account for the entire carbon and hydrologic cycles (Tallis et al., 2011). However, these models provide a sound starting point to quantify these ecosystem services.

3.6 Conclusion

Many municipalities are beginning to recognize the importance of ecosystem service valuation and are taking steps to incorporate them into their planning and decision-making processes. However, many polities and local agencies do not have clear knowledge on how to begin the process. I have provided a unique, accessible framework that makes the co-benefits of ecosystem services explicit. The methodology will inform biodiversity sensitive urban planning and corridor design and provide a framework for integrating biodiversity conservation into broader planning initiatives. For D.C. specifically, these results will raise awareness of the importance of biodiversity conservation in this unique urban area and point to specific plots of land that are not only currently benefiting the environment, human health, and quality of life, but indicate to what degree they are beneficial and identify areas that could easily be beneficial with proper planning. The high-resolution data allows city planners to make point-specific trade-offs – perhaps allowing more development on land of less ecological value while preserving or restoring specific plots with high value for biodiversity or storm water management. This targeted approach with visual representations of the relative value of land also allows for a more transparent and more equitable distribution of ecological gains.

Chapter 4: An Ad-hoc Translocation of Eastern Box Turtles (*Terrapene Carolina Carolina*).

Abstract

As the human population grows and wildlife habitat is lost, managers are increasingly reliant on hands-on management strategies such as translocation. Without much evidence suggesting success, moving a species or population from one location to another has been controversial for some taxa, especially for wildlife in urban areas. To examine the effects of repeated relocation of eastern box turtles (*Terrapene carolina carolina*) in an urban environment, and to facilitate the development of proper management strategies for box turtle conservation, I used data collected on 10 relocated turtles, from 2011 to 2014, to analyze movements and the establishment of home ranges of translocated turtles. Of the 10 translocated turtles, four settled immediately after the initial relocation and six turtles exhibited homing behavior and required repeated relocation back to the same release site. My analysis suggested two types of movement patterns for the four established turtles that required no repeated relocations after the initial release: (1) they settled immediately; and (2) their 50% and 95% kernel density estimator home range size decreased over time. For the six unestablished turtles that required repeated relocation back to the release site, results showed that the turtles exhibited a slight decrease in directional movement and began to settle as repeated relocation attempts increased. Overall results revealed a significant negative correlation between each relocation attempt and both size of movement area and step-lengths. My study population was small, but results suggested that home-range establishment may occur with repositioning individuals

post-release. Translocations should not proceed without sufficient investment from managers to carry out long-term monitoring of released individuals.

4.1 Introduction

According to the United Nations, over half of the world's human population now lives in urban areas (Grimm et al. 2000) and recent U.S. Census Bureau numbers reveal that 80.7 percent of the U.S. population lives in urban areas (United States Census 2010). Metropolitan areas contribute significantly to changes in biodiversity through the loss and fragmentation of habitat, and drastically alter land cover far beyond the city boundaries (Grimm et al. 2000). Such significant anthropogenic impacts on biodiversity provide a strong rationale for developing proactive management strategies for urban wildlife conservation.

Typically, urban wildlife communities are made up of generalist species that are adept at utilizing human resources. However, most wildlife species struggle in the face of extensive urbanization. For example, long-term studies of eastern box turtle (*Terrapene carolina carolina*) populations have indicated considerable declines in population size and density in areas experiencing only moderate anthropogenic impacts (Schwartz and Schwartz 1974; Stickel 1978; Williams and Parker 1987; Budischak et al. 2006). Many such species could benefit from retained city green space, increased urban landscape connectivity, and proactive population management. One such proactive management option is animal translocation, especially within the urban/suburban matrix. Historically, wildlife translocation was used to augment existing populations, but in more recent years, it has been used as a means to mitigate human-wildlife conflict. This method has been viewed as controversial and for many turtle species, unsuccessful, and therefore, undertaken only as a last option (Rittenhouse et al. 2008). However, in highly urbanized areas such as the District of Columbia (D.C.), this option is often the only one available to managers. In fact, translocation has become a prevalent management strategy for the conservation of many endangered or threatened species and for species with low mobility when natural areas are being developed. While some translocation efforts have shown to be successful for mammals, many studies have indicated little success as a conservation technique for reptiles and these studies recommend that additional methods are necessary to specifically decrease mortality in translocated turtles (Dodd and Seigel 1991; Hester et al. 2008).

Eastern box turtles have a high site fidelity with a developed knowledge of spatial locations of resources (Rittenhouse et al. 2008). They tend to restrict their movement to a well-established home range. In fact, displaced turtles tend to show irregular movement patterns as a result of homing behavior (Hester et al. 2008). Because of the documented vulnerability of eastern box turtle to urbanization, many studies have been conducted that involve translocation (Gould 1957; Iglay et al. 2007; Hester et al. 2008; Farnsworth and Seigel 2013). Furthermore, because the eastern box turtle is listed as a Species of Greatest Conservation Need (SGCN) in the District's State Wildlife Action Plan (SWAP), efforts to conserve the species will be extra critical and translocation could be an important mitigation strategy to protect the species.

Previous studies documenting the effects of relocation, translocation, or repatriation typically involved an initial release of a marked or radio-telemetered animal with minimal further disturbance of the turtle and no repeated attempts at relocation (Hester et al. 2008; Farnsworth and Seigel 2013). My exhaustive search of the literature did not discover any studies that attempted to analyze the effects of repeated relocations on the same individual. Given the inherent difficulties with long-distance animal translocations and jurisdictional issues, urban wildlife managers are often faced with limited placement choices when considering translocating herpetofauna. Therefore, I evaluated the effects of repeated relocations (back to the same release site) of the same 10 eastern box turtles, over a period of 4 years, on movement path metrics (step lengths, bearings, and size of movement area), and range establishment within a highly urbanized environment. I hypothesized that there would be a negative correlation between the increased number of relocation attempts and calculated movement metrics, more specifically: as the number of relocation attempts of an individual increased, its homing would decrease, and it would begin to settle.

4.2 Data and Methods

4.2.1 Study Site

Both the capture site, previously inhabited by the study animal, and translocation site are located in the northeast portion of D.C. (Figure 4-1). The size of the capture site was approximately 42 acres and consisted of a tidal wetland located along the Anacostia River and a forested wetland located within a hardwood forest. Additionally, the site contained stands of dry and wet evergreen forests and four ephemeral wetlands. Because this parcel of land had been approved for future commercial development, this population of turtles was translocated to a protected site, of similar habitat, within D.C. The translocation site was approximately 1.33km south of the capture site and contained a resident population of eastern box turtles, indicating that the habitat was likely suitable for release of the translocated population. The translocation site's habitat consists of hardwood forest, managed meadows, early succession habitats, edge habitats, forested wetlands, floodplains, and emergent non-tidal wetlands and emergent tidal wetlands with rooted vegetation that remained throughout the year. This release site/park was chosen because it was the largest suitable parcel of land that could likely prevent turtles from roaming outside of the park if they homed.

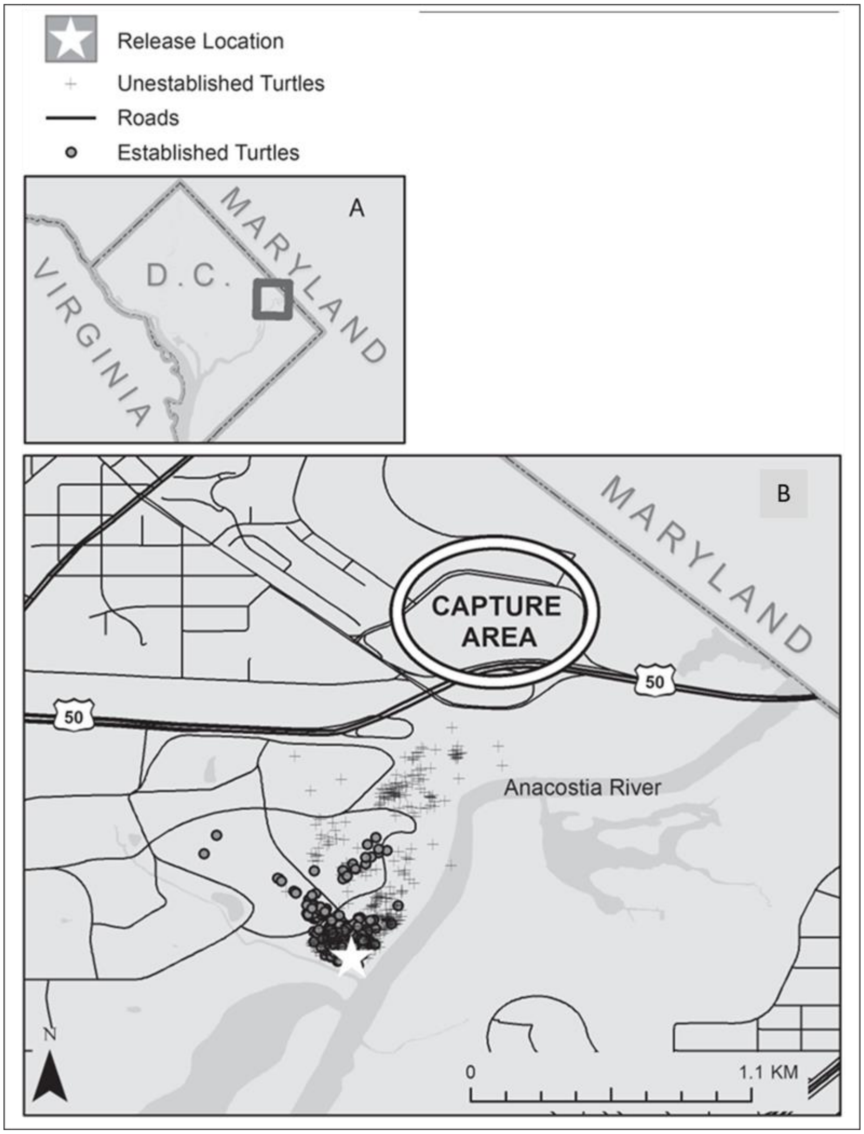

Figure 4-1: (A) Map indicating study area (dark outlined square) within the District of Columbia (D.C.). (B) Detailed map of capture and release sites, with capture area shown within oval outline and the post-release locations of the 4 established turtles (filled dots) and 6 unestablished turtles (crosses), the latter indicating homing behavior toward the capture site. The release point for all translocated turtles is indicated by the white star.

4.2.2 Telemetry Data

I used radio-telemetry data to form the basis of my analysis. From 2011 to 2014. The Department of Energy and Environment (DOEE), Fisheries and Wildlife Division collected telemetry data that included >850 individual locations from 10 individual box turtles that were translocated (three adult females and seven adult males). DOEE biologists fitted the turtles with PD-2 radio transmitters (Holohil Systems Ltd., Woodlawn, ON, Canada) and later R1680 transmitters (Advanced Telemetry Systems, Isanti, MN). They mounted the transmitters to the carapace with PC-7 slow drying epoxy, just above the back leg. Transmitters weighed approximately 3.6–3.8 grams and did not exceed five percent (5%) of turtle body weight which ranged from 145 to 490 grams. In addition to deploying transmitters, DOEE biologists notched each turtle's marginal scutes with a unique identifier in the event of transmitter failure. Ten turtles were radio tracked from 2011 to 2014 for an average of 222.90 days ± 21.93 (SD). Generally, the active season for turtles was from May to November each year. Of the 10 turtles that were translocated and monitored from May 2011 to November 2014, nine turtles had a minimum of 30 tracked locations between relocation attempts and were included in my full statistical analysis. These nine turtles had at least 195 tracked locations across all four years with at least 75% of those locations occurring during the active season.

Relocated turtles were found and released over various times and days but during the same season and at the same release site. Turtle ABC was found and released on 5/2/11; turtle ABH on 4 May 2011; turtle ABI on 13 May 2011; turtle ABJ on 31 May 2011; turtle ABK on 2 June 2011; turtle ABL on 17 June 2011; turtles ABM, ABN, and ABO were all found and released on 19 August 20118; and turtle ABP on 26 August 2011. One turtle was found and released but not tracked. Biologists used radio telemetry to monitor the movements, activities, and other behaviors of the eastern box turtles in the relocation area. The turtles were tracked using TRX-16 Rech receivers and 3-element folding Yagi antenna (Wildlife Materials

Inc., Murphysboro, IL) a minimum of once per week during the active season of May to November, and occasionally during the winter or hibernation months, December to March, as in Haxton and Berrill (2001) and Milam and Melvin (2001). Turtles showing consistent directional movements back to their original capture site or unusual behavior were tracked more frequently, up to daily tracking (Cook 2004) and if the turtle reached the perimeter fence, approximately 1000m due north of the release site location, indicating homing, it was located and moved back to the original release point. A global positioning system (GPS) was used to mark the retrieval and release locations of the turtles which I then mapped using ArcGIS, version 10.1, (Environmental Systems Research Institute, Inc., Redlands, CA).

