ABSTRACT

Title of Dissertation:	THE EFFECTS OF HISTORICAL AND CURRENT LAND USE ON THE HABITAT USE AND COMMUNITIES OF URBAN WILDLIFE IN THE WASHINGTON, D.C. METROPOLITAN AREA
	Merri K. Collins, Doctor of Philosophy in Environmental Science and Technology, 2024
Dissertation directed by:	Dr. Travis Gallo, Department of Environmental Science and Technology

Understanding past legacies of urban land use is important to identify ecological processes and inform best management practices for wildlife-friendly cities in the future. My first dissertation chapter is an overview of my personal research philosophy and how it relates to this dissertation. The second chapter is a systematic literature review that addresses the state of global urban wildlife research. Urban wildlife research is predominantly conducted in North America, Europe, and Australia by academic researchers, and less so in the Global South. The third chapter explores how a gregarious species, the Eastern wild turkey (*Meleagris gallopavo silvestris*) once extant from the Washinton, D.C. landscape, is making a comeback. Wild turkey had a higher probability of occupying sites further from roads and at lower elevations. The fourth and concluding chapter looks at historic neighborhood valuation in Washington, D.C. to identify any legacy effects of racist and discriminatory urban planning on mammal communities. While I did not find any relationship, I did find similar mammal communities across the city regardless of neighborhood categorization and I derive management implications from this information.

THE EFFECTS OF HISTORICAL AND CURRENT LAND USE ON THE HABITAT USE AND COMMUNITIES OF URBAN WILDLIFE IN THE WASHINGTON, D.C. METROPOLITAN AREA

by

Merri K. Collins

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]
[2024]

Advisory Committee:
Professor [Travis Gallo], Chair
[Jennifer Mullinax]
[Katherine Edwards]
[Mitchell Pavao-Zuckerman]
[L. Jen Shaffer]

Dedication

To the women who have been underestimated. To the women who have been torn down and oppressed. To the women who were told you will never succeed on your own, you are not smart enough, you are weak, you are small and insignificant. To the women who had to bow to a man but held a smirk on their faces and a fire in their hearts while they were face down in the dirt. To the women who had no choice but to survive it all. I see you. My successes stand on your shoulders. I am in awe of your strength. You are the salt of this earth. They are terrified of our strength, our resilience, our power. This one is for you.

Acknowledgements

I first want to thank my advisor, Dr. Travis Gallo. You saw something in me and gave me the opportunity to pursue this degree. Starting this journey together both as newbies, during the height of the Covid-19 pandemic, was quite the ride—but I would not have wanted to be mentored by anyone else. Your kindness, respect, and empathy were seen and valued over the last four years. I think we both watched one another go through personal and professional growth, and I am immensely thankful for your constant support. Your exciting career as a professor, guided by your kindness, is going to be filled with many happy and grateful alumni of the URBANxNATURE Lab.

I also want to thank my fiancé Jacob Blandford, you have been my cheerleader, shoulder to lean on, and source of laughter even in the dark—I am so thankful to have you during this wild time in my life.

Thank you to all my committee members both at UMD and GMU, I appreciate the support and time you all have given me. Thank you to my lab mates (as well as the members of the Luther, Freeman, and Mullinax labs), undergraduates who barred the brunt of the photo ID work that made this research possible, and the graduate students who laughed with me through so many practice talks, conferences, and events. Cheers to you and your budding careers. I am lucky to call you, my colleagues.

Thank you to my family, especially my mom, Bev, and my late grandparents, Rusell and Betty for nurturing my empathy, curiosity, and my compassion for all living things. Thank you to ALL my friends who have helped me survive over the years. Liz, Amy, Chelsea, Mario, Jackie, Eileen, Heather, Zeb, Ski, Ashley, Erin, Kassey, the sea lab crew, and so many more…. y'all are truly my family, and it would take an entire dissertation to list you all and the things you have done to support me. I love you; I cherish you, and would not be here figuratively, or literally without the grace, love, and encouragement you have shown me. I do not know what I did for life to gift me this kind of unwavering support system, but I will remain eternally grateful.

Finally, I want to acknowledge the bulk of this work was conducted during the height of the COVID-19 pandemic. Many lives, loved ones, and beautiful souls were lost during this time throughout the STEM community. Many graduate students were lost to this disease or suffered the loss of family members. It is not lost on me that aging is a privilege, and I am privileged to be here, alive, writing this. I am so thankful.

Thank you all.

Table of Contents

This Table of Contents is automatically generated by MS Word, linked to the Heading formats used within the Chapter text.

List of Tables

List of Figures

Chapter 1: Creating an Understanding of Ecological Past, Present, and Future

Introduction

It is well known that humans have irrevocably altered the earth's landscapes. While cities – with environments, climates, and even species compositions entirely shaped by human decisions – are the most extreme example (Shochat et al., 2010). Humans have left signs of our presence even in the most remote areas of earth's mountain tops and oceans (Lewis et al., 2015). Pollutants like microplastics have been found in the deep ocean, effecting the ecology of marine systems, even if barely touched by man (Horton & Barnes 2020). Humans leave our footprint everywhere, in the air, in the water, in the make-up of species assemblages and landscape composition (Lewis et al., 2015).

However, not all hope is lost despite the changes human development has caused, as nature has shown resiliency through time and despite human disturbances. Species extirpated from entire landscapes have started to return – and some, like white-tailed deer (*Odocoileus virginianus*) and wild turkey (*Meleagris gallopavo*), in large numbers (Cardoza 1993; Waller & Alverson 1997). Indigenous and local knowledge has always held a place for prioritizing nature in human systems, often not separating the two, but decision makers have only recently started prioritizing ecological conservation as a sustainability goal (Mata et al., 2020). Trees and vegetation have been allowed to regrow around major cities, and the value in adding public parks and green spaces for urban communities to experience nature is being realized (Faeth et al., 2011).

Policy makers and resource managers are beginning to see that human only landscapes are, in many ways, bad for humans (Mata et al., 2020). Laws have been passed to improve and regulate air and water quality, and people have begun to understand the benefits of nature to human health and the beauty that nature can bring to cityscapes (Dearborn & Kark 2010). Integrating green infrastructure and applying ecological solutions to anthropogenic problems is gaining traction in urban planning (Mata et al., 2020). Ecological research focusing on urban wildlife began only in the 1990s (Magle et al., 2012). At this time, researchers acknowledged that there was little understanding of urban ecology, and since the 1990's there has been a steady uptick in urban wildlife research specifically (Collins et al., 2020; Magle et al., 2012).

Now, in 2024, we have specific conferences and even university departments dedicated to studying and conserving nature in cities. In fact, networks of urban researchers are being established across the country (Magle et al., 2019). To date, there is a strong interdisciplinary focus on applied urban research—understanding how to effectively nurture and conserve species in recovering environments, expand suitable habitats, and peacefully allow the return of many once locally extirpated species to human dominated landscapes (Collins et al., 2021; Dover 2015; Magle et al., 2019).

Urban ecology research is especially challenging, as it requires the consideration of and appreciation for a balance between human well-being, human infrastructure, and biological conservation in landscapes historically designed solely for people (Cox et al., 2013). With this level of complexity, creating research plans that consider the ecological and social past, present, and future is crucial for success (Foster et al., 2003; Szabó 2010). This framework for addressing ecological problems provides a balanced approach to management that promotes proactive solutions based on all available scenarios. For instance, how can we know what conservation strategies will work, if we have yet to acknowledge which strategies did not work? How can we deliver ecosystem services to those that need them the most, if we do not recognize historic inequities in nature access? And can we know where research efforts need to be directed, if we do not know what areas of our study system lack data? For this reason, I have chosen to structure my own dissertation questions and resulting chapters around the framework of "Past, Present, and Future."

Connecting My Research to My Ethos

As someone who grew up in a rural landscape and had, until this point, only conducted rural based ecological research, undertaking an urban wildlife study was quite a novel challenge and new adventure for me. I had to change the way I thought about cities, and open my mind to embracing new theories, practices, and ideas about how people and wildlife interact and intersect.

Conducting research in a major city allowed me to develop my own new philosophy about conservation that incorporates ecological past, present, and future needs of wildlife and people. I plan to integrate this personal philosophy as a framework into future research questions because it creates a balance of understanding and a solid foundation to ask impactful research questions.

Looking to the Past

With no experience working as an ecologist in urban landscapes, I wanted to prepare myself as best I could to tackle research in this arena. I needed to build a roadmap for myself by understanding where the current state of urban wildlife research was, and where it is heading. While all aspects of my dissertation chapters include a description of ecological past, I wanted to understand the overall discipline of urban wildlife better. To understand the current state of urban wildlife research, I conducted a systematic literature review that updated a literature review conducted by Magle et al. (2012) a decade before.

The results from this literature review highlighted past trends of urban wildlife research, the present state of research globally, and identified topics with significant data gaps that warrant future research. These findings also provided an important framework for choosing topics for my third and fourth chapters. Herein, you will find this review structured as chapter two of this dissertation.