Because of the disparate number of locations per individual and the relatively small number of turtles monitored, I created standardized rules for the inclusion of locational data. Specifically, locations were retained for use if an individual turtle had a minimum of 30 data points per relocation attempt. Then, only active season locational data were used (May – November), assuming at least 75% of locations were obtained during those active months. I classified the retained individuals with radio tags into two groups: 1) Established (turtles that required no further relocation after the initial release; four individuals) and 2) Unestablished (turtles that required repeated relocation back to the release site; six individuals). There were no notable differences in the size, age, and sex of the two groups. Additionally, I divided all ten turtles into six sub-groups based on the number of relocations required before the turtle exhibited a settling behavior (Table 4-1). I formulated these sub-groups by subtracting the relocation attempt number from the total number of relocations of that individual. This was done to better illustrate the correlation between relocation attempts and calculated movement metrics.

Table 4-1: Description of how the nine eastern box turtles (Established and Unestablished) were sub-divided for analysis based on the number of relocation attempts, where Group = (R – r). R = Total number of relocations for that individual, r = relocation attempt number for that individual, and n = the number of turtles within that analysis group.

Group (R-r)	n	Description
0	9	Initial translocation (settled)
1	5	1 relocation away from settling
2	3	2 relocations away from settling
3	2	3 relocations away from settling
4	1	4 relocations away from settling
5	1	5 relocations away from settling

4.2.3 Home Range Evaluation and Movement Metrics

To establish a baseline range size for the established group, I used Geospatial Modelling Environment (GME), with the open-source software R as the statistical engine (Beyer 2012), to calculate 50% and 95% fixed kernel home ranges (Worton 1989), with a Least-squares cross-validation (LSCV) smoothing parameter (Figure 4-2). This was done because using individual radiolocations as the sampling units (Type III habitat selection, Johnson 1980) may be biased because of unequal sample sizes, telemetry error, and temporal biases in the telemetry data (Kauhala and Tiilikainen 2002). Also, individual telemetry locations are often correlated, which can produce biases, especially at low samples sizes. However, such biases have little influence on fixed kernel density home range estimates given adequate sample sizes (Moser and Garton 2007). I calculated the area of the established turtles' 50% and 95% kernel density estimates across four years to determine if there were any significant changes in range size.

Previous research has shown that translocated turtles travel greater total distances, and make more directional, linear movements when compared to resident turtles (Rittenhouse et al. 2007). Therefore, using additional analyses within GME, I calculated movement metrics for the period between each relocation for all turtles, including step lengths and bearings and calculated movement areas by generating minimum convex polygons (MCP) around the individual's telemetry locations. I then identified the geographic center (or center of concentration) of each turtle's movement area, using the spatial statistics toolbox in ArcMap 10.1, and determined the shortest straight-line distance from the release point. These measures allowed me to assess the direction and relative magnitude of the turtles' movements during individual relocation attempts, as well as compare the size of overall movement area between attempts.

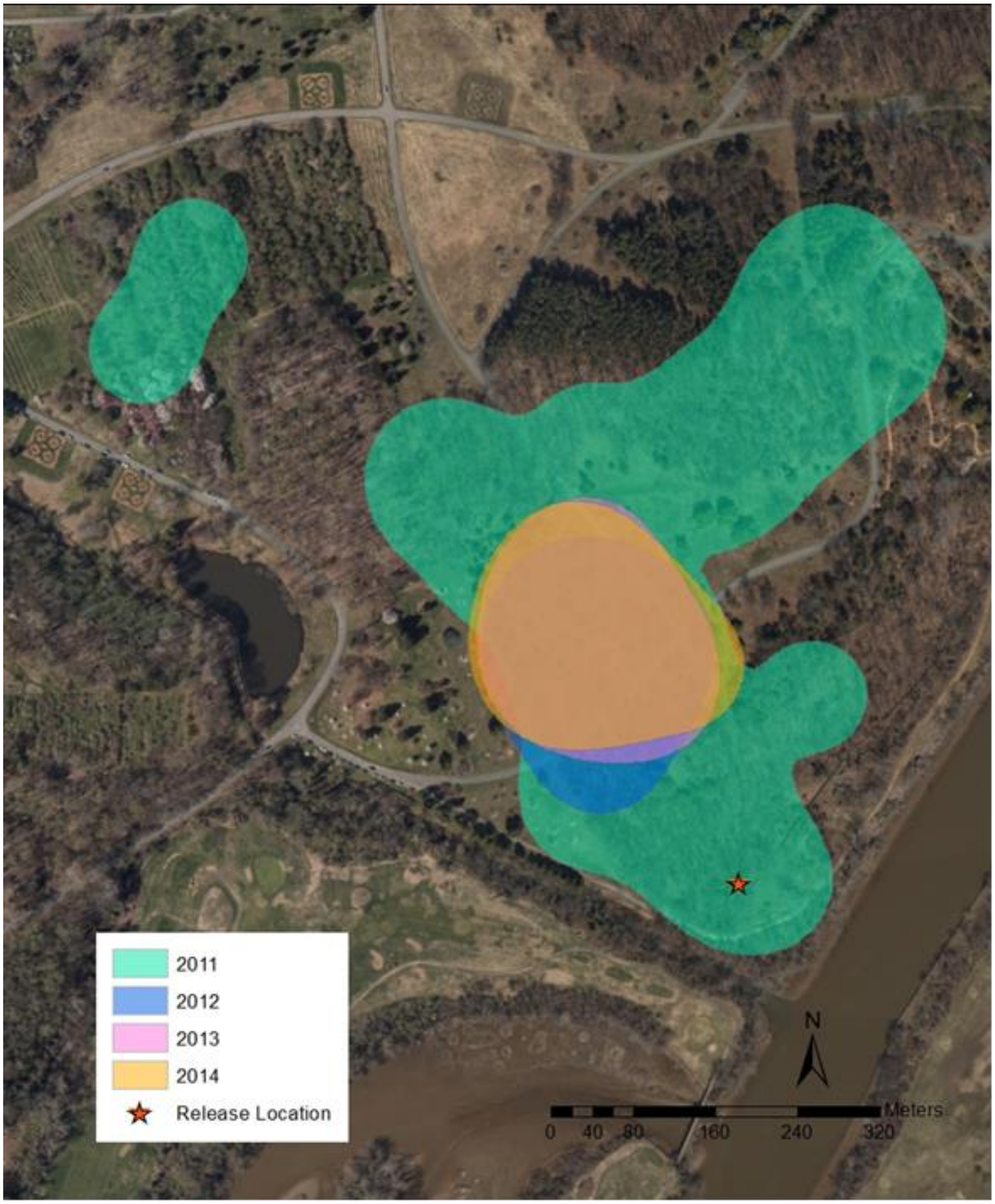

Figure 4-2: Example of 95% fixed kernel density home ranges calculated for each year for turtle ABH; one of the four established eastern box turtles that was monitored from May 2011 to November 2014, at the relocation site, in Washington D.C., USA.

To first determine whether significant differences existed in the movement patterns of established versus unestablished turtles, I used individual Mann-Whitney-Wilcoxon tests, to compare turtles' bearing (the direction between location points), distance from release point to the mean center (geographic center) of the movement area, and step-lengths (the distance between location points). This test was used under the assumption that the two groups are independent of each other. To further determine whether these characteristics were different across the six relocation sub-groups, I used the Kruskal–Wallis one-way analysis of variance (ANOVA) on ranks (McKight and Najab 2010). Lastly, I used the random forest regression method (Breiman 2001; Liaw and Weiner 2002) to measure the relative importance of each of my calculated variables (step lengths, bearings, and movement area size) to relocation efforts.

4.3 Results

I considered four turtles as established: they settled immediately and required no additional relocations back to the release site after the first release on May 2nd, 2011 (Table 4-2). I found that the size of the home ranges decreased over time, based on my analysis with either 50% or 95% kernels (Figure 4-3). I considered six turtles unestablished: as they exhibited homing behavior and required repeated relocations back to the original release site. All six turtles required relocation back to the release site later in 2011, three turtles (ABK, ABL, ABC) required three relocations, two relocations, and one relocation, respectively, back to the release site in 2012, one turtle (ABK) required four relocations in 2013, and two turtles (ABK, ABM) required six relocations and, one relocation, respectively, in 2014 (Figure 4-4). Turtles ABI and ABO required just one additional relocation in 2011 and appeared to settle in 2012. Turtles ABC and ABL appeared to settle after two years of repeated relocations. I removed turtle ABK from the analysis, as she did not fall within the inclusion criteria for further statistical analysis. She continually exhibited homing behavior, requiring a total of 22 repeated relocations across the four study years and there were less than 30 location points between all 22 repeated relocations.

Table 4-2: Description of the 10 radio-tracked eastern box turtles that settled immediately and turtles that required multiple relocation including mean number of days tracked, across all four years of monitoring from May 2011 to November 2014, at the relocation site, in Washington D.C., USA. * Turtle ABK was removed from statistical analysis.

Turtle	Group	Total Number of Days Tracked (2011 -2014)
ABC	Unestablished	246
ABH	Established	244
ABI	Unestablished	238
ABJ	Established	233
ABK*	Unestablished	249
ABL	Unestablished	232
ABM	Unestablished	197
ABN	Established	197
ABO	Unestablished	198
ABP	Established	195
Mean	Established and Unestablished	222.9
SD	Established and Unestablished	21.93

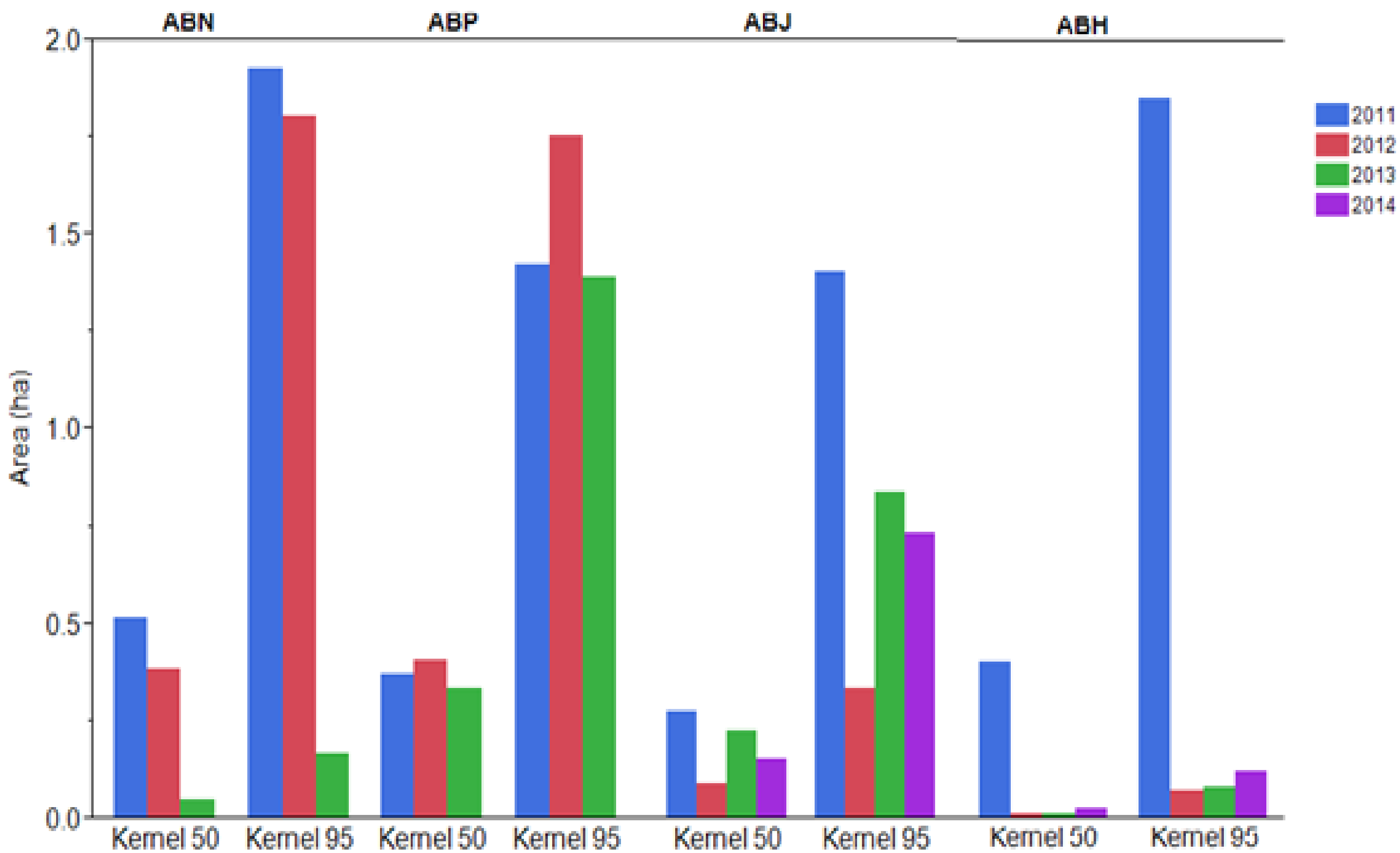

Figure 4-3: Area, in hectares, from the 50% and 95% fixed kernel home ranges, with a least-squares cross-validation smoothing parameter, calculated across all study years for the group of four established eastern box turtles during monitoring from May 2011 to November 2014, at the relocation site, in Washington D.C., USA.

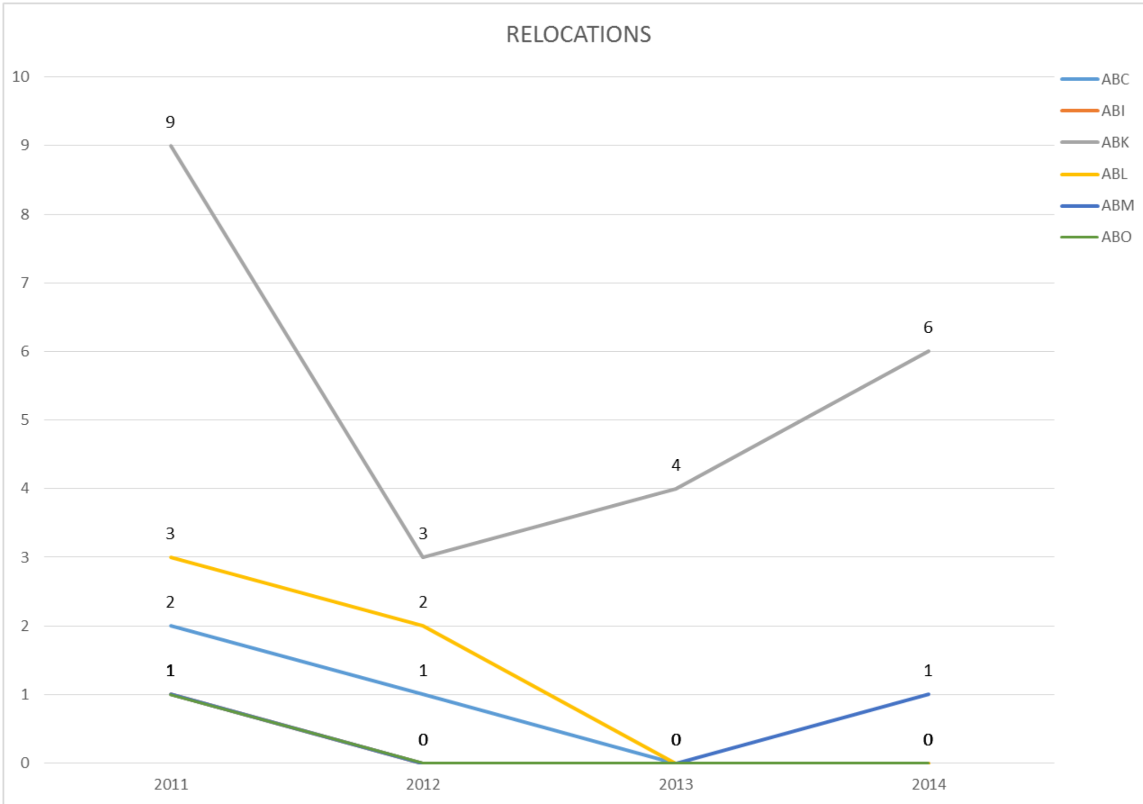

Figure 4-4: Number of repeated relocations across each of the study years for the unestablished group of eastern box turtles that required repeated relocation (n=6) monitored from May 2011 to November 2014, at the U.S Relocation site, in Washington D.C., USA.

Results of the Wilcoxon test indicated a significant difference in distance moved from release point to mean center of movement area between the established group (n=4) and the unestablished groups (n=5) (W = 41,660, $P < 0.001$); the unestablished group travelled a median distance from their release point that was 53% greater (156m) than those that settled (Table 4-3). I found that the unestablished turtles also had a median step length that was significantly greater than the established group (W= 59,070, P = 0.006). I did not find any significant difference in bearing between the two groups (W = 69,355, P = 0.864) (Figure 4-5).

Table 4-3: Median distance, step lengths, and bearing and Interquartile Range (IQR) for established and unestablished eastern box turtles monitored from May 2011 to November 2014, at the relocation site, in Washington D.C., USA. Significant differences between the 2 groups are indicated with an asterisk (*). The interquartile range is a measure of spread when dealing with skewed and/or data with outliers.

Group	**Distance to mean Center of Movement Area (m)**		**Step-lengths (m)**		**Bearing from original capture location (degrees)**	
	Median	**IQR**	**Median**	**IQR**	**Median**	**IQR**
Established (n = 4)	135.25*	93.86 to 319.90	27.17*	13.42 to 50.15	198.66	110.77 to 286.51
Unestablished (n = 5)	291.67	136.89 to 479.87	31.79	15.72 to 68.48	199.78	80.57 to 280.30

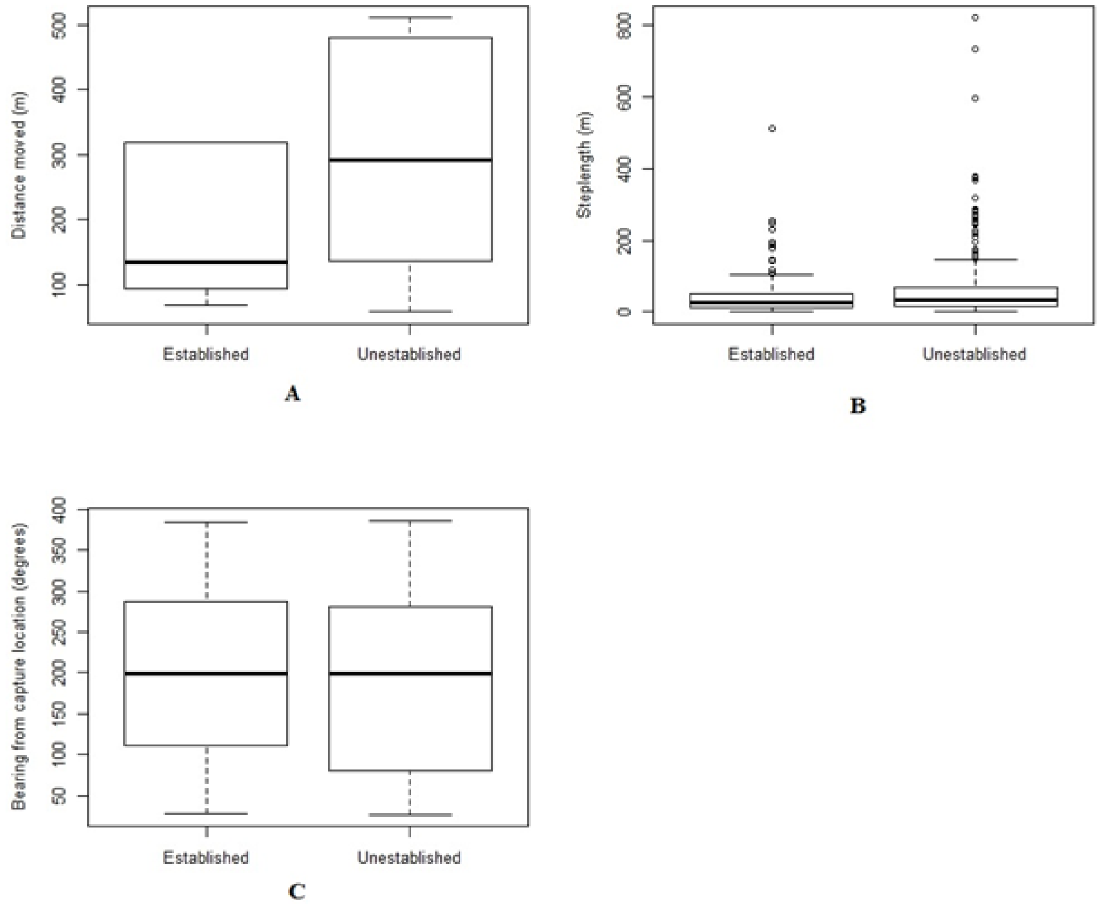

Figure 4-5: Box and whisker plots comparing the distance from release point to mean center of movement area (A), step lengths (B), and bearing (C) for the four eastern box turtles that required no repeated relocation (established) and the six turtles that required more than the initial relocation (unestablished) during monitoring from May 2011 to November 2014, at the relocation site, in Washington D.C., USA. The box indicates the shape of the data distribution, the bold center line represents the central value (median), and the whiskers represent variability in the data.

Results of the Kruskal–Wallis one-way analysis of variance indicated the size of movement area differed significantly between two or more of the repeated relocated sub-groups (H=38.37, df =5, P <0.001) (Table 4-4). Additionally, I found a significant difference in step-lengths (H=38.71, df=5, P <0.001). Box plot results further illustrate a positive relationship between the number of relocation attempts and size of movement area, as well as step-lengths (Figure 4-6).

Table 4-4: Test statistic for the Kruskal Wallis test (H) for 3 observations across the six relocation sub- groups. Because there are six comparison groups and three observations in each of the comparison groups, it can be shown that the test statistic H approximates a chi-square distribution with df=k-1, where k=the number of comparison groups, we found the critical value in the table of critical values for the chi-square distribution for df = 5 and α=0.05.

Observations	**Critical Value for**	**H**	**df**	**P**
Distance	11.07	38.37	5	< 0.001
Step-Length	11.07	38.71	5	< 0.001
Bearing	11.07	3.6685	5	0.598

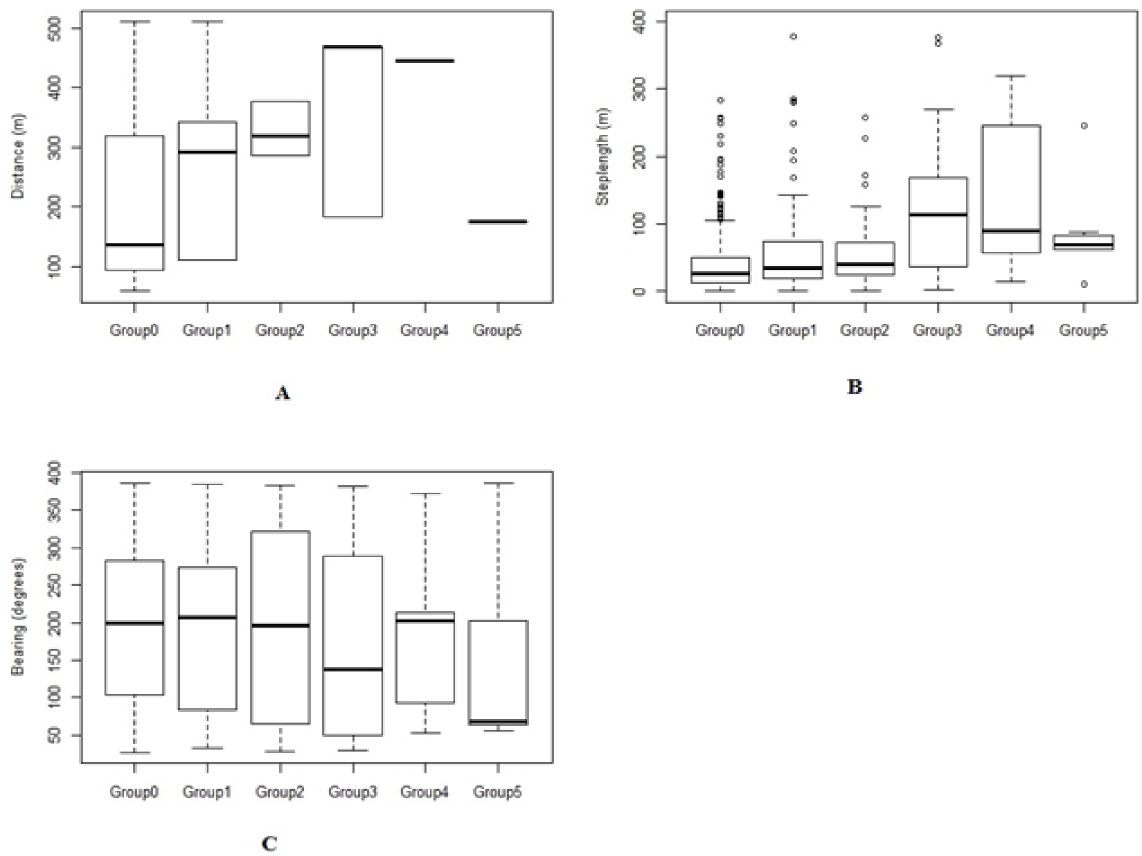

Figure 4-6: Box and whisker plots comparing the distance from release point to mean center of movement area (A), step lengths (B), and bearing from capture location (C) across the six sub-groups of eastern box turtles during monitoring from May 2011 to November 2014, at the relocation site, in Washington D.C., USA. The box indicates the shape of the data distribution, the bold center line represents the central value (median), and the whiskers represent variability in the data.