Working in the Present

In Chapter 2, I found that there are many studies on urban birds. However, I found a lack of publications on urban wild turkeys specifically (Tinsley 2014), despite the national media attention urban turkeys were receiving for having thriving populations across eastern U.S. cities. National news outlets were reporting that urban turkeys are doing so well in cities, that growing populations were beginning to cause conflict with urban residents. Despite populations thriving in cities, rural wild turkey populations have recently been estimated to be declining due to unknown causes

(Chamberlain et al., 2022). This surge in urban populations, but decline in rural populations piqued my curiosity about urban turkeys in our region, Washington, D.C.

Turkeys were once abundant across my study region in Washington, D.C. but became extirpated in the early 20th century due to habitat loss and overharvest (Cardoza 1993). However, I had captured wild turkey on wildlife cameras (described in Chapter 3) often at our field sites. These observations caused me to ask two key questions: 1) how is a species that was once extirpated from urban areas (Cardoza 1993) able to return to and even thrive in a modern urbanized landscape? and 2) with such drastic habitat changes due to urbanization, what modern landscape characteristics are turkeys drawn to that increases their probability of occupying urban sites? Herein, Chapter 3 tells the story of wild turkeys successfully returning to an urban landscape, and what habitat characteristics are influencing present day occupancy of this native species.

Looking to the Past to Plan for the Future

Additionally, my literature review (Chapter 2) found that little research focused on the intersection of socioecological research and urban wildlife (Collins et al., 2021) – particularly regarding environmental justice. Schell et al., (2020) provided a greater understanding of the intersection between wildlife diversity, environmental justice, and social inequities in cities. Wherein, distinct urban conditions that affect wildlife and humans are created by landscape modifications and urban design decisions, and these decisions do not provide fair equity to all residents. These inequities include limiting or removing access to nature due to systematic historic racism in the urban planning process (Schell et al., 2020). As someone who has always had access to nature from an early age and therefore access to its benefits, learning that residents of cities have been cut off from nature due to social inequities and racial segregation was alarming to me.

I believe nature should be accessible to everyone. The mental and physical benefits of having access to natural areas, especially in cities, where greenspaces can provide refuge from heat, noise, and poor air quality (Lee & Maheswaran, 2011), should be accessible to all residents regardless of income level or race. However, redlining, a form of historic neighborhood segregation based on race, has shaped the structure of U.S. cities including where community resources are distributed, where green spaces are accessible, and subsequently, the distribution of plants and animals on the landscape.

Focusing on the history of housing segregation in our study area, prompted me to ask 1) has redlining (and thus how resources have been historically distributed across the city) influenced the mammal communities in our region and the probability that a mammal species can persist in these different categorized neighborhoods? And 2) how can we use this information to improve current and future management strategies to simultaneously conserve urban wildlife and address social inequities in access to nature? Herein, you will find the story of redlining and the relationship between housing grades and mammal community composition in our study region as Chapter 4 of this dissertation.

Conclusion

To conclude, I hope this dissertation illustrates my growth as a researcher, scientist, and ultimately a person. I believe that the development of my own research philosophy and personal world views has expanded through this work. I have challenged myself to tackle questions that integrate a holistic framework for thinking about ecological problems as equal parts past, present, and future. I am hopeful that we are reaching a turning point in ecological studies, where the majority will recognize that our past mistakes can be a roadmap for future successes if we shift from reactive to initiative-taking management strategies. It is my hope that I can use the knowledge gained from this dissertation throughout my career as a scientist to promote a healthy relationship between people and the natural world, no matter where my research takes place.

Literature Cited

Cardoza, J. E. (1993). The Wild Turkey: An Update. Bird Observer. 21(5).

Chamberlain, M. J., Hatfield, M., & Collier, B. A. (2022). Status and distribution of wild turkeys in the United States in 2019. Wildlife Society Bulletin, 46(2), e1287.

Collins, M. K., Magle, S. B., & Gallo, T. (2021). Global trends in urban wildlife ecology and conservation. Biological Conservation, 261, 109236.

Cox, D. T., Hudson, H. L., Shanahan, D. F., Fuller, R. A., & Gaston, K. J. (2017). The rarity of direct experiences of nature in an urban population. *Landscape and urban planning, 160,* 79-84.

Dearborn, D. C., & Kark, S. (2010). Motivations for conserving urban biodiversity. *Conservation biology, 24*(2), 432-440.

Dover, J. W. (2015). Green infrastructure: incorporating plants and enhancing biodiversity in buildings and urban environments. Routledge.

Edwards, G. A. (2019). Coal and climate change. Wiley Interdisciplinary Reviews: Climate Change, 10(5), e607.

Faeth, S. H., Saari, S., & Bang, C. (2012). Urban biodiversity: Patterns, processes, and implications for conservation. eLS.

Foster, D., Swanson, F., Aber, J., Burke, I., Brokaw, N., Tilman, D., & Knapp, A. (2003). The importance of land-use legacies to ecology and conservation. BioScience, 53(1), 77-88.

Hirschman, E. C. (2022). Climate change migration and the economic rebirth of central appalachia. Social Sciences, 11(10), 462.

Horton, A. A., & Barnes, D. K. (2020). Microplastic pollution in a rapidly changing world: Implications for remote and vulnerable marine ecosystems. Science of The Total Environment, 738, 140349.

Lee, A. C., & Maheswaran, R. (2011). The health benefits of urban green spaces: a review of the evidence. Journal of Public Health, 33(2), 212-222.

Lewis, S. L., & Maslin, M. A. (2015). A transparent framework for defining the Anthropocene Epoch. *The Anthropocene Review*, *2*(2), 128-146.

Magle, S. B., Hunt, V. M., Vernon, M., & Crooks, K. R. (2012). Urban wildlife research: past, present, and future. *Biological Conservation*, *155*, 23-32.

Magle, S. B., Fidino, M., Lehrer, E. W., Gallo, T., Mulligan, M. P., Ríos, M. J., ... & Drake, D. (2019). Advancing urban wildlife research through a multi-city collaboration. *Frontiers in Ecology and the Environment*, *17*(4), 232-239.

Mata, L., Ramalho, C. E., Kennedy, J., Parris, K. M., Valentine, L., Miller, M., ... & Cumpston, Z. (2020). Bringing nature back into cities. People and Nature, 2(2), 350-368.

McNabb, D. E., & Swenson, C. R. (2023). Drought’s Role in a Water Crisis. In America’s Water Crises: The Impact of Drought and Climate Change (pp. 75-89). Cham: Springer Nature Switzerland.

Schell, C. J., Dyson, K., Fuentes, T. L., Des Roches, S., Harris, N. C., Miller, D. S., ... & Lambert, M. R. (2020). The ecological and evolutionary consequences of systemic racism in urban environments. *Science*, *369*(6510), eaay4497.

Shochat, E., Lerman, S. B., Anderies, J. M., Warren, P. S., Faeth, S. H., & Nilon, C. H. (2010). Invasion, competition, and biodiversity loss in urban ecosystems. *BioScience*, *60*(3), 199-208.

Szabó, P. (2010). Why history matters in ecology: an interdisciplinary perspective. Environmental Conservation, 37(4), 380-387.

Tinsley, K. A. (2014). Wild turkeys in the urban matrix: how an introduced species survives-and thrives-in a multifunctional landscape. Dissertation. University of Minnesota.

Waller, D. M., & Alverson, W. S. (1997). The white-tailed deer: a keystone herbivore. Wildlife Society Bulletin, 217-226.

Chapter 2: Global trends in urban wildlife ecology and conservation

Introduction

Human experiences with wildlife are often thought to be limited to more naturally occurring ecosystems. This is primarily due to historic knowledge of wildlife ecology being rooted in more rural areas. The United Nations now estimates two-thirds of the human population – an estimated seven billion people – will be living in urban areas by 2050 (United Nations, 2018). This shift to urban environments will continue to change how humans and wildlife interact and reshape what we know about wildlife ecology in human-dominated environments (Aronson et al., 2017).

Urbanization has transformed the way researchers, managers, and city planners approach studying wildlife (Apfelbeck et al., 2020; Magle et al., 2012). Since the 1990s, research on urban wildlife has expanded as cities become viewed as novel ecosystems rather than anthropogenic sinks devoid of nature (Forman, 2016; Gallo et al., 2017; Grimm et al., 2000; Lopucki & Kitowski, 2017). Many entities including government agencies and academic labs have expanded their wildlife conservation focus to include urban wildlife research (Magle et al., 2019; Sexton et al., 2015). A new focus on urban ecosystems – and the wild animals that reside there – brings new challenges, opportunities, and solutions to integrate human needs with those of wildlife. The range of research topics within the urban wildlife field continues to grow as the importance of understanding how wildlife live, move, and adapt under anthropogenic conditions becomes paramount in the face of larger environmental problems like climate change, landscape fragmentation, and habitat loss (Forman, 2016; McKinney, 2002; Rastandeh et al., 2018). Solving these large-scale problems will require innovative solutions as urban areas increase across the globe. As a new field, urban wildlife research is in flux, continuing to change and adapt as new questions and issues emerge (Magle et al., 2012). Therefore, it is imperative that practitioners understand the current state of urban wildlife research to identify areas where research is lacking and develop research agendas that inform best practices in management and conservation (Apfelbeck et al., 2020). Looking to past research identifies areas of strength and weakness, reveals potential gaps in current work, and can provide direction for biodiversity conservation within urban areas.