Results from the random forest regression analysis showed the most important variable for predicting relocation attempts was distance from release point (mean decrease in accuracy = 71.27, node impurity/Gini index = 197.20), whereas bearing was shown to have the least importance (mean decrease in accuracy = 3.60, node impurity/Gini score = 7.78) (Figure 4-7). Mean decrease in accuracy indicates how much the model accuracy decreases if I remove that variable and the mean decrease gini shows the measure of variable importance based on the Gini impurity index used for the calculation of splits in trees.

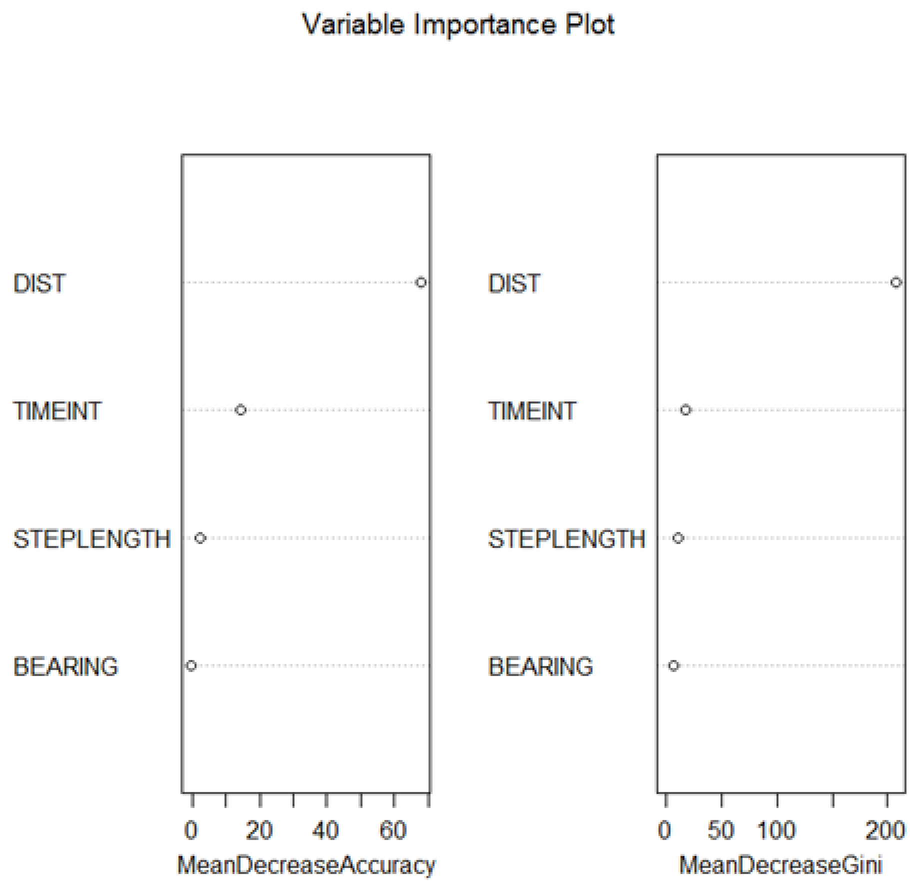

Figure 4-7: Variable importance plots for predictor variables from random forests (RF) classifications used for predicting relocation attempts of eastern box turtles during monitoring from May 2011 to November 2014, at the relocation site, in Washington D.C., USA.

4.4 Discussion

Wild animals are increasingly encountering people as cities continue to sprawl into undeveloped regions. Urban, suburban, and exurban growth increases edge habitat, and human and wildlife welfare depend on an understanding of urban wildlife and effective management options for them. The concept of translocating a free-ranging animal or population of animals away from an area where they are at risk to an area that is currently and/or historically occupied by that species, has raised many questions regarding effectiveness, especially for reptiles (Dodd and Seigel 1991, Bowen et al. 2004, Hester et al. 2008). My research resulted from efforts to mitigate human-wildlife conflicts by preserving an intact wildlife population when human development of eastern box turtle habitat was inevitable. In this instance, there was no option of translocating turtles outside of D.C. and there were limited potential translocation sites within the district. Furthermore, very little was known of existing eastern box turtle populations at the relocation site. Managers were left with few options aside from translocation and radio telemetry along with a willingness to attempt to modify translocated turtle homing behavior through assisted, repeated relocation.

Understanding the time required for an eastern box turtle to settle when displaced to a new range, aids wildlife managers in planning the time, effort, and resources required for mitigation where translocating is the best or only alternative. These findings are particularly significant in highly urbanized areas such as the D.C., where large parcels of land with suitable habitat are minimal. My observation that the unestablished turtles may have needed fewer repositions over time supports the findings of Farnsworth and Seigel (2013), who found that turtles in Maryland overwintered in their release locations but were repositioned an average of 2.58 times across 4 years. These authors suggest the use of physical barriers to prevent turtles from homing across dangerous areas such as construction sites (Farnsworth and Seigel 2013). In a more rural area, Seibert and Belzer (2013) found that turtles needed at least 10 years to settle into new home ranges, and some never settled even when repositioning was used to help guide. This study suggests that if eastern box turtles must be translocated within an urbanized area, management agencies should be committed to tracking and repeatedly relocating turtles for a minimum of three years to obtain >50% retention of turtles in the general settlement area. Without meeting the minimum conditions of three years, efforts may prove unsuccessful. Furthermore, this study suggests that analysis of movement areas of relocated turtles may increase the efficiency of tracking efforts as managers may be able to identify individuals that require more intensive tracking and more relocations. My results support consistent repeated relocation as a viable conservation strategy for eastern box turtles that require relocation in a limited urban or suburban environment. Nevertheless, it is important to note that additional studies with larger sample sizes and data on the existing population at the relocation site would be necessary to determine the long-term effects of this novel management strategy.

Chapter 5: Conclusions

5.1 Summary of Contributions

This research has demonstrated the substantial value of leveraging new and traditional species-related geospatial datasets to advance urban planning and policy goals and to inform management strategies that support biodiversity conservation. The results show that by changing biodiversity's participatory function in urban planning from a passive role to an active role, city planners and policymakers can more holistically include the needs of both people and the environment in strategic planning (Hernandez-Santin, Bekessy and Desha 2023). While urban comprehensive plans typically have provisions for the preservation of large tracts of open space, greenways, and parks, they have not consistently included the need for connections between those larger habitat patches (Hardy et al. 2022), which are important for wildlife movement and overall species fitness. Quantifying and mapping the biophysical value of connecting greenspaces could support strategic habitat conservation initiatives.

Chapters 2 and 3 leveraged traditional site-specific species survey data and new high-resolution geospatial datasets to identify and map target restoration and conservation opportunities within a broader biodiversity inclusive corridor design network. The study in Chapter 2, specifically identified local urban conservation and restoration sites that mimic or can be restored to mimic the structural habitat requirements of biodiverse parks in the region. These sites were mapped spatially, to improve habitat connectivity efforts, and were assessed for prioritization based on the potential risk of future gray infrastructure development. This analysis demonstrated that placing biodiversity at the forefront of urban planning can support an array of benefits, from conservation to community restoration to climate resilience.

Chapter 3 directly addressed complex issues in urban land use and land management decisions in Washington D.C., including the availability of critical resources and the allocation of these resources among competing social, cultural, and environmental needs (Aronson et al 2017; Shwartz 2014). In this chapter, the biophysical value of a variety of urban ecosystem services were attributed to viable wildlife corridors to identify locations with the highest benefits per unit cost for conservation and restoration. By making the co-benefits of ecosystem services explicit, the methodology and results presented in this chapter raised awareness of the importance of biodiversity conservation in urban areas, helped inform biodiversity sensitive urban planning and corridor design, and provided a framework for integrating biodiversity conservation into broader conservation initiatives.

Many local governments are now beginning to see the connections among comprehensive planning, neighborhood development and revitalization, ecological and community health, food policy, and sustainability. Nevertheless, as cities expand and transform to meet the needs of their populations, urban development is unavoidable and urban wildlife populations are negatively impacted by the effects of habitat loss and fragmentation. Therefore, managers are forced to make difficult decisions regularly and many are becoming increasingly reliant on hands-on management strategies.

The study presented in Chapter 4 examined the effects of repeated relocation of eastern box turtles (*Terrapene carolina carolina*) in an urban environment. Since urbanization is expected to continue, it is probable that similar situations will become more frequent across cities in the U.S., and the use of repeated relocation will likely increase as a commonly used, triage-type management tool. Although the study population was small, the results of this case study suggested that home-range establishment may occur in a new location by repositioning individuals post-release. Furthermore, while results were provided for a small population, successful conservation efforts at any scale are necessary for and can contribute to conservation success globally (Sodhi et al. 2011). Chapter 2 and 3 identified several potential translocation sites, such as the US national Arboretum and Anacostia Park.

5.2 Future Work

As urban ecosystems are continually exposed to direct and indirect anthropogenic stressors, increased rates of biodiversity decline are likely (Diaz et al. 2019). Because the extent of this impact is still unclear, more work should be done to include climate projections and socioeconomic processes in urban ecological studies to guide appropriate policy decisions as outlined below. This fact also highlights the significance of using long-term, high-resolution, and standardized information linked to ecosystems' abiotic and biotic components to increase our understanding of urban ecosystem dynamics and to develop suitable conservation and management strategies (McCord et al. 2021). While the research presented across the past three chapters incorporated much of this information, each chapter presents unique opportunities for future work that may include one or more of the following considerations.

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5.2.1 Urban Wildlife Surveys and Biodiversity Metrics

Urban wildlife surveys and species tracking (utilized in Chapters 2 and 4) require careful consideration of resource constraints, limitations of available methods, and potential sources of bias, in order to produce accurate and meaningful results. Similar to D.C., many urban wildlife management programs have restricted resources which can limit the number of species being assessed and impact the geographic scope and quality of urban wildlife surveys. Limited resources often leads to sampling bias, which occurs when the sampling method used to collect data is not representative of the target population. Using multiple survey methods can help overcome bias in any one method. For example, combining visual surveys with acoustic monitoring can provide a more comprehensive understanding of the presence, distribution, and abundance of urban wildlife. Some studies have indicated that the most effective and economical way for identifying reptile and mammal species in remnants of urban forests is the combination of at least two complimentary techniques (Garden et al. 2007). The species presence data utilized in chapter 2 were derived from a variety of inventory techniques (including a small number of remote cameras and acoustic surveys), however, significant improvements in sensing technologies and their rising accessibility have greatly increased capacity for data gathering and are fundamentally altering how ecological data is being sampled (Allen et al. 2018). By increasing the use of remote cameras and acoustic surveys and/or incorporating additional technologies such as drone surveys (all of which autonomously gathering high-resolution sounds and images), human sampling can be improved, surpassed, or even replaced (De Bondi et al. 2010; Wellbourne et al. 2015;

Darrus et al. 2018). Moreover, employing such technologies can help reduce the cost of traditional survey techniques, increase efficiency, and provide more accurate survey results. Additionally, while the biodiversity metrics were calculated from long-term, multi-taxon wildlife surveys that spanned 10 years, rare species can often still go undetected in long term monitoring surveys, especially in diverse communities (Gotelli and Colwell 2001; Longino, Coddington, & Colwell, 2002). This could preclude effective biodiversity conservation policies. Therefore, additional assessments of richness and diversity using multi-species occupancy models (MSOM), which account for species-specific variation in detection, could provide superior estimates that also account for uncertainty more accurately (Tingley et al. 2020).

5.2.2 Climate Considerations

Ongoing climate change is altering the characteristics of urban forests and greenspaces, which is likely to have significant implications for the ecological functioning and biodiversity of urban ecosystems (Dallimer et al 2016; Pecl et al. 2017; Pretzsch et al. 2017; Esperon-Rodriguez et al. 2022). Specifically, a warming climate and the associated ecosystem changes is expected to alter wildlife species movement and distribution patterns (Lawler et al. 2006; Lister et al. 2015). Furthermore, the barriers to movement caused by gray infrastructure development will likely restrict the ability of species to keep up with these changes. The corridors identified in Chapter 2 and utilized in Chapter 3 were mapped based on current forest distribution, and while conservation and restoration areas were prioritized based on future development risk, additional climate modeling could help target more climate adaptive species and anticipate how corridors may need to adjust as species' behaviors, movement, and migration patterns change.

The urban ecosystem services assessment in Chapter 3 was based on current knowledge and capacity of existing ecosystems, current range of variability, and average contemporary conditions, however, climate change will have an impact on the potential services these systems can provide in the future (Pecl et al. 2017). As a result, future iterations of this work should include additional climate modeling which may be helpful to understand a range of potential conditions and inform strategies for adaptative management to maximize benefits over the long term. It is important to note that while some aspects of urban ecosystems can be integrated into climate models, urban-scale modeling often requires a more specialized, detailed approach beyond the scope of broad global or regional climate models. Urban environments create their own microclimates, characterized by variations in temperature, wind, and air pollution at a fine scale. High-resolution climate models can be used to simulate these microclimates, which are essential for understanding the implications for human comfort, health, and energy consumption. Cities also often have complex water management systems that include stormwater management, wastewater treatment, and water supply. These models help to assess the effects of climate change on urban water resources and to develop strategies for sustainable urban water management under changing climatic conditions. Additionally, models that estimate the carbon footprint of cities and assess the effectiveness of mitigation strategies can be linked to broader climate models to evaluate the collective impact of urban centers on global climate change. Lastly, as cities face increasing risks from climate change impacts, models that assess urban vulnerability, resilience, and adaptation strategies are crucial for planning and policy development to mitigate the effects of climate change on urban populations and infrastructure.

The case study in Chapter 4 addressed direct habitat loss for a species of greatest conservation need and supported mitigation/intervention strategies. Repeated relocation of a species requires a great deal of resources, and even if the initial impetus for a species' relocation is due to a direct loss of habitat, it is essential to consider the permanence of the new habitat under a changing environment to ensure the persistence of the translocated population. Climate change data is essential for monitoring the success of habitat relocation efforts over time. Continuous monitoring of temperature, precipitation, and other climatic factors in both the original and relocated habitats helps assess the effectiveness of the relocation and whether additional interventions are needed. Furthermore, climate change data can help identify species that are particularly vulnerable to climate-related impacts in their current habitats. By analyzing projected changes in temperature and precipitation, conservationists and urban planners can determine which species are at risk of local extinction and may require habitat relocation to more suitable areas. Lastly, urban areas can employ climate mitigation strategies, such as urban greening, green roofs, and cool pavement, to create more favorable microclimates for species that may struggle to cope with rising temperatures. Such measures can supplement habitat relocation efforts and increase the chances of species survival in the urban environment.

5.2.3 Socioeconomic Considerations

Several studies have indicated that the patterns of biodiversity within a city are largely driven by socio-economic characteristics (Kinzig et al. 2005; Uchida et al. 202), underscoring the importance of explicitly integrating socioeconomic processes in ecological studies of human-dominated systems.

5.2.3.1 Economic valuation

The ecosystem assessment in Chapter 3 quantified and mapped the current biophysical value of the land within the proposed biodiversity corridors. By also making the economic value of these systems explicit, local policymakers can better communicate the broader societal benefits of conservation efforts and engage a wider range of stakeholders in conservation. Additionally, this information can help guide conservation efforts and ensure that limited resources are used in the most effective way possible. For example, by monitoring changes in the economic value of the proposed biodiversity corridors over time, it is possible for policymakers, landowners, and conservation organizations to assess the effectiveness of conservation efforts and adjust as needed. This can help ensure that incentive payments and programs are achieving their intended goals. Moreover, understanding the economic value of urban ecosystems is important because the economic impacts of land-use change has direct costly impacts on species, given the cost of mitigation strategies for vulnerable species once their habitats are lost.

5.2.3.2 Equity Considerations

The corridors identified in Chapter 2 were based on value to biodiversity and the conservation and restoration of the economically valuable lots within the corridors will require buy-in and support of local communities. However, conservation efforts have a long track record of not engaging local communities in the design of conservation and restoration projects (Porter, Hurst, and Grandinetti, 2020; Taylor et al. 2022). Incorporating community perspectives, including the perspectives of marginalized and underrepresented communities, can help ensure that future research and recommended management strategies are culturally appropriate and responsive to the needs and interests of local communities. This can involve conducting community surveys or focus groups and incorporating community feedback into management plans.

While the ecosystem service assessment in Chapter 3, implicitly targeted human benefit and accounted for the disparities in distribution and supply, future iterations of this work can explicitly incorporate the socioeconomic factors that may influence access to natural resources and exposure to environmental risks. This can include demographic data on income, race, ethnicity, and education level. For instance, overlaying the highest impact corridors with the new federal Climate and Economic Justice Screening Tool (CEJST) (Climate and Economic Justice Screening Tool 2022) would draw attention to areas that have historically been underserved and overburdened and may currently have a lower relative supply of ecosystem services, but would benefit most from investments in green infrastructure. Additionally, the proposed biodiversity corridors could also be overlayed with the recently released ArcGIS Historical Redlining spatial data layer (found on the ArcGIS Living Atlas) to identify where conservation and restoration priorities would best serve historically disenfranchised communities. Specifically, the historical redlining spatial data layer can be used to identify areas where historically disenfranchised communities were subjected to discriminatory policies that denied them access to resources and opportunities. These same communities are often disproportionately impacted by environmental pollution and land degradation. Once these areas of environmental injustice have been identified, conservation and restoration efforts can be prioritized based on the level of need and potential impact. For example, areas with high levels of pollution and low access to green spaces could be targeted for tree planting initiatives and the creation of urban green spaces, while areas with degraded wetlands could be prioritized for restoration.

Appendix A: Supplementary Material for Chapter 2

A.1 Taxon Specific Biodiversity Metrics

A.1.1 Birds

The highest values of bird diversity (q1 and q2) across my surveyed parks were found on Heritage Island, where 132 different species were detected (Table A-1). The Shannon index (q1) was 108.9 and the inverse Simpson index (q2) was 86.7. Anacostia – section G, USNA, Kingman Island, and Heritage Island differ significantly from the other parks in terms of species richness (Figure A-1). However, I did not see significant differences in species richness between Heritage Island and Kingman Island. My inferences were made at a significant level of $\alpha = 0.05$ (95% confidence interval). Relative to Shannon diversity (q1), the park biodiversity ordering did not correspond to the species richness, and Heritage and Kingman Islands appeared as the most diverse in species followed by Anacostia – Section G.

The highest sample completeness was detected for Anacostia– Section G and USNA because the q0 curve, after an initial phase of sampling effort, becomes almost horizontal indicating the number of detected species does not increase (Figure A-1; Figure A-4A). At the other extreme, Anacostia Section C exhibits sustained increase, meaning that new species are detected with additional survey sites. The extrapolation curve was modeled for 116 survey sites (double the reference sample) and its increasing character shows that with continued sampling the number of detected species is expected to grow.

Table A-1: Number of survey sites (n) and Hill numbers of species richness (q0), Shannon diversity (q1), and Simpson diversity (q2) for each taxon for the 27 surveyed parks in the District of Columbia.

	BIRDS				MAMMALS				HERPS			
	(n)	**q0**	**q1**	q2	(n)	q0	q1	q2	(n)	q0	q1	q2
Anacostia.Park -F	2	32	32	32	3	6	6	5	3	6	5	5
Anacostia.Park -G	58	**179**	109	87	6	9	8	7	25	**21**	**18**	**17**
Anacostia.Park -C	4	51	46	41	4	**11**	**10**	**10**	5	9	9	9
Anacostia.Park -D					2	5	5	5	1	4	4	4
Archbold.Parkway	6	32	26	23								
Barnard.Hill	2	9	9	9	2	4	4	4				
Battery.Ricketts	2	8	8	8								
C & O Canal									2	6	6	6
Dumbarton.Oaks.Park	9	87	71	63	2	6	6	6				
East.Potomac.Park	2	8	8	8								
Fort.Chaplin.Park	7	26	21	19	2	4	4	4				
Fort.Dupont.Park	20	74	51	42	4	6	6	5	4	12	12	12
Fort.Mahan.Park	10	45	34	28	2	4	4	4				
Fort.Stanton.Park	3	23	22	20								
Fort.Totten.Park					2	7	7	7	1	2	2	2
Glover.Parkway	7	40	33	29	3	6	6	6	2	2	2	2
HERITAGE.ISLAND	10	132	**115**	**106**	3	2	2	2	3	5	5	4
KINGMAN.ISLAND	13	133	110	98					3	10	9	8
Oxon.Run.Parkway	3	32	30	29	2	3	3	3	1	3	3	3
Park (National Zoo)	3	19	18	18								
Potomac.Palisades.Prkwy	6	73	63	56	4	6	6	6	34	20	16	14
Rock.Creek.Park	22	79	53	42	5	10	9	9	15	17	13	10
Shepherd.Parkway	3	20	19	18								
Theodore.Roosevelt.Island	16	77	50	39	3	4	4	4	2	4	4	4
USNA	44	131	88	72	7	**11**	9	7	19	21	15	13
West.Potomac.Park.	2	32	32	32								
Whitehaven.Parkway	4	35	31	29	2	4	4	4				

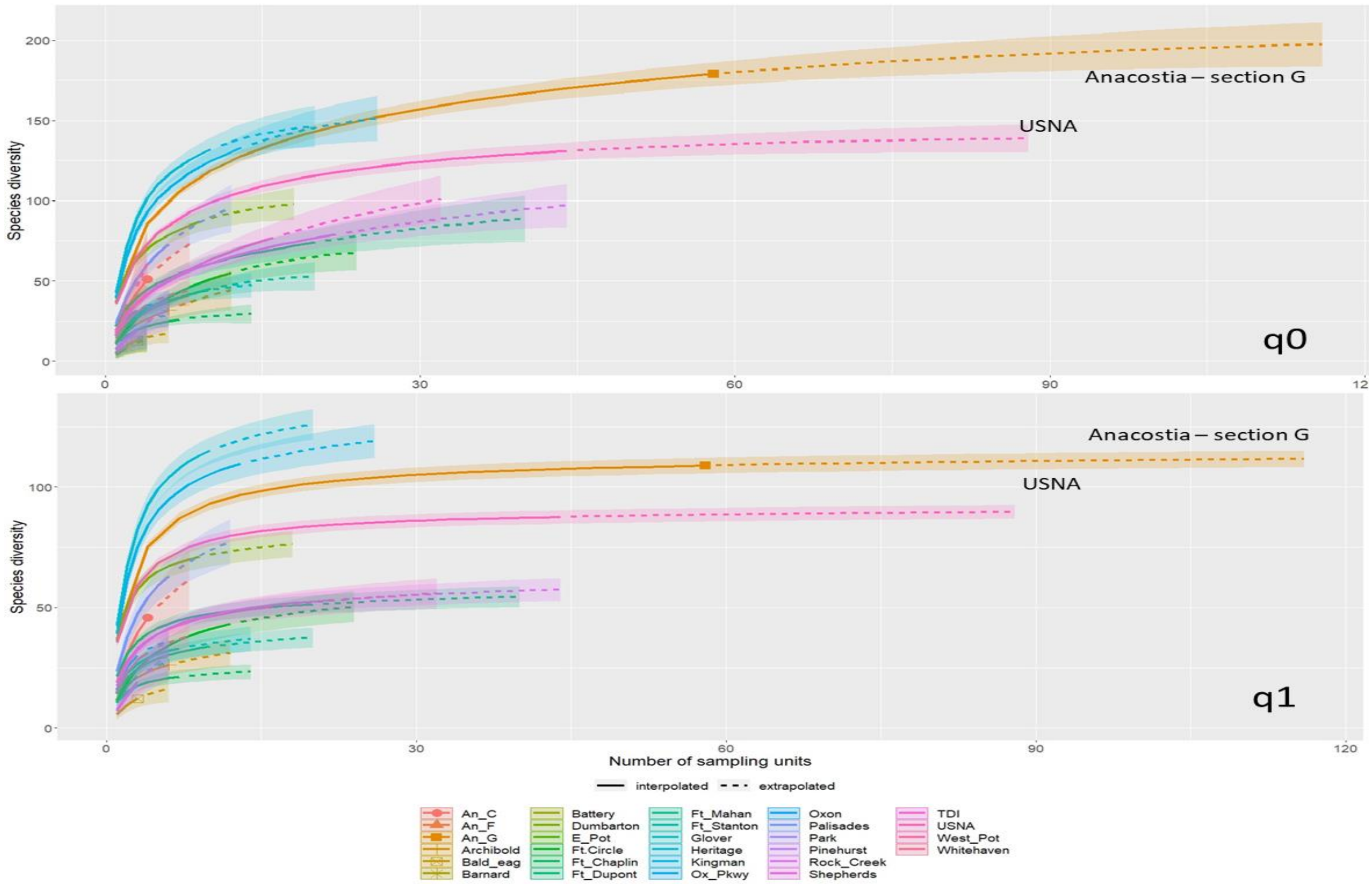

Figure A-1: Integrated curves of rarefaction/extrapolation for q0 (species richness) and q1 (Shannon index), presented as a function of the sample size for birds. Extrapolation goes up to double the size of the reference sample; n = 116 for the birds, n = 34 for the mammals, and n = 38 for reptiles and amphibians. The shaded area represents 95% confidence intervals obtained using the bootstrap method based on 100 repetitions.