Magle et al. (2012) conducted a literature review of urban wildlife research from 1971-2010. Their review found that urban wildlife research – while limited – was projected to continue increasing past 2010. They also called attention to the need for, and the lack of, integration of various ecological disciplines to answer both fundamental and applied questions within urban wildlife topics. Urban wildlife research is uniquely interdisciplinary, requiring a diverse mix of practitioners, scientists, planners, educators, policy makers, and citizen support (Belaire et al., 2016; Dearborn & Kark, 2010; Schell et al., 2020). To meet the needs of such an interdisciplinary network, it is crucial to understand the status of urban wildlife research within different subjects, contexts, and management areas.

Here I examine the last decade (2011-2020) of urban wildlife research, assess whether the field has addressed research gaps highlighted in Magle et al. (2012), and explore how the field has changed since Magle et al. (2012) first reviewed the literature. To systematically compare the last two decades of urban wildlife research I replicated the literature review methodology used in Magle et al. (2012) for the years 2011- 2020. To identify emerging themes in areas outside of the natural sciences, I conducted an additional systematic review that broadened the Magle et al. (2012) search criteria. I used the upward trend data from Magle et al. (2012) to hypothesize that urban wildlife publications have continued to increase between 2011-2020, and I further predicted that themes understudied during the previous decade (2000-2010) have emerged as common themes due to the increased demand for understanding urban ecosystems. These up-to-date findings provide researchers and urban wildlife practitioners guidance and clarification on knowledge gaps, best practices, and topics of importance.

Materials and Methods

To assess the current state of urban wildlife research I took a two-tier approach to conducting a literature review. First, to directly compare results with Magle et al. (2012), I followed the exact methodology outlined in Magle et al. (2012) – an approach that was limited to sixteen high impact journals in ecology and wildlife research. Second, I expanded our literature review by conducting an additional review that did not limit our search to select journals. Both approaches are outlined below.

Updating Magle et al. (2012) Foundational Review

Literature Search Criteria

Following the methods in Magle et al. (2012), I searched each of the following high impact ecology and wildlife-related journals using Web of Science (WOS) and the term “urban*”: *Animal Behaviour*, *Behavioral Ecology*, *Behavioral Ecology and Sociobiology American Naturalist, Biological Conservation,* and *Conservation Biology*, *Ecology*, *Ecology Letters*, *Journal of Applied Ecology*, *Journal of Wildlife Management*, *Wildlife Research*, *Wildlife Society Bulletin*, *Landscape Ecology*, *Landscape and Urban Planning*, *Nature*, and *Science*. I included the asterisks modifier to capture related terms like “suburban,” “exurban,” and “periurban.” I limited our search to January 2011-December 2020. Following Magle et al. (2012) I sorted results based on titles and abstracts to only include original research conducted on urban wildlife, excluding papers such as letters, reviews, theses, dissertations, and papers that did not directly study wildlife (Magle et al., 2012; Roberts et al., 2006).

Categorization and Analysis

Selected publications were grouped into categories based on the journal discipline (animal behavior, conservation, ecology, general science, landscape ecology, and wildlife biology; Table 1), author affiliation (academic, government, non-government organization, or private industry), taxa of study (mammal, bird, arthropod, herptile, fish, or non-taxa); continent of study area, and the scientific topic of each study (animal behavior, population ecology, community ecology, landscape ecology, conservation, human-wildlife conflict, human dimensions, evolution and genetics, or disease ecology).

To assess trends and changes over the last 20 years, I calculated the proportion of urban wildlife publications in each journal and the proportions of studies for each category. I obtained the original data from Magle et al. (2012) and calculated the same proportions from these data for 2000-2010. I found one error that changed the total number of papers in the 2001-2010 dataset from 429 to 431. This error was verified by the authors and the corrected value was used in our analysis. To quantify changes over time, I compared proportions from 2001-2010 with the proportions from 2011-2020 using chi-squared tests for comparing two proportions (also known as a z-test) in R ver. 3.6 (R Core Team, 2013; Kim, 2017). In some categories, data from Magle et al. (2012) were aggregated for 1971-2010 and I was unable to isolate the results for 2000-2010. In these cases, I only report the percent change between the two datasets.

Broadened Literature Search

Literature Search Criteria

I conducted an additional review using a broader approach. I searched WOS and Google Scholar using the search term "urban* wildlife." An asterisk modifier was again applied to include related terms like "exurban," "suburban", and "periurban" in the search results. I limited our search to the years 2011-2020, but included all scientific journals indexed by WOS and Google Scholar. Google Scholar limits article downloads to the first 1,000 results, regardless of the number of results returned. Therefore, I limited our Google Scholar results to the first 1,000 articles returned by Google Scholar. I did not limit our journal selection by selecting topic specific journals, or only high impact journals to objectively broaden our search. I sorted results based on titles and abstracts to only include original research conducted on urban wildlife and excluded letters, reviews, dissertations, and papers that did not directly study wildlife.

Classification and Analysis

I classified our results in an equivalent way with some additions to each category to accommodate the wider breadth of included journals. “Crustacea” was added to the taxa category, “environmental,” “human dimensions,” “veterinary,” and “zoology,” were added to the journal disciplines based on the journal titles. Research topics, “social science” and “spatial ecology” were added due to the number of journal articles reflecting these topics. Taxa were then classified further into subgroups (e.g., bird to raptor, mammal to carnivore, arthropod to pollinator, herptile to reptile), or into a subtopic if study fell into the “non-taxa” category (e.g., methods, human dimensions). Of these, the subgroup, discipline, and topic with the highest number of publications were classified further to species or subject to examine more fine scale trends and research gaps (Table 4).

Finally, I classified all papers as either “fundamental” or “applied” research based on reading abstracts and discussions sections. Any paper that was founded or built on current knowledge of a subject to improve current theories, but did not have direct or immediate problem-solving application, was classified as fundamental research. Papers classified as applied research included a direct call to action to update policy, urban planning, management, conservation, current methods, or solving immediate problems.

Results

Updating Magle et al. (2012) Foundational Review

Using the search term “urban*” within the sixteen selected journals yielded 2,172 results between January 2011- December 2020. Of these, 532 were determined to fit our inclusion criteria. Of the 1,640 publications excluded from our data, 93.10% (1527/1640) did not research wildlife and 11.46% (188/1640) were not original research articles. The total number of urban wildlife publications from these journals continued to increase between 2011-2020 (0.02% per year), yet at a slightly lower rate than the previous decade (0.06% per year; Fig. 1).

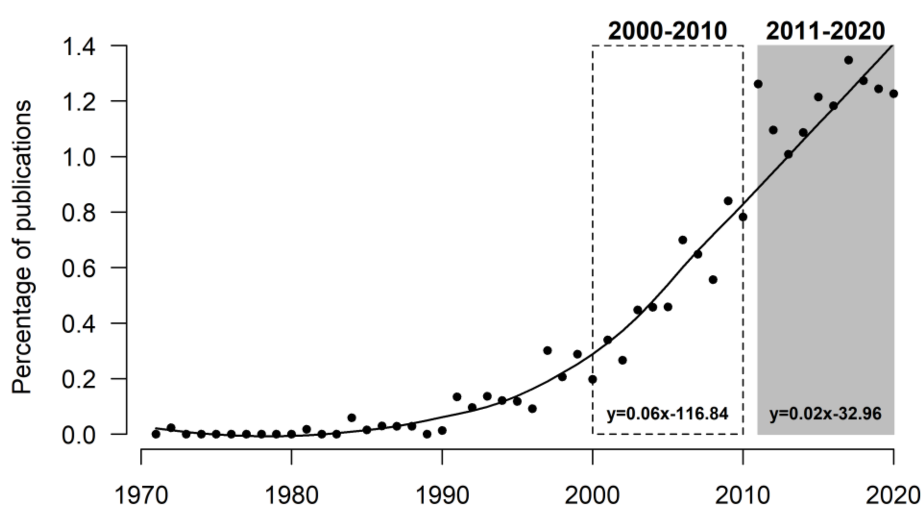

Figure 1. Percentage of urban wildlife publications from sixteen selected journals from 1971-2020.