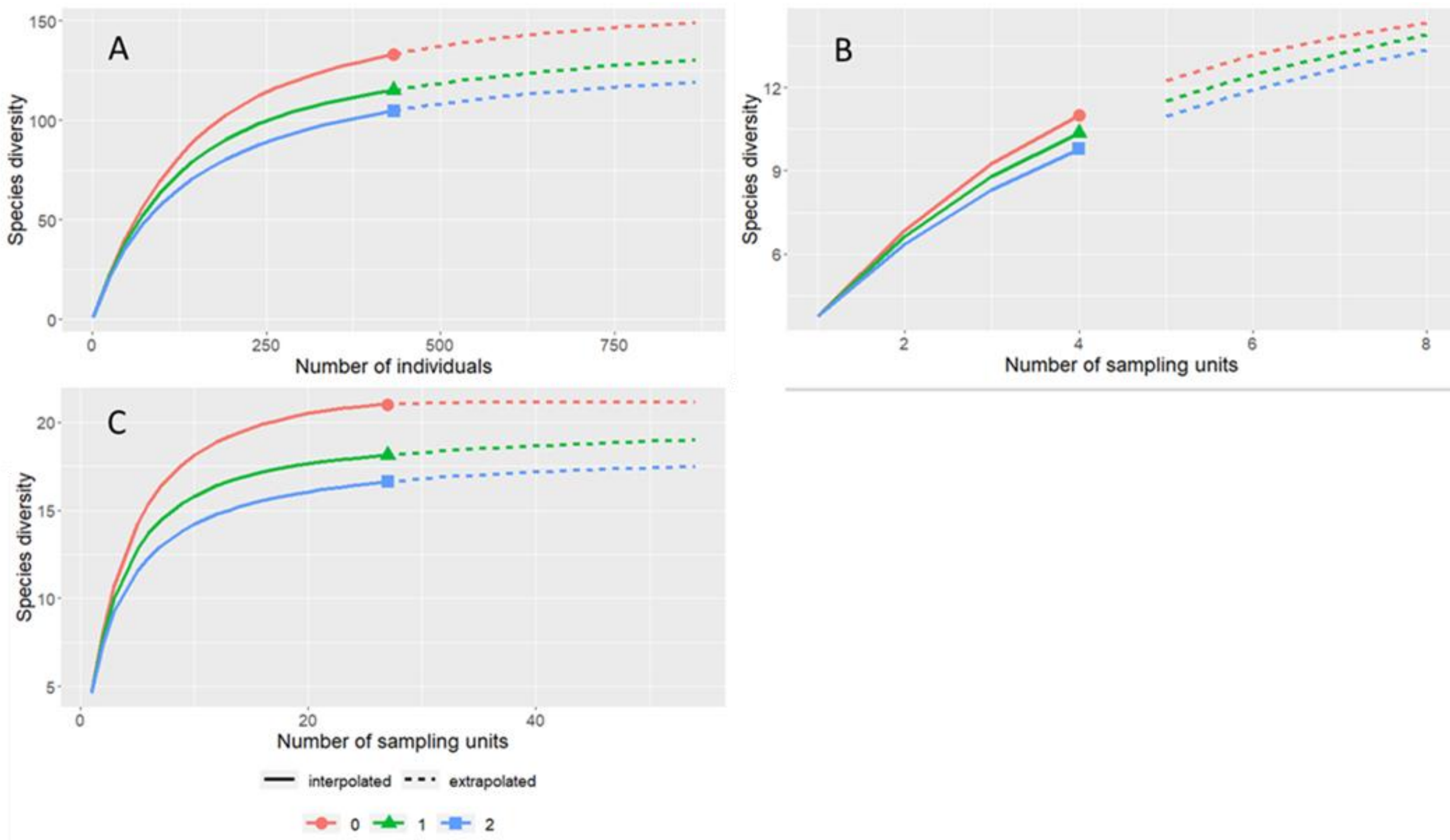

Figure A-2: Rarefaction/extrapolation biodiversity curves for parks with the highest species diversity for each taxon: Birds; Heritage Island (A), Mammals; Anacostia Park- Section C (B), Reptiles/Amphibians; and Anacostia Park – Section G (C) - based on Hill's numbers with q0 (species richness – red line), q1 (Shannon index- green line), and q2 (Inverse Simpson index- blue line). The solid line is the rarefaction curve, and the dotted line is the extrapolation curve, which goes up to double the reference sample size (mammals and reptiles/amphibians) or number of individuals (birds and SGCN species)

A.1.2 Mammals

Anacostia Section C had the highest value of diversity for mammals, where 11 different species were detected (Table A-1). The Shannon index and the inverse Simpson index (q2) were 10.4 and 9.8, respectively. The biodiversity comparison across parks for the mammals is based on 14 sites. The species richness curves (q0) intersect each other, therefore, the order of parks in terms of their biodiversity depends on the size of the sample to be analyzed (the number of surveyed sites). There is significant intersection of curves for both the q0 and q1 estimates, indicating that the representation of biodiversity across the parks is dependent on the amount of survey sites and the overlap of confidence intervals indicate that there is insufficient data in many of the parks to provide reliable estimators and associated standard errors (Figure A-2). Both the species richness curve and the Shannon index for Anacostia Park – Section C grew significantly along with an increasing sampling effort and still indicated lack of completeness at the end of the extrapolation (Figure A-4B). While the curves representing species richness and the Shannon index for the remainder of the surveyed parks grew considerably only initially, their growth was very limited upon extrapolation (the dotted line on the graph), indicating relatively limited growth with the increasing sampling effort and completeness of the sample was achieved at the beginning of the extrapolation.

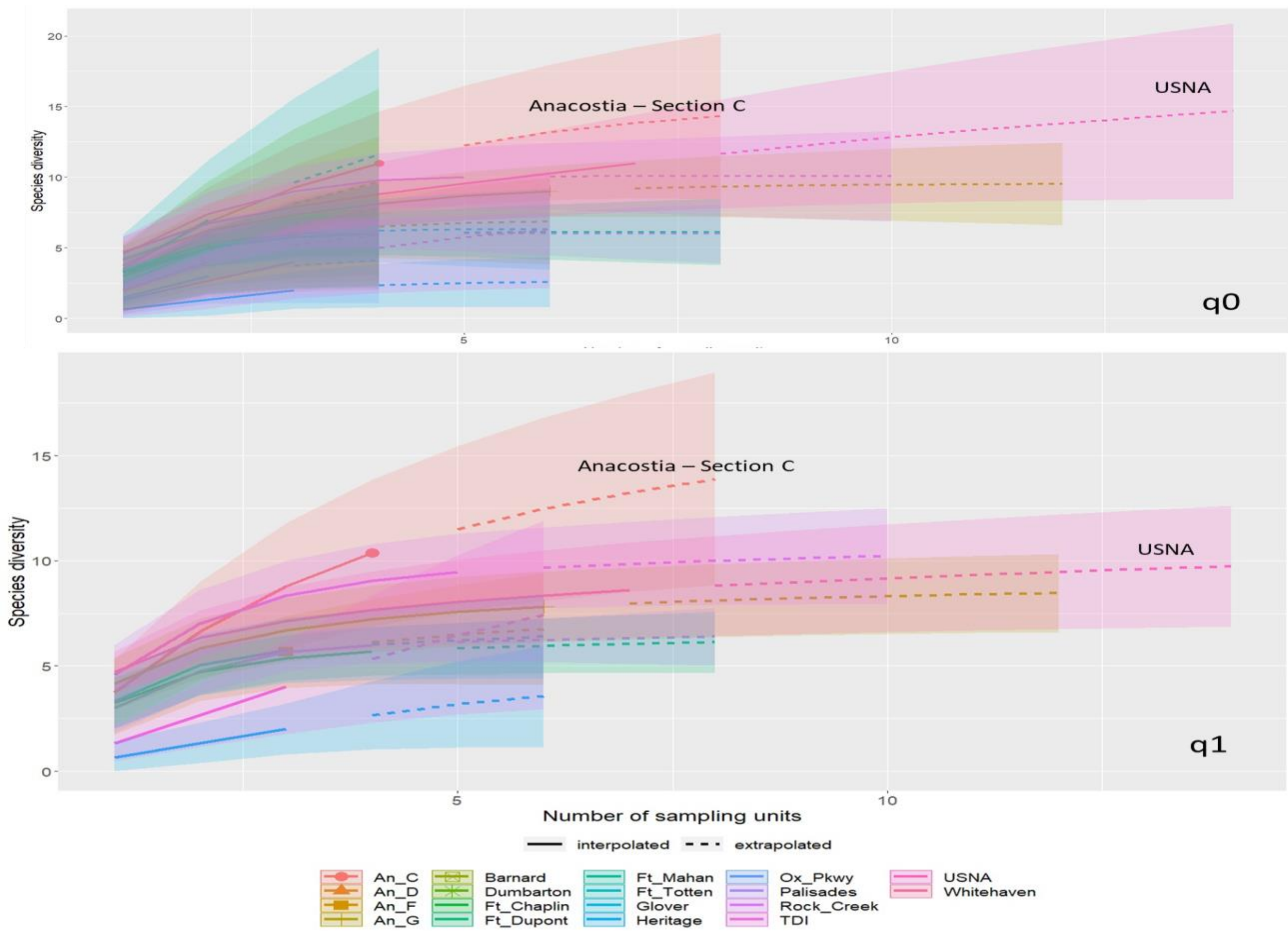

Figure A-3: Integrated curves of rarefaction/extrapolation for q0 (species richness) and q1 (Shannon index), presented as a function of the sample size for mammals. Extrapolation goes up to double the size of the reference sample; n = 116 for the birds, n = 34 for the mammals, and n = 38 for reptiles and amphibians. The shaded area represents 95% confidence intervals obtained using the bootstrap method based on 100 repetitions

A.1.3 Reptiles and Amphibians

Reptile/Amphibian diversity was highest in Anacostia Park – section G (21 species detected) with a q1 of 18.1 and q2 of 16.6 (Table A-1). Bird species richness was highest in Anacostia Park- section G. The comparison of reptile and Amphibian biodiversity across parks is based on 68 sites; double the reference sample size of Potomac Palisades Parkway. There is significant overlap in confidence intervals for the q0 estimate indicating that the representation of biodiversity across the parks is dependent on the amount of survey sites, and in some parks, there is insufficient data to provide reliable estimators and associated standard error (Figure A-3). The q0 curves for Anacostia Park – Sections G, Fort Dupont, and Potomac Palisades Parkway grew considerably initially (Figure A-4C). However, upon reaching extrapolation their growth leveled off, suggesting limited growth with increased sampling effort and completeness of sample. While the curves representing species richness for the remainder of the surveyed parks grew considerably, along with an increasing sampling effort, the lack of completeness at the end of the extrapolation indicated that continued sampling would not result in the detection of more species.

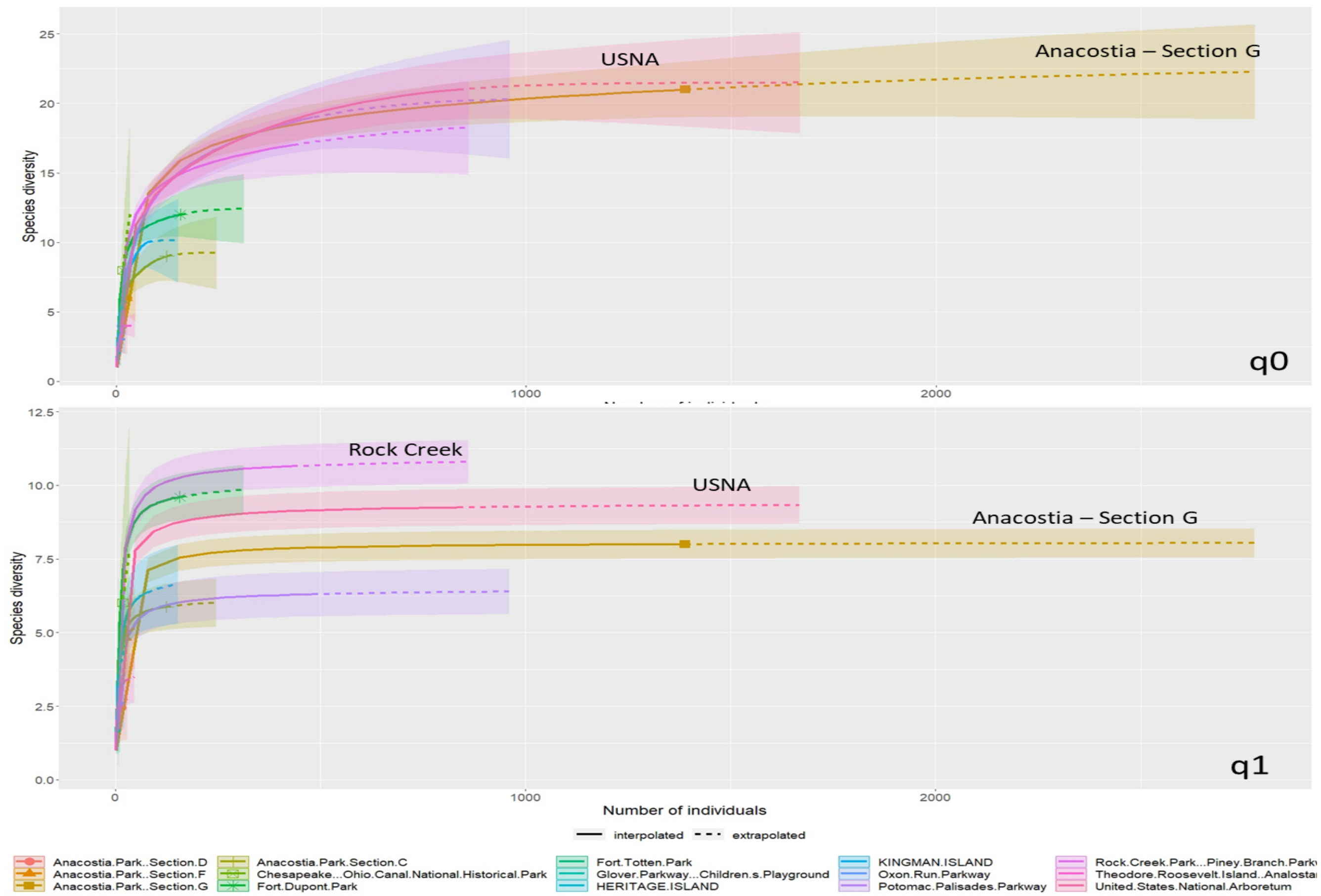

Figure A-4: Integrated curves of rarefaction/extrapolation for q0 (species richness) and q1 (Shannon index), presented as a function of the number of individuals for reptiles/amphibians. Extrapolation goes up to double the size of the reference sample; n = 116 for the birds, n = 34 for the mammals, and n = 38 for reptiles and amphibians. The shaded area represents 95% confidence intervals obtained using the bootstrap method based on 100 repetitions