Journals that typically publish behavior studies saw a significant increase within the last decade (6.5% to 18.2%; χ^2=5.54, p=0.018, df=1). Conservation-related journals significantly decreased in percentage of overall urban wildlife publications, dropping from 32.8% in 2001-2010 to 18.2% in 2011-2020 (χ^2= 3.79, p=0.05, df=1). Additionally, publications in landscape journals continued to rise but had a substantial decrease in urban wildlife publications between 2013-2015 (Fig. 2). Regarding specific journals, Landscape and Urban Planning (5.9%), Wildlife Research (5.3%), and Landscape Ecology (3.6%) continued to produce the highest number of urban wildlife publications of the sixteen selected journals (Table 1). Science (600%; 1 to 5) and Behavioral Ecology and Sociobiology (455%; 7 to 20) had the highest increase of urban wildlife publications compared to the previous analysis, followed by Behavioral Ecology (420%; 11 to 52; Table 1). Although the overall percentages remain low, both Nature (0.04%) and Science (0.07%) experienced increases in urban wildlife publications within the last decade.

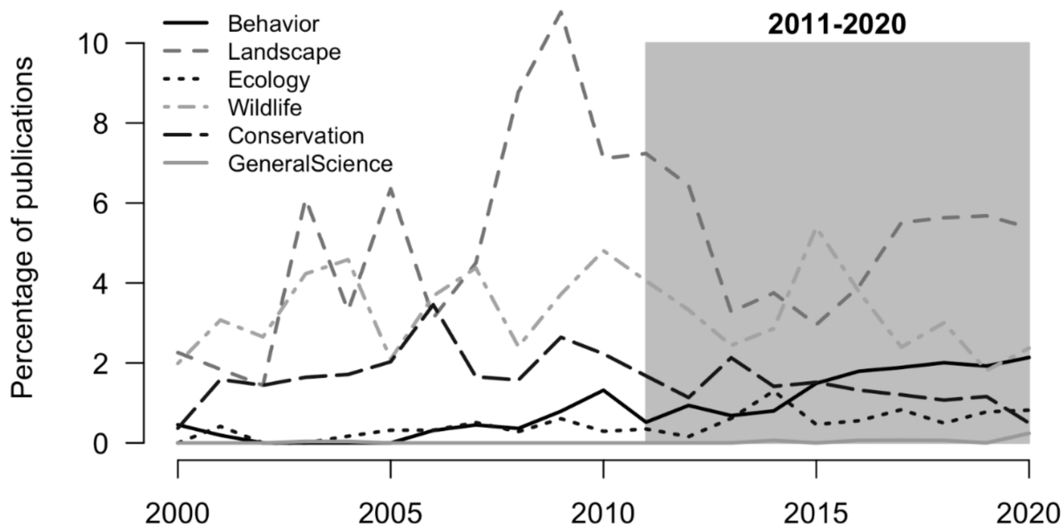

Figure 2. The percentage of urban wildlife publications from 2000-2020 was based on field of study in the sixteen selected journals reviewed. Grey polygon indicates the period of the most recent decade reviewed.

Table 1. A comparison of the total number of publications and the total number of urban wildlife publications between 1971-2010 and 2011-2020 in each of the sixteen high impact ecology and wildlife-related journals reviewed.

		1971-2010			2010-2020			
Journal Discipline	Journal	Total	Urban Wildlife	% Total	Total	Urban Wildlife	% Total	% Change
Anim Behav	Anim Behav	9479	10	0.11	3172	25	0.8	627
	Behav Ecol	2211	11	0.5	1952	52	2.6	420
	Behav Ecol Sociobio	3926	7	0.18	1980	20	1.0	455
Conserv	Am Nat	5456	5	0.09	1754	5	0.3	233
	Biol Conserv	5237	117	2.23	3876	75	1.9	-13
	Conserv Biol	3863	56	1.45	1758	17	0.9	-37
Ecology	Ecol	9401	18	0.19	3262	9	0.3	57
	Ecol Lett	1524	1	0.07	1780	3	0.1	43
	J Appl Ecol	3726	16	0.43	2074	34	1.6	272
Gen Science	Nature	121290	1	<0.01	8568	3	0.04	300
	Science	90350	1	<0.01	7415	5	0.07	600
Land Ecol	Land Urban Plan	2305	85	3.69	1935	116	5.9	59
	Land Ecol	1301	42	3.23	1542	56	3.6	12
Wildlife Biol	J Wildlife Mgmt	6231	77	1.24	1751	38	2.1	75
	Wildlife Res	1405	58	4.13	766	41	5.3	29
	Wildlife Soc B	2931	66	2.25	1014	33	3.2	45

Author Affiliation and Geographic Area

Overall, between 1971- 2020, 80.81% of all urban wildlife publications from the sixteen selected journals had first author affiliations with academic institutions (Table 2). While not statistically significant, academic affiliations increased from 75.75% of total publications in 2001-2010 to 88.90% in 2011-2020. Government institution affiliated authorships in urban wildlife publications decreased from 13.28% in 2001-2010 to 7.10% in 2011-2020, causing an overall total decrease in government affiliated publications from 1971-2020 as reported in Magle et al. (2012; -46%). The overall percentage of NGO affiliated authors significantly decreased within the last two decades from 11.18% in 2001-2010 to 3.42% in 2011-2020 (χ^2= 4.12, p= 0.0427, df=1; Table 3). Private industry urban wildlife publications remain the lowest overall first author affiliation (0.45%, Table 2). Geographically, North America had the most urban wildlife publications within the selected journals (41.7%; n=222), followed by Europe (29.3%; n=156), and Australia (16.1%; n=86), following the same trend identified in Magle et al. (2012). Further, consistent with Magle et al. (2012), Asia (6.2%; n=33), South America (3.9%; n=21), and Africa (2.6%; n=14) remain the lowest publishing continents on urban wildlife research within the sixteen selected journals.

Table 2. Sector of first author affiliations on urban wildlife publications from sixteen selected journals between 1971- 2020 expressed as percentages.

Decade	Urban Wildlife Publications	% Academic	% Government	% NGO	% Private
1971-1980	1	100	0	0	0
1981-1990	13	69.23	23.08	0	7.69
1991-2000	128	66.41	20.31	11.72	1.56
2001-2010	431	75.75	13.28	11.18	0.23
2011-2020	532	88.90	7.10	3.42	0.50
Total	1105	80.81	11.22	7.33	0.45

Scientific Topic and Taxa

The most frequently represented scientific topics in urban wildlife publications from 2011-2020 were animal behavior (23.1%; 123/532), conservation (13.9%; 74/532), and wildlife management (13.1%; 70/532). Other topics represented from 2011-2020 were landscape ecology (12.4%; 66/532), population ecology (12.0%; 64/532), community ecology (9.3%; 50/532), human-wildlife conflict (5.4%; 29/532), disease ecology (5.4%; 29/532), evolution and genetics (3.3%; 19/532) and human dimensions (2.2%;12/532). Notably, topics of disease ecology had an 80% increase in publications over just a ten-year period (5.4% of urban wildlife publications in 2011-2020 compared to 2.6% over four decades – 1971-2010). Similarly, topics in evolution and genetics had a 37% increase over the same time periods from 2.4% between 1971-2010 to 3.3% in 2011-2020.

Urban wildlife publications in the sixteen selected journals remained consistent with Magle et al. (2012) in terms of focusing on specific taxa. Studies were conducted on birds (41.7%) and mammals (30%; Fig. 3). Fish were the least studied taxa representing only 1.3% of all publications from 2011-2020 (7/532). I found no meaningful change in taxa studied between 2001-2010 and 2011-2020 (Table 3). Although not statistically significant, I did see a notable increase in urban arthropod studies and decrease in urban mammal studies between the 2000-2020 decades (Fig. 3).

Table 3. Chi-squared results comparing percentage of urban wildlife publications from 2000-2010 to 2011-2020 by first author affiliation, geographic location of the study, and main taxa of study. Bold p-value indicates a significant change.

	Category	2000-2010	2011-2020	X^2	p-value	df
Author Affiliation	Academia	75.75	88.90	0.9241	0.3364	1
	Government	13.28	7.10	1.4474	0.2291	1
	NGO	11.18	3.42	3.7855	**0.0517**	1
	Private	0.23	0.50	0.1649	0.6846	1
Geographic Location	N. America	51.0	41.7	1.0194	0.3127	1
	Europe	20.7	29.3	1.6398	0.2004	1
	Australia	18.4	16.1	0.6566	0.1976	1
	Asia	6.8	6.20	0.05	0.8231	1
	S. America	3.7	3.90	0.0116	0.9139	1
	Africa	2.8	2.60	1	0	1
Taxa	Arthropod	11.86	17.29	1.0115	0.3145	1
	Bird	41.76	41.72	1.9166	0.9965	1
	Fish	4.17	1.31	1.5291	0.2162	1
	Herptile	10.2	7.33	0.4759	0.4903	1
	Mammal	38.0	30.0	0.9411	0.3321	1
	Non-Taxa	4.17	2.25	0.57302	0.4491	1

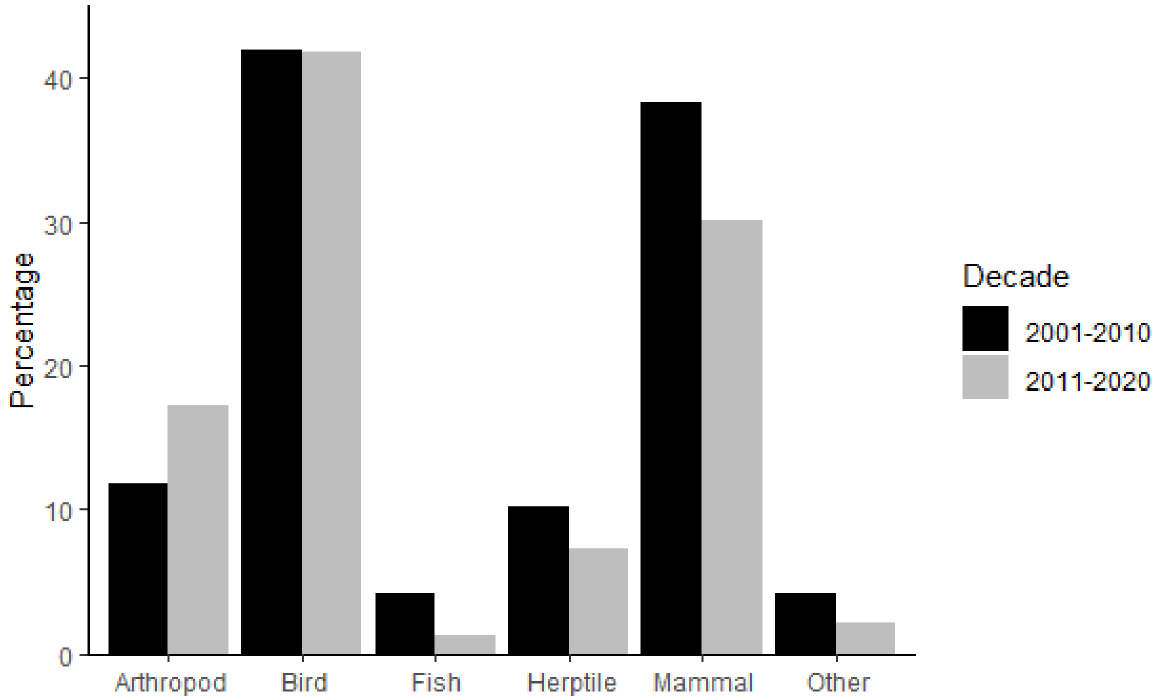

Figure 3. Urban wildlife publications categorized by taxa studied and their percentage of the overall number of urban wildlife studies in the selected sixteen ecology and wildlife journals reviewed.

Broadened Literature Search

Our broadened search using a combination of WOS (n=532) and Google Scholar (n=305) yielded 794 results. Within these results, 203 publications were duplicate publications from both search engines, leaving us with a total of 634 urban wildlife studies from 2011-2020. Within this search, 214 journals were identified and categorized by discipline, as compared to the sixteen journals used in Magle et al. (2012) methods. Disciplines in urban wildlife research included Wildlife Biology (22.9%, n=145), Landscape Ecology (18.9%, n=120), Ecology (10.6%, n=67), Conservation (10.1%, n=64), General Science (8.5%, n=54), Environmental Science

(8.5%, n=54), Veterinary (7.1%, n=45), Human Dimensions (6.6%, n=42), Zoology (4.6%, n=29), and Behavior (2.2%, n=14).

Scientific Topic

Within the broader literature search, urban wildlife research topics consisted of management (19.4%, n=123), disease ecology (12.3%, n=77), social science (10.7%, n=68), behavior (10.2%, n=65), human/wildlife conflict (10.1%, n=64), population ecology (9.9%, n=63), landscape ecology (7.9%, n=50), conservation (6.6%, n=42), community ecology (5.0%, n=32), spatial ecology (4.1%, n=26), and evolution/genetics (3.8%, n=24). Most notably, social sciences had the highest publication percentages within the 2020 year (19.2%, n=10).

Main Taxa and Subgroups

Of the 634 urban wildlife publications, mammal studies represented 46.4% of the total publications (n=294), birds 26.5% (n=168), non-taxa studies 16.4% (n=104), herptiles 4.6% (n=29), arthropods 4.1% (n=26), multiple taxa studies 1.6% (n=10), fish 0.3% (n=2), and crustaceans 0.1% (n=1). I further categorized each of these urban wildlife taxa into subgroups (Table 4). The top three groups within mammal research were carnivores (31.6%, n=93), rodents (17.3%, n=51), and invasive species (12.6%, n=37). Of carnivores, coyotes (*Canis latrans*, 36.5%, n=34) and red foxes (*Vulpes vulpes*, 18.3%, n=17) made up the most studied species. Rodents were represented by studies focused equally on rat species as well as studies encompassing multiple rodent species, (*Rodentia*, 23.5%, n=12). Of the studies investigating multiple rodent species, 50% (n=6) studied rat and mouse species (*Muridae*).

Invasive mammal species were represented by free-roaming domesticated cats (*Felis catus*, 56.8%, n=21), and feral swine (*Sus scrofa*, 30.0%, n=11). Birds were most represented by subclasses songbirds, raptors, and aquatic species. Of these, songbirds were most studied across multiple groups (71.8%, n=51/71). Raptors were most represented by owls (*Strigiformes*, 26.6%, n=8/30) and Cooper's hawks (*Accipiter cooperii*, 23.3%, n=7/30), and of aquatic birds, ibises were the most represented (*Threskiornithinae*, 26.9%, n=7/26). Lizards were the most studied herptile (*Lacertilia*, 34.4%, n=10/29) and pollinators were the highest studied arthropod class (60.0%, n=15/26). Of pollinators, bee species were the most researched (*Anthophil*a, 40%, n=6/15) with butterflies following closely behind (*Rhopalocera*, 33.3%, n=5/15). The most represented subclass of the category "non-taxa" were papers researching urban wildlife methodologies. Of these, methodologies in education (40.2%, n=39/97), development/planning (23.7%, n= 23/97), and statistical modeling (12.4%, n=12/97) were the most prevalent.

When comparing urban wildlife taxa and research topics, mammals were most studied for management (20.7%, n=61/294), disease ecology (14.3%, n=42/294), and human/wildlife conflict research (14.3%, n=42/294). Birds were the most studied for landscape ecology (16.1%, n=27/168) and disease ecology (14.3%, n=24/168). The non-taxa category contained studies most focused on the topics of social sciences (37.7%, n=43/114) and management (35.1%, n=40/114). Herptiles were most represented in studies on population ecology (27/6%, n=8/29). Arthropods were most represented in studies on conservation (30.7%, n=8/26). All fish studies (n=2) were examples of disease ecology research, and the single urban crustacean study was focused on population ecology.

Table 4. Subgroups of taxa (common name) in urban wildlife publications from 2011-2020. The number of total publications is listed by each subgroup of taxa. Abbreviations within mammals C= carnivore, Inv= invasive, B= bats, R= rodents, H= herbivores, MP= marsupials. Within birds RT= raptors, SB= songbirds, AQ= aquatic. For arthropods POL= pollinators. Within herptiles REP= reptiles, and AM= amphibians.

Mammals (234)					
Coyote (C)	34	Multiple Mammals	6	Jackal (C)	1
Dom Cat (Inv)	21	Megachiroptera (B)	5	Skunk (C)	1
Methods	19	Big cats (C)	5	Chipmunk (R)	1
Microchiroptera (B)	19	Prairie dogs (R)	5	Bighorn Sheep (H)	1
Fox (C)	17	Bandicoot (MP)	3	Viscacha (R)	1
Primates	14	Moose (H)	3	Vole (R)	1
Multiple carnivores	14	Multiple Herbivores	3	Woodchuck (R)	1
Multiple rodents	12	Genets (C)	3	Dom Dog (Inv)	1
Rats (R)	12	Hedgehogs	3	Elk (H)	1
Feral Swine (Inv)	11	Mustelids (C)	3	Opossums (Inv)	1
Deer (H)	10	Opossums (MP)	3	Sloth (H)	1
Squirrel (R)	9	Possums (MP)	3	Beaver (R)	1
Raccoon (C)	7	Possums (Inv)	3	Wallaby (MP)	1
Mice (R)	7	Mongoose (C)	2	Multiple Marsupials	1
Kangaroo (MP)	7	Koala (MP)	2		
Lagomorph (H)	6	Hamsters (R)	2		
Bears (C)	6	Dingo (C)	1		
Birds (168)					
Multiple Songbirds	51	Peregrine Falcon (RT)	3	Mockingbird (SB)	1
Parrots	11	Ducks (AQ)	3	Myna (SB)	1
Multiple bird taxa	10	Kites (RT)	3	Nightingale (SB)	1
Owl (RT)	8	Swan (AQ)	3	Eagle (RT)	1
Ibis (AQ)	7	Corvids	2	Red Tailed Hawk (RT)	1
Cooper's Hawk (RT)	7	Chickadee (SB)	2	Sparrow Hawk (RT)	1