Appendix B: Supplementary Material for Chapter 3

B.1 Chesapeake Conservancy 2017/18 Land Use/Land Cover Classification – 2022 version

11-15 Water (WATR) = the Chesapeake Bay, lakes and reservoirs, riverine and terrene ponds, large rivers, and water within smaller channels visible through the tree canopy. Included with this class are NWI or state wetlands that are mapped as water in the land cover (MMU = 25m2)

21 Impervious Roads (ROAD) = Paved, and some unpaved, roads and bridges. Dirt and gravel roads may be mistakenly mapped as impervious depending on the spectral characteristics of the substrate (Minimum Mapping Unit (MMU) = 9 square meters).

22 Impervious, Structures (IMPS) = Human-constructed objects made of impervious materials that are greater than approximately 2 meters in height. Houses, malls, and electrical towers are examples of structures (MMU = 9 square meters).

23 Impervious, Other (IMPO) = Human-constructed surfaces through which water cannot penetrate, and that are below approximately 2 meters in height, e.g., sidewalks, parking lots, runways, field-mounted solar panels, rail lines, and some private roads. Barren, low vegetation, scrub-shrub, and emergent wetland cover types within 3 meters of rail lines were reclassed to impervious surfaces and included in this class (MMU = 9 square meters).

24-26 Tree Canopy over Impervious Surfaces (TCIS) = Tree cover that overlaps with roads, structures, or other impervious surfaces rendering them partially or completely invisible from above (MMU = 9 square meters).

27 Tree Canopy over Turf Grass (TCTG) = Tree cover within 30-ft of structures or adjacent turf grass and other impervious in rural wooded areas and within 60-ft of structures or adjacent turf

grass and other impervious in developed areas. Developed areas include U.S. Census Bureau defined urban areas and clusters. Rural areas include all lands outside Census urban areas and clusters. The understory in all TCTG areas is assumed to be turf grass or otherwise altered through compaction, removal of surface organic material, and/or fertilization.

28 Turf Grass (TURF) = Low vegetation associated with residential, commercial, industrial, and recreational areas that is assumed to be altered through compaction, removal of organic material, and/or fertilization. These include low vegetation lands within small, developed parcels (≤ 5 acres with ≥ 55 m2 of impervious cover), recreational fields, and other turf-dominated land uses (e.g., cemeteries, shopping centers, golf courses, airports, hospitals, amusement parks, etc.).

29; 35; 51-53 Pervious Developed, Other (PDEV) = Barren lands in developed parcels and barren or low vegetation lands that may represent the early stages of development, utility rights of-way, portions of road rights-of-way, landfills, and the pervious portions of solar fields adjacent to panel arrays. 32 Harvested Forest (HARF) = Barren and low vegetation resulting from recently cleared forests and other tree canopy in association with a timber harvest permit (DE, MD, PA, VA, WV) or having a land use history of forest rotation since the mid 1980's. Timber harvest permit data were not reported to the Chesapeake Bay Program by either New York or the District of Columbia.

37-38 Extractive (EXTR) = Barren lands and impervious surfaces within quarries, surface mines, and other surficial excavation sites.

41; 65; 75; 95 Forest (FORE) = All contiguous patches of trees ≥1 acre in extent with a patch width ≥240-ft somewhere in the patch. The 240-ft girth references potential altered microclimate conditions extending inwards up to 120-ft from the patch edge. The forest understory is assumed to be undisturbed/unmanaged. Forests that are also wetlands are included in this class.

42; 64; 74; 94 Tree Canopy, Other (TCOT) = All trees that do not qualify as “Forest” but are presumed to have an undisturbed/unmanaged understory. Such areas include narrow windbreaks adjacent to cropland and roads and tree canopy patches not qualified as “forest” that are fully surrounded by agriculture. Wetlands with “other tree canopy” are included in this class.

16; 54-56 Natural Succession (NATS) = Barren, herbaceous, or scrub-shrub lands that are not classed as cropland, pasture, turf grass, or pervious developed. These are areas that are presumed to be undergoing either natural or managed succession and will eventually become forested although this process may take years to decades to complete. Abandoned mine lands are included in this class.

61-63 Riverine Wetlands, Non-forested (RIVW) = National Wetlands Inventory (NWI) non pond, non-lake wetlands, emergent wetlands along streams mapped from high-resolution imagery outside Virginia, state designated wetlands, and potential non-tidal wetlands (for Pennsylvania only) located within the FEMA designated 100-year floodplain, DEM-aligned 1:24,000 scale buffered stream network, SSURGO hydric or frequently flooded soils.

71-73 Terrene Wetlands, Non-forested (TERW) = National Wetlands Inventory (NWI) non pond, non-lake wetlands, emergent wetlands mapped from high-resolution imagery outside Virginia, state designated wetlands, and state potential non-tidal, non-floodplain wetlands (for Pennsylvania only). These are spatially isolated wetlands on ridges and slopes that are most prevalent in the coastal plain where streams may originate from wetland complexes.

81-82; 87-88 Cropland (CROP) = Barren and low vegetation lands on large parcels (> 5 acres) that are mapped as cropland in the 2018 Cropland Data Layer 83-85 Pasture/Hay (PAST) = Barren, low vegetation, and scrub shrub lands on large parcels (> 5 acres) that are mapped as pasture in the 2019 National Land Cover Dataset or the 2018 Cropland Data Layer

91-93 Tidal Wetlands, Non-forested (TDLW) = All wetlands mapped as estuarine or marine according to National Wetlands Inventory (NWI) plus any adjacent freshwater emergent wetlands, and emergent wetlands mapped from high-resolution imagery outside Virginia must be within 1-ft of adjacent tidal water elevations derived from NOAA’s Sea Level Rise dataset. (https://www.fws.gov/wetlands/Documents/Wetlands-and-Deepwater-Habitats-Classification-chart.pdf)

B.2 Biophysical input tables for each InVEST

Table B-1: Flood Mitigation Model - CN_A, CN_B, CN_C, and CN_D are the curve number values for each land use/land cover (LULC) type corresponding to soil hydrologic groups A, B, C, or D, respectively.

Flood Mitigation Model						
Land Use Code	General Land Use	Land Use Description	CN_A	CN_B	CN_C	CN_D
11	Water	Estuarine/Marine	100	100	100	100
12	Water	Lakes and Reservoirs	100	100	100	100
13	Water	Riverine Ponds	100	100	100	100
14	Water	Terrene Ponds	100	100	100	100
15	Water	Lotic Water (fresh)	100	100	100	100
21	Impervious Roads	Roads	98	98	98	98
22	Impervious Structures	Structures	98	98	98	98
23	Impervious, Other	Other Impervious	98	98	98	98
24	Tree Canopy over Impervious	Tree Canopy Over Roads	98	98	98	98
25	Tree Canopy over Impervious	Tree Canopy Over Structures	98	98	98	98
26	Tree Canopy over Impervious	Tree Canopy Over Other Impervious	98	98	98	98
28	Turf Grass	Turf Grass	67.5	75	79.5	82.8
29	Pervious Developed, Other	Transitional Barren	77.5	86	90.8	92.6
51	Pervious Developed, Other	Suspended Succession Barren	77.5	86	90.8	92.6
52	Pervious Developed, Other	Suspended Succession Herbaceous	68.9	76	80.3	83.3
53	Pervious Developed, Other	Suspended Succession Scrub/Shrub	59	69	74.8	79.3
27	Tree Canopy over Turf Grass	Tree Canopy Over Turf Grass	67.5	75	79.5	82.8
41	Forest	Forest	32	50	60	68.5
42	Tree Canopy, Other	Other Tree Canopy	46.2	60	67.8	74.2
54	Natural Succession	Natural Succession Barren	77.5	86	90.8	92.6
55	Natural Succession	Natural Succession Herbaceous	68.9	76	80.3	83.3
56	Natural Succession	Natural Succession Scrub/Shrub	59	69	74.8	79.3
83	Pasture/Hay	Pasture/Hay Barren	77.5	86	90.8	92.6

84	Pasture/Hay	Pasture/Hay Herbaceous	68.9	76	80.3	83.3
85	Pasture/Hay	Pasture/Hay Scrub/Shrub	59	69	74.8	79.3
91	Wetlands, Tidal Non-forested	Tidal Wetlands Barren	88.8	90	91.2	91.3
92	Wetlands, Tidal Non-forested	Tidal Wetlands Herbaceous	88.8	90	91.2	91.3
93	Wetlands, Tidal Non-forested	Tidal Wetlands Scrub/Shrub	88.8	90	91.2	91.3
94	Tree Canopy, Other	Tidal Wetlands Tree Canopy	88.8	90	91.2	91.3
95	Forest	Tidal Wetlands Forest	87.4	89	90.4	90.7
62	Wetlands, Riverine Non-forested	Riverine Wetlands Herbaceous	88.8	90	91.2	91.3
63	Wetlands, Riverine Non-forested	Riverine Wetlands Scrub/Shrub	88.8	90	91.2	91.3
64	Tree Canopy, Other	Riverine Wetlands Tree Canopy	89	90	91	91
65	Forest	Riverine Wetlands Forest	87.4	89	90.4	90.7
72	Wetlands, Terrene Non-forested	Terrene Wetlands Herbaceous	88.8	90	91.2	91.3
73	Wetlands, Terrene Non-forested	Terrene Wetlands Scrub/Shrub	88.8	90	91.2	91.3
74	Tree Canopy, Other	Terrene Wetlands Tree Canopy	88.8	90	91.2	91.3
75	Forest	Terrene Wetlands Forest	87.4	89	90.4	90.7

Table B-2: Stormwater Retention Model - RC_A, RC _B, RC _C, and RC _D are the stormwater runoff coefficients corresponding to soil hydrologic groups A, B, C, or D, respectively. EMC_Pollutant refers to the event mean concentration of pollutant in stormwater (total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP)).