Pigeons	6	Martin (SB)	2	Terns (AQ)	1
Invasive	6	Lapwing (AQ)	2	Cormorant (AQ)	1
Tits (SB)	5	Blue Heron (AQ)	1	Egret (AQ)	1
Multiple Raptors	5	Finches (SB)	1	Geese (AQ)	1
Multiple Aquatic	4	House Sparrow (SB)	1	Red-legged Seriema	1
Methods	4	Lark (SB)	1	Hummingbird	1
Blackbirds (SB)	3	Miner (SB)	1		
Gulls (AQ)	3	American Kestrel (RT)	1		
Non-taxa (104)		Arthropods (26)		Herptiles (29)	
Education (Methods)	39	Bees (POL)	6	Lizard (REP)	10
Development (Methods)	23	Multiple (POL)	6	Multiple	5
Modeling (Methods)	12	Butterfly (POL)	5	Snake (REP)	4
Human Dimensions	8	Ticks	2	Turtle (REP)	3
Medicine (Methods)	6	Dragonfly	1	Frog (AM)	3
Technology (Methods)	6	Mosquito	1	Salamander (AM)	2
History (Methods)	5	Blowflies	1	Crocodile (REP)	1
Citizen Science (Methods)	5	Methods	1	Alligator (REP)	1
		Weevils	1		
		Beetle	1		
		Ant	1		
Fish (2)		Crustacea (1)		Multiple Taxa Studies (10)	
Freshwater	2	Water fleas	1		10

Fundamental vs. Applied Research

Of all 634 urban wildlife studies, 66.1% were applied (n=419) and 33.9% (n=215) were considered fundamental or foundational research. Fish had the highest percentage of applied research studies (100%, n=2) followed by herptiles (73.1%, n=19/26), and birds had the lowest percentage of applied research papers (55.9%, n=81). Papers within the “non-taxa” category had a 93.2% applied research rate due to the majority (n=97) being methods papers.

Discussion

Urban wildlife publications have continued increasing within the last decade, indicating urban wildlife research remains an important and expanding field of wildlife and conservation science. Our results indicate new emerging trends within urban wildlife research. While updating Magle et al. (2012), behavior, conservation, and wildlife management were the leading topics during the 2011-2020 decade. However, within our broadened search I found management, disease ecology, and social sciences to be the most studied topics. I also found that applied studies, including papers on research methods, made up a significant percentage of urban wildlife research within the last decade. Trends remaining the same between decades included a geographical bias to North America, and mammals and birds leading research by taxa. Recognizing understudied areas within urban wildlife research can aid researchers in identifying where more information is needed to manage and conserve urban wildlife.

The frequency of published disease ecology research doubled between 2000-2020. After I broadened our literature search to include more journals, disease ecology became the second most studied topic behind management. These results are a likely indication that urban wildlife research is trending towards a more interdisciplinary field as veterinarians, health care professionals, and managers seek to identify possible zoonotic spillover risks in cities and assess linkages between human and wildlife health (Himsworth et al., 2014; Liebler et al., 2018). This trend aligns with the recent increase in the adoption of a "One Health" approach, wherein human health and ecological health are considered one, versus separate issues

(Destoumieux-Garzón et al., 2018). Rapid urbanization impacts surrounding environments and the wildlife residing within them, creating novel opportunities for zoonotic spillovers that would otherwise not be possible. This has been the case with the Ebola, Nipah, and SARS outbreaks within the last decade, as well as the 2020 pandemic resulting from spillover of the novel COVID-19 virus (Mackenzie & Smith, 2020). The need for more advanced zoonotic disease research in urban areas will remain paramount to aid in predicting and modeling emergent vectors and geographic hotspots at risk for zoonotic spillover (Santiago-Alarcon & MacGregor-Fors, 2020).

Methods papers were also a new and prominent addition within our broadened literature search. Educational techniques were the most common subtopic within methods papers. These included urban ecology school program planning, community gardening and ecology program planning, and sociological studies on efficacy of urban wildlife outreach programs (Larson et al., 2016; Patterson et al., 2017; Wieczorek, 2012). These results demonstrate a greater effort to include the public in decision making processes, scientific studies, and a new emphasis on the importance of educating the public about urban wildlife. These findings are encouraging, as a continued focus on education can help mitigate common problematic interactions between humans and wildlife such as wildlife feeding, vehicle collisions, and direct conflicts such as damage management (Awasthy et al., 2012; Hobbs & White, 2016; Hunold, 2020).

An additional emergent topic within our broad review was social science. Social sciences represented 10.7% of all urban wildlife publications between 2010-2020. Papers within this topic include research on urban residents' perceptions of nature (Jacobs et al., 2012; Wieczorek, 2012), surveys on public opinions regarding wildlife management (Jacobs et al., 2014; van Eden et al., 2019), and how social and socioeconomic identities play a role in acceptance or rejection of wildlife management practices (Gledhill & James, 2012; Farmer et al., 2013; Palamar et al., 2013). These topics align with a growing focus on the importance of understanding coupled human-natural systems and urban socio-ecological systems, including linking biodiversity to historical urban development and social inequalities (Schell et al., 2020; Liu et al., 2007). Additionally, my assessment of applied versus fundamental research reveals that most urban wildlife research conducted within the last decade is applied, likely due to the proximity of which humans and wildlife live in cities and the need to develop strategies to coexist with wildlife. Humans and wildlife share the same habitat. Thus, management decisions regarding wildlife have a direct impact on humans. Therefore, our understanding of these ripple effects has begun gaining prominence within literature. Fundamental research on urban wildlife may be less prevalent due to many urban species being inherently common species and thus we already have a strong understanding of the life history and biology of these animals. Understanding the interconnected relationships between people and wildlife will be central to creating spaces where wildlife and people can peacefully coexist (Liu et al., 2007).

I also found shortcomings where urban wildlife research has not improved over the last decade. Academia continues to lead in urban wildlife research publications. This is due to major funding sources in developed countries, such as governmental funding, not allocating substantial funds toward urban wildlife research. Despite most of the human population residing in cities, research funding is still predominantly funneled into rural ecosystem projects (Adams, 2005). Additionally, North America, Europe, and Australia continue to lead publications in urban wildlife studies. This leaves a significant knowledge gap in urban wildlife studies in Asian, South American, and African countries – all rapidly urbanizing continents (UN, 2018). Complex dynamics of politics, economics, and inequality lead to a lack of overall urban ecology research within countries on these continents (Freire, 2006). Many of these areas contain unique and biodiverse species. For example, Sub-Saharan Africa is cited as the most rapidly urbanizing area of the globe, with 40% of land classified as urban in 2015. This area is also home to irreplaceable bird biodiversity to which the region has been designated a global conservation priority (Brooks et al., 2006; DiMarco et al., 2016). The rate of urbanization in these data deficient regions, coupled with the number of endemic species, reveals a critical knowledge gap. Expanding research funding in these biodiverse areas would significantly aid global wildlife conservation, especially in growing international cities.

Finally, regarding specific taxa, I found that herptiles, arthropods, and fish remained the least studied taxa groups over the last two decades. While studies of urban mammals and birds are invaluable to conservation, increasing research on other taxa groups will be crucial for future biodiversity conservation. According to the International Union for Conservation of Nature (IUCN), 39.2% of all known amphibian species, 23.9% of all known reptile species, and 27.0% of all known arthropod species are considered ‘vulnerable’ to extinction (IUCN, 2020).

Urbanization and housing development are the number one cause of concern for population declines across all three of these taxa (IUCN, 2020). Focusing research on the ecology of these taxa in urban areas will contribute to global conservation efforts. Fish species also continued to be overlooked within the field of urban wildlife research (Fig. 3). Freshwater fish are often used as bioindicators of water quality, stream health, and early indicators of possible chemical contamination in freshwater resources (Requea et al., 2017). Therefore, focused research and monitoring of urban fish populations could provide insight into effective water management in urban ecosystems. It is unclear why there is such an overall lack of herptile, arthropod, and fish research within urban systems. Larger and more charismatic wildlife that are often associated with human-wildlife conflict receive the bulk of funding and research resources (Brooke et al., 2014). However, expanding the taxa studied to encompass a broader range of species can assist our overall understanding of how species interact in urban settings, as well as how urban ecosystems function overall.

Conclusion

Urban wildlife management and conservation remains a young field of research, and our results highlight a continued steady increase in urban wildlife research, as well as new emerging topics. However, significant knowledge gaps can still be found. The field would benefit from more studies of herpetofauna, arthropods and fish, and there is still a need for increased urban wildlife research in the rapidly urbanizing global South. I found that academics continue to make up a substantial portion of first authors within literature, leaving a significant gap of valuable research contributions from other career fields – particularly government agencies and NGO's.

Although I report remaining research gaps, I also found advancements in urban wildlife research. I identified social sciences and disease ecology as emerging priority topics, and papers on new research methodologies, particularly in educational research. Urban wildlife management and conservation will benefit from continuing to expand the breadth of interdisciplinary research topics and including more topics outside of the natural sciences, such as sociology, education, outreach, urban planning, policy, and economics.