Stormwater Retention Model									
Land Use Code	General Land Use	Land Use Description	RC_A	RC_B	RC_C	RC_D	EMC_TSS	EMC_TN	EMC_TP
11	Water	Estuarine/Marine	0	0	0	0	0	0	0
12	Water	Lakes and Reservoirs	0	0	0	0	0	0	0
13	Water	Riverine Ponds	0	0	0	0	0	0	0
14	Water	Terrene Ponds	0	0	0	0	0	0	0
15	Water	Lotic Water (fresh)	0	0	0	0	0	0	0
21	Impervious Roads	Roads	0.85	0.85	0.85	0.85	150	2.7	0.4
22	Impervious Structures	Structures	0.85	0.85	0.85	0.85	150	2.7	0.4
23	Impervious, Other	Other Impervious	0.85	0.85	0.85	0.85	150	2.7	0.4
24	Tree Canopy over Impervious	Tree Canopy Over Roads	0.82	0.82	0.82	0.82	150	2.7	0.4
25	Tree Canopy over Impervious	Tree Canopy Over Structures	0.82	0.82	0.82	0.82	150	2.7	0.4
26	Tree Canopy over Impervious	Tree Canopy Over Other Impervious	0.82	0.82	0.82	0.82	150	2.7	0.4
28	Turf Grass	Turf Grass	0.1	0.13	0.16	0.2	50	1.8	0.14
29	Pervious Developed, Other	Transitional Barren	0.1	0.13	0.16	0.2	50	1.8	0.14
51	Pervious Developed, Other	Suspended Succession Barren	0.1	0.13	0.16	0.2	50	1.8	0.14
52	Pervious Developed, Other	Suspended Succession Herbaceous	0.1	0.13	0.16	0.2	50	1.8	0.14
53	Pervious Developed, Other	Suspended Succession Scrub/Shrub	0.1	0.13	0.16	0.2	50	1.8	0.14

27	Tree Canopy over Turf Grass	Tree Canopy Over Turf Grass	0.1	0.13	0.16	0.2	50	1.8	0.14
41	Forest	Forest	0.05	0.08	0.11	0.15	50	1.8	0.14
42	Tree Canopy, Other	Other Tree Canopy	0.05	0.08	0.11	0.15	50	1.8	0.14
54	Natural Succession	Natural Succession Barren	0.1	0.13	0.16	0.2	50	1.8	0.14
55	Natural Succession	Natural Succession Herbaceous	0.1	0.13	0.16	0.2	50	1.8	0.14
56	Natural Succession	Natural Succession Scrub/Shrub	0.1	0.13	0.16	0.2	50	1.8	0.14
83	Pasture/Hay	Pasture/Hay Barren	0.1	0.13	0.16	0.2	50	1.8	0.14
84	Pasture/Hay	Pasture/Hay Herbaceous	0.1	0.13	0.16	0.2	50	1.8	0.14
85	Pasture/Hay	Pasture/Hay Scrub/Shrub	0.1	0.13	0.16	0.2	50	1.8	0.14
91	Wetlands, Tidal Non-forested	Tidal Wetlands Barren	0	0	0	0	0	0	0
92	Wetlands, Tidal Non-forested	Tidal Wetlands Herbaceous	0	0	0	0	0	0	0
93	Wetlands, Tidal Non-forested	Tidal Wetlands Scrub/Shrub	0	0	0	0	0	0	0
94	Tree Canopy, Other	Tidal Wetlands Tree Canopy	0	0	0	0	0	0	0
95	Forest	Tidal Wetlands Forest	0	0	0	0	0	0	0
62	Wetlands, Riverine Non-forested	Riverine Wetlands Herbaceous	0	0	0	0	0	0	0
63	Wetlands, Riverine Non-forested	Riverine Wetlands Scrub/Shrub	0	0	0	0	0	0	0
64	Tree Canopy, Other	Riverine Wetlands Tree Canopy	0	0	0	0	0	0	0
65	Forest	Riverine Wetlands Forest	0	0	0	0	0	0	0

72	Wetlands, Terrene Non-forested	Terrene Wetlands Herbaceous	0	0	0	0	0	0	0
73	Wetlands, Terrene Non-forested	Terrene Wetlands Scrub/Shrub	0	0	0	0	0	0	0
74	Tree Canopy, Other	Terrene Wetlands Tree Canopy	0	0	0	0	0	0	0
75	Forest	Terrene Wetlands Forest	0	0	0	0	0	0	0

Table B-3: Urban Cooling Model - Kc is an evapotranspiration coefficient. The value of green area indicates whether that LULC class is considered a green area. Shade represents the proportion of area in each LULC class that is covered by tree canopy at least 2 meters high. Albedo represents the proportion of solar radiation that is directly reflected by each LULC class. Building intensity represents the ratio of building floor area to footprint area.

Urban Cooling Model							
Land Use Code	General Land Use	Land Use Description	Shade	Kc	Albedo	Green area	Building intensity
11	Water	Estuarine/Marine	0	0.9	0.06	1	0
12	Water	Lakes and Reservoirs	0	0.9	0.06	1	0
13	Water	Riverine Ponds	0	0.9	0.06	1	0
14	Water	Terrene Ponds	0	0.9	0.06	1	0
15	Water	Lotic Water (fresh)	0	0.9	0.06	1	0
21	Impervious Roads	Roads	0	0.1	0.2	0	0
22	Impervious Structures	Structures	0	0.1	0.15	0	1
23	Impervious, Other	Other Impervious	0	0.1	0.18	0	0
24	Tree Canopy over Impervious	Tree Canopy Over Roads	1	1	0.15	0	0
25	Tree Canopy over Impervious	Tree Canopy Over Structures	1	1	0.15	0	1
26	Tree Canopy over Impervious	Tree Canopy Over Other Impervious	1	1	0.15	0	0
28	Turf Grass	Turf Grass	0	0.85	0.2	1	0
29	Pervious Developed, Other	Transitional Barren	0	0.5	0.28	0	0
51	Pervious Developed, Other	Suspended Succession Barren	0	0.5	0.28	1	0
52	Pervious Developed, Other	Suspended Succession Herbaceous	0	0.85	0.2	1	0
53	Pervious Developed, Other	Suspended Succession Scrub/Shrub	0	1	0.23	1	0

27	Tree Canopy over Turf Grass	Tree Canopy Over Turf Grass	1	1	0.2	1	0
41	Forest	Forest	1	1	0.15	1	0
42	Tree Canopy, Other	Other Tree Canopy	1	1	0.15	1	0
54	Natural Succession	Natural Succession Barren	0	0.5	0.28	0	0
55	Natural Succession	Natural Succession Herbaceous	0	0.85	0.2	1	0
56	Natural Succession	Natural Succession Scrub/Shrub	0	1	0.23	1	0
83	Pasture/Hay	Pasture/Hay Barren	0	0.5	0.28	0	0
84	Pasture/Hay	Pasture/Hay Herbaceous	0	0.85	0.2	1	0
85	Pasture/Hay	Pasture/Hay Scrub/Shrub	0	1	0.23	1	0
91	Wetlands, Tidal Non-forested	Tidal Wetlands Barren	0	1.1	0.28	1	0
92	Wetlands, Tidal Non-forested	Tidal Wetlands Herbaceous	0	1.1	0.2	1	0
93	Wetlands, Tidal Non-forested	Tidal Wetlands Scrub/Shrub	0	1.1	0.23	1	0
94	Tree Canopy, Other	Tidal Wetlands Tree Canopy	1	1.1	0.2	1	0
95	Forest	Tidal Wetlands Forest	1	1.1	0.15	1	0
62	Wetlands, Riverine Non-forested	Riverine Wetlands Herbaceous	0	1.1	0.2	1	0
63	Wetlands, Riverine Non-forested	Riverine Wetlands Scrub/Shrub	0	1.1	0.23	1	0
64	Tree Canopy, Other	Riverine Wetlands Tree Canopy	1	1.1	0.15	1	0
65	Forest	Riverine Wetlands Forest	1	1.1	0.15	1	0

72	Wetlands, Terrene Non-forested	Terrene Wetlands Herbaceous	0	1.1	0.2	1	0
73	Wetlands, Terrene Non-forested	Terrene Wetlands Scrub/Shrub	0	1.1	0.23	1	0
74	Tree Canopy, Other	Terrene Wetlands Tree Canopy	1	1.1	0.15	1	0
75	Forest	Terrene Wetlands Forest	1	1.1	0.15	1	0

Table B-4: Carbon Storage Model - C_above, C_below, C_soil, and C_dead refers to the Carbon density of aboveground biomass, belowground biomass, soil, and dead matter, respectively (metric tons of C).

Carbon Storage Model						
Land Use Code	General Land Use	Land Use Description	C_above	C_below	C_soil	C_dead
11	Water	Estuarine/Marine	0	0	0	0
12	Water	Lakes and Reservoirs	0	0	0	0
13	Water	Riverine Ponds	0	0	0	0
14	Water	Terrene Ponds	0	0	0	0
15	Water	Lotic Water (fresh)	0	0	0	0
21	Impervious Roads	Roads	0	0	0	0
22	Impervious Structures	Structures	0	0	0	0
23	Impervious, Other	Other Impervious	0	0	0	0
24	Tree Canopy over Impervious	Tree Canopy Over Roads	90	25	63	0
25	Tree Canopy over Impervious	Tree Canopy Over Structures	90	25	63	0
26	Tree Canopy over Impervious	Tree Canopy Over Other Impervious	90	25	63	0
28	Turf Grass	Turf Grass	8	0	0	0
29	Pervious Developed, Other	Transitional Barren	4	0	0	0
51	Pervious Developed, Other	Suspended Succession Barren	4	0	0	0
52	Pervious Developed, Other	Suspended Succession Herbaceous	50	0	0	0
53	Pervious Developed, Other	Suspended Succession Scrub/Shrub	50	0	0	0
27	Tree Canopy over Turf Grass	Tree Canopy Over Turf Grass	104	25	0	0
41	Forest	Forest	104	25	63	13
42	Tree Canopy, Other	Other Tree Canopy	104	25	63	0
54	Natural Succession	Natural Succession Barren	4	0	25	0
55	Natural Succession	Natural Succession Herbaceous	50	0	25	0
56	Natural Succession	Natural Succession Scrub/Shrub	50	0	25	0

83	Pasture/Hay	Pasture/Hay Barren	4	0	25	0
84	Pasture/Hay	Pasture/Hay Herbaceous	50	0	25	0
85	Pasture/Hay	Pasture/Hay Scrub/Shrub	50	0	25	0
91	Wetlands, Tidal Non-forested	Tidal Wetlands Barren	4	0	88	0
92	Wetlands, Tidal Non-forested	Tidal Wetlands Herbaceous	50	0	88	0
93	Wetlands, Tidal Non-forested	Tidal Wetlands Scrub/Shrub	50	0	88	0
94	Tree Canopy, Other	Tidal Wetlands Tree Canopy	104	25	88	0
95	Forest	Tidal Wetlands Forest	104	25	63	13
62	Wetlands, Riverine Non-forested	Riverine Wetlands Herbaceous	50	0	88	0
63	Wetlands, Riverine Non-forested	Riverine Wetlands Scrub/Shrub	50	0	88	0
64	Tree Canopy, Other	Riverine Wetlands Tree Canopy	104	25	88	0
65	Forest	Riverine Wetlands Forest	104	25	63	13
72	Wetlands, Terrene Non-forested	Terrene Wetlands Herbaceous	50	0	88	0
73	Wetlands, Terrene Non-forested	Terrene Wetlands Scrub/Shrub	50	0	88	0
74	Tree Canopy, Other	Terrene Wetlands Tree Canopy	104	25	88	0
75	Forest	Terrene Wetlands Forest	104	25	63	13

B.3 InVEST UES Model Parameters

Table B-5 1: InVEST model parameters and sources.

InVEST Model /Tool Name (Urban ecosystem service modelled)	Parameters	Input	Source
InVEST – Carbon Storage and Sequestration (CC)	Current LULC raster	Chesapeake Conservancy 1-meter LULC raster	Chesapeake Bay Program Office (CPBO 2022)
	Carbon Pools (above ground and below ground biomass)	Carbon Dioxide Information Analysis Center's (CDIC) global biomass carbon lookup tables	(Ruesch & Gibbs 2008)
	Carbon Pools (leaf litter and soils)	IPCC (2006) reference tables	(wetland soils; Aalde et al. 2006).
InVEST - Urban Cooling Model (HM)	LULC raster	Chesapeake Conservancy 1-meter LULC raster	Chesapeake Bay Program Office (CPBO 2022)
	Reference evapotranspiration raster	Global ET0 1970-2001	(Trabucco and Zomer, 2018)
	Area of interest	Corridors shapefile and D.C neighbourhood clusters shapefile	https://opendata.dc.gov/datasets/DCGIS::neighborhood-clusters/explore
	UHImax	D.C Max temp: 105F / 40.5c	NOAA (2021)
	Reference air temperature	Woodbine MD: 86F / 30c	NOAA (2021)

	Air temperature maximum blending distance	500m	(Sharp *et al.*, 2020)
	Green area maximum cooling distance	450m	(Sharp et al., 2020)
	Biophysical table (based on LULC)	See Table B-3	(Allen *et al.*, 1998; Stewart and Oke, 2012)
2. InVEST - Urban Flood Risk Mitigation (Runoff Retention)	LULC raster	Chesapeake Conservancy 1-meter LULC raster	Chesapeake Bay Program Office (CPBO 2022)
	Soil hydrological groups	USDA's Natural Resources Conservation Service (NRCS) Soil Survey Geographic Database (SSURGO) raster	(Soil Survey Staff 2022)
	Areas of Interest	Corridors shapefile and D.C neighbourhood clusters shapefile	https://opendata.dc.gov/datasets/DCGIS::neighborhood-clusters/explore
	Depth of rainfall	211.33mm	(NOAA 2006)
	Biophysical table (based on LULC)	See Table B-1	(USDA 1986; Westenbroek et al. 2010)
InVEST -Urban Stormwater Retention (SWR)	LULC raster	Chesapeake Conservancy 1-meter LULC raster	Chesapeake Bay Program Office (CPBO 2022)
	Soil hydrological groups	USDA's Natural Resources Conservation Service (NRCS) Soil Survey Geographic Database (SSURGO)	(Soil Survey Staff 2022)
	Precipitation	1062 mm/year	NOAA Climate Prediction Center
	Biophysical table (based on LULC)	See Table B-2	(McCuen 2016; DOEE 2016)

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