As urbanization continues to expand across the globe, urban wildlife ecology remains a pertinent and growing field of study within the sciences. As we begin to better understand how to manage and conserve biodiversity within cities, new questions will continue to emerge. Making room for interdisciplinary and diverse players within the field will help solve global conservation issues. A continued expansion of urban wildlife research will allow for more resilient urban ecosystems making cities more livable for both humans and wildlife.

Literature Cited

Adams, L. W. (2005). Urban wildlife ecology and conservation: a brief history of the discipline. *Urban Ecosystems*, *8*(2), 139-156.

Ackley, J. W., Wu, J., Angilletta Jr, M. J., Myint, S. W., & Sullivan, B. (2015). Rich lizards: how affluence and land cover influence the diversity and abundance of desert reptiles persisting in an urban landscape. *Biological Conservation, 182*, 87-92.

Apfelbeck, B., Snep, R.P., Hauck, T.E., Ferguson, J., Holy, M., Jakoby, C., MacIvor, J.S., Schär, L., Taylor, M. & Weisser, W.W. (2020). Designing wildlifeinclusive cities that support human-animal co-existence. *Landscape and Urban Planning, 200*.

Aronson, M.F., Lepczyk, C.A., Evans, K.L., Goddard, M.A., Lerman, S.B., MacIvor, J.S., Nilon, C.H. & Vargo, T. (2017). Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment, 15(4).*

Awasthy, M., Popovic, A. Z., & Linklater, W. L. (2012). Experience in local urban wildlife research enhances a conservation education programme with school children. Pacific *Conservation Biology, 18(1),* 41-46.

Barredo, J. I., Demicheli, L., Lavalle, C., Kasanko, M., & McCormick, N. (2004). Modeling future urban scenarios in developing countries: an application case study in Lagos, Nigeria. *Environment and Planning B: Planning and Design*, *31*(1), 65-84.

Belaire, J. A., Westphal, L. M., & Minor, E. S. (2016). Different social drivers, including perceptions of urban wildlife, explain the ecological resources in residential landscapes. *Landscape Ecology*, *31*(2), 401-413.

Brooke, Z. M., Bielby, J., Nambiar, K., & Carbone, C. (2014). Correlates of research effort in carnivores: body size, range size and diet matter. *PloS One*, *9*(4), e93195.

Brooks, T.M., Mittermeier, R.A., da Fonseca, G.A., Gerlach, J., Hoffmann, M., Lamoreux, J.F., Mittermeier, C.G., Pilgrim, J.D. & Rodrigues, A.S. (2006). Global biodiversity conservation priorities. *Science, 313*.

Dearborn, D. C., & Kark, S. (2010). Motivations for conserving urban biodiversity. *Conservation Biology, 24(2),* 432-440.

Destoumieux-Garzón, D., Mavingui, P., Boëtsch, G., Boissier, J., Darriet, F., Duboz, P., Fritsch, C., Giraudoux, P., Le Roux, F., Morand, S. & Paillard, C. (2018). The one health concept: 10 years old and a long road ahead. *Frontiers in Veterinary Science, (5),* 14.

DiMarco, M., Brooks, T., Cuttelod, A., Fishpool, L.D., Rondinini, C., Smith, R.J., Bennun, L., Butchart, S.H., Ferrier, S., Foppen, R.P. and Joppa, L. (2016). Quantifying the relative irreplaceability of important bird and biodiversity areas. *Conservation Biology, 30*(2).

Farmer, M. C., Wallace, M. C., & Shiroya, M. (2013). Bird diversity indicates ecological value in urban home prices. *Urban Ecosystems*, *16*(1), 131-144.

Freire, M. (2006). Urban planning: Challenges in developing countries. In *International congress on human development,* Madrid.

Gallo, T., Fidino, M., Lehrer, E. W., & Magle, S. B. (2017). Mammal diversity and metacommunity dynamics in urban green spaces: implications for urban wildlife conservation. *Ecological Applications*, *27*(8), 2330-2341.

Gledhill, D. G., & James, P. (2012). Socio-economic variables as indicators of pond conservation value in an urban landscape. *Urban Ecosystems, 15*(4), 849-861.

Grimm, N. B., Grove, J. G., Pickett, S. T., & Redman, C. L. (2000). Integrated approaches to long-term studies of urban ecological systems: Urban ecological systems present multiple challenges to ecologists—pervasive human impact and extreme heterogeneity of cities, and the need to integrate social and ecological approaches, concepts, and theory. *BioScience*, *50(7*), 571-584.

Himsworth, C. G., Jardine, C. M., Parsons, K. L., Feng, A. Y., & Patrick, D. M. (2014). The characteristics of wild rat (Rattus spp.) populations from an inner-city neighborhood with a focus on factors critical to the understanding of rat-associated zoonoses. *PLoS One, 9(3),* e91654.

Hobbs, S. J., & White, P. C. (2016). Achieving positive social outcomes through participatory urban wildlife conservation projects. *Wildlife Research*, *42*(7), 607-617.

Hunold, C. (2020). Urban Greening and Human-Wildlife Relations in Philadelphia: From Animal Control to Multispecies Coexistence? *Environmental Values, 29(1),* 67-87.

IUCN 2020. The IUCN Red List of Threatened Species. Version 2020-2. https://www.iucnredlist.org. Downloaded on 09 July 2020.

Jacobs, M. H., Vaske, J. J., & Roemer, J. M. (2012). Toward a mental systems approach to human relationships with wildlife: The role of emotional dispositions. *Human Dimensions of Wildlife*, *17*(1), 4-15.

Jacobs, M. H., Vaske, J. J., & Sijtsma, M. T. (2014). Predictive potential of wildlife value orientations for acceptability of management interventions. *Journal for Nature Conservation*, *22*(4), 377-383.

Kim, H. Y. (2017). Statistical notes for clinical researchers: Chi-squared test and Fisher's exact test. *Restorative Dentistry & Endodontics*, *42*(2), 152.

Larson, L. R., Cooper, C. B., & Hauber, M. E. (2016). Emotions as drivers of wildlife stewardship behavior: Examining citizen science nest monitors' responses to invasive house sparrows. *Human Dimensions of Wildlife, 21(1),* 18-33.

Liu, J., Dietz, T., Carpenter, S.R., Alberti, M., Folke, C., Moran, E., Pell, A.N., Deadman, P., Kratz, T., Lubchenco, J. & Ostrom, E. (2007). Complexity of coupled human and natural systems. *Science, 317(5844),* 1513-1516.

Łopucki, R., & Kitowski, I. (2017). How small cities affect the biodiversity of ground-dwelling mammals and the relevance of this knowledge in planning urban land expansion in terms of urban wildlife. *Urban Ecosystems, 20*(4), 933-943.

Mackenzie, J. S., & Smith, D. W. (2020). COVID-19—A Novel Zoonotic Disease: A Review of the Disease, the Virus, and Public Health Measures. *Asia Pacific Journal of Public Health*, *32*(4), 145-153.

Magle, S. B., Hunt, V. M., Vernon, M., & Crooks, K. R. (2012). Urban wildlife research: past, present, and future. *Biological Conservation, 155*, 23-32.

Magle, S. B., Lehrer, E. W., & Fidino, M. (2016). Urban mesopredator distribution: examining the relative effects of landscape and socioeconomic factors. *Animal Conservation, 19(2),* 163-175.

Magle, S.B., Fidino, M., Lehrer, E.W., Gallo, T., Mulligan, M.P., Ríos, M.J., Ahlers, A.A., Angstmann, J., Belaire, A., Dugelby, B. and Gramza, A., (2019). Advancing urban wildlife research through a multi-city collaboration. *Frontiers in Ecology and the Environment, 17*(4).

McCance, E. C., Decker, D. J., Colturi, A. M., Baydack, R. K., Siemer, W. F., Curtis, P. D., & Eason, T. (2017). Importance of urban wildlife management in the United States and Canada. *Mammal Study, 42(1),* 1-16.

McKinney, M. L. (2002). Urbanization, Biodiversity, and Conservation: The impacts of urbanization on native species are poorly studied but educating a highly urbanized human population about these impacts can greatly improve species conservation in all ecosystems. *Bioscience, 52(10),* 883-890.

Palamar, M. B., Peterson, M. N., Deperno, C. S., & Correa, M. T. (2013). Assessing rabies knowledge and perceptions among ethnic minorities in Greensboro, North Carolina. *The Journal of Wildlife Management*, *77*(7), 1321-1326.

Patterson, L., Kalle, R., & Downs, C. (2017). A citizen science survey: perceptions and attitudes of urban residents towards vervet monkeys. *Urban Ecosystems*, *20*(3), 617-628.

Rastandeh, A., Zari, M. P., & Brown, D. K. (2018). Components of landscape pattern and urban biodiversity in an era of climate change: a global survey of expert knowledge. *Urban Ecosystems, 21(5),* 903-920.

Santiago-Alarcon, D., & MacGregor-Fors, I. (2020). Cities and pandemics: urban areas are ground zero for the transmission of emerging human infectious diseases. *Journal of Urban Ecology*, *6*(1).

Schell, C. J., Dyson, K., Fuentes, T. L., Des Roches, S., Harris, N. C., Miller, D. S., Woelfle-Erskine, C.A. & Lambert, M. R. (2020). The ecological and evolutionary consequences of systemic racism in urban environments. *Science, 369*(6510).

Sexton, N. R., Ross-Winslow, D., Pradines, M., & Dietsch, A. M. (2015). The Urban Wildlife Conservation Program: Building a Broader Conservation Community. *Cities and the Environment (CATE), 8(1),* 3.

United Nations, Department of Economic and Social Affairs, Population Division (2018). World Urbanization Prospects: The 2018 Revision (ST/ESA/SER.A/420). New York: United Nations.

van Eeden, L. M., Newsome, T. M., Crowther, M. S., Dickman, C. R., & Bruskotter, J. (2019). Social identity shapes support for management of wildlife and pests. *Biological Conservation*, *231*, 167-173.

Wieczorek Hudenko, H. (2012). Exploring the influence of emotion on human decision making in human–wildlife conflict. *Human Dimensions of Wildlife, 17(1),* 16-28.

Wine, S., Gagné, S. A., & Meentemeyer, R. K. (2015). Understanding human–coyote encounters in urban ecosystems using citizen science data: what do socioeconomics tell us? *Environmental Management, 55(1),* 159-170.

Chapter 3: Returning neighbors: Elevation and distance to roadways influence Eastern Wild Turkey (*Meleagris gallopavo silvestris*) occupancy in an urban landscape.

Introduction

Most of Earth's human population lives in cities, with an estimated seven billion people predicted to inhabit cities by 2050 (United Nations 2018). While urbanization has contributed to biodiversity loss worldwide, urban areas harbor species of conservation concern or provide novel habitats for species range expansions (Luna et al. 2018; Hansen et al. 2020). For example, Johnston et al. (2019) found that urban Chicago, Illinois (USA) provides important sources of milkweed rich habitat for migratory monarch butterflies (*Danaus plexippus*), and major cities like New York City, New York (USA) provide novel, but productive breeding habitat for peregrine falcons (*Falco peregrinus*) (Luniak 2004). Thus, understanding native species' capacity to persist in urban ecosystems is crucial for conservation and management initiatives (McCance et al. 2017).

Eastern wild turkey (*Meleagris gallopavo silvestris;* hereafter, "turkey") is a widespread Galliformes species native to Eastern and Central North America (Kark et al. 2007; Dickson 1992). Turkeys were once extirpated from much of their historic range during the early 20th century due to overharvest and habitat destruction (Cardoza 1993; Dickson 1992; Kennamer 1992). After successful reintroduction efforts and subsequent population rebounds, wild turkey is among the great conservation success stories in the U.S. (Cardoza 1993; Ogden 2015). However, within the last decade research has documented wild turkey populations declining by as much as 13%, and lack of occurrence and abundance data from state managers has made this decline difficult to track (Chamberlain et al., 2022). Despite recent declines, growing populations of wild turkeys have been documented throughout major cities of the Eastern and Midwest US over the last twenty years (Niedzielski and Bowman 2016). This increase in urban turkey populations has led to human-wildlife conflicts, such as property damage, traffic hazards, and aggressive behavior including highly publicized accounts in local and national media outlets (e.g., see New York Times 2020; Miller 2018). However, wild turkeys play important ecological roles as dispersers of seeds, controlling pest insect populations (Wood et al. 2018), and they are economically important (Otieno and Frenette 2017; Chapagain et al., 2020) as game species (in non-urban areas). Given the recent documented declines in wild turkey populations in rural systems, but increased presence in urban systems, it is critical to understand how turkeys are using the urban landscape to better inform management and conservation of this species.

Currently, most – if not all – natural history information on wild turkeys comes from studies in rural landscapes. Due to a wide variety of plants and insects in wild turkey diet, and the plasticity of wild turkey behavior to adapt to changing landscapes, turkeys are a habitat generalist (Vander Haegen et al. 1989; Dickson 1992; Badyaev 1995). From rural studies we know that turkeys frequently use open fields and heavy canopy forest (Holbrook et al. 1987; Thogmartin 1999). Open fields are used for foraging, loafing, and socializing during the day, while forested areas with mature trees provide cover to roost at night (Cardoza 1993; Kilpatrick 1988).

These edge habitats also provide low-lying vegetation to hide nesting sites from predators and allow easy access to food resources for hens and poults (Thogmartin 1999; Porter 1978). Further, research in rural landscapes has suggested that proximity of roost sites to water is a critical component, as turkeys seek water during winter months due to low moisture content in food resources (Kilpatrick 1988; Wheeler 1948). Yet, even for species that are considered generalists, studies have shown that habitat use, breeding, and behaviors of animals in urban ecosystems can be profoundly different compared to more natural ecosystems (Mennechez and Clergeua 2006; Delaney et al. 2010; Gallo et al., 2022).

Urban areas are highly fragmented and have higher rates of disturbance, creating greater heterogeneity in land cover types (Angel et al. 2012). This heterogeneity creates a more diverse mosaic of habitat types compared to rural environments. Habitat patches in cities are often smaller, more disjunct (Gibb and Hochuli 2002), and embedded in a matrix of urbanization that is sometimes considered unhospitable to wildlife (Delaney et al. 2010). Smaller and disjunct habitat patches create hard habitat edges and more edge habitat in general. Urban areas are also created by and dominated by people and thus have greater human activity throughout the landscape at all times of the day (Santini et al. 2019). Therefore, our current understanding of wild turkey habitat use may not generalize to populations living in urban environments (Rodewald et al. 2011; Santini et al. 2019). Thus, crucial knowledge gaps about turkey habitat use remain that could prove important for turkey management as turkey populations increase in cities.

Here, I used remotely triggered trail cameras to collect data on wild turkey presence in the Washington, D.C. metropolitan region. The goal of this project was to assess the distribution of wild turkeys in the Washington D.C. region and identify important landscape characteristics that correlate with turkey presence. Based on rural studies, I hypothesized that turkey occupancy would be positively correlated with habitat heterogeneity, tree canopy height, tree canopy cover, and distance to roads (Table 1). I also hypothesized that turkey occupancy would be negatively correlated with human population density, elevation, distance to trails, and distance to water (Table 1). Our findings provide added information about the habitat use of urban turkeys that can help inform the management, conservation, and co-existence of wild turkeys in cities.

Table 1. Descriptive information about the predictor variables used to estimate occupancy and detection probability of wild turkey in the Washington, D.C. metropolitan region and accompanying hypotheses of why each variable was selected.

Variable	Unit	Mean (Range)	Description	Hypothesis
Tree Cover	Proportion	4.77 (1.1-8.0)	Proportion of tree cover within a 2-km fixed radius buffer around each site	Turkeys are known to use heavily forested rural systems (Holbrook et al. 1987; Thogmartin 1999) therefore, canopy cover will be positively correlated with turkey occupancy.
Veg	Proportion	2.3 (1-3.5)	Proportion of vegetation (low vegetation combined with shrubland) within a 2-km fixed radius buffer around each site	Low vegetation/ground cover has been correlated with habitat use due to the important of these variables for nesting and providing vegetative food resources (Chamberlain 2020) therefore turkey occupancy will be positively correlated with low veg.
Distance to water	Meters	591 (7.7-2057)	Distance to the nearest raster cell categorized as water	Distance to open water (in riparian areas) has been recorded as commonly used habitat (Perlichek et al., 2009; Wheeler 1948), therefore turkey occupancy will be negatively correlated with distance to water.

Distance to road	Meters	102 (0.07-300)	Distance to nearest raster cell categorized as a roadway	Turkeys are known to have keen hearing and eyesight (Thogmartin 1999), traffic noise might hinder a turkey’s ability to communicate or locate predators, therefore turkey occupancy will be negatively correlated with distance to roadways.
Elevation	Meters	55 (3.1-109)	Mean landscape elevation within a 2km fixed radius buffer around each site	Low elevation floodplains in our study could provide nutrient rich food resources for turkeys, while also allowing easy access to forest cover required for night roosting (Perlichek et al.,2009) thus turkey occupancy will be negatively correlated with elevation.
Canopy Height	Meters	9.3 (0.5-19.9)	Mean canopy height within a 2-km fixed radius buffer around each site	Turkeys select the tallest available roost trees (Perlicher et al., 2009; Kilpatrick et al., 1988). Given the lack of mature forest options in our urban study area, I hypothesize turkeys will choose areas with the tallest trees, thus canopy height will be positively correlated with occupancy.
Log Housing Density	Number of housing units per sq. km	7.1 (5.5-8.6)	Mean number of human housing units within a 2-km fixed radius buffer around each site transformed to log number	Areas of high human development have been categorized as poor-quality habitat for wild turkeys in the past (Gustafson et al., 1994), therefore I hypothesize turkey occupancy will be negatively correlated with human housing density.

Habitat heterogeneity	Proportion	0.9 (0.7-0.9)	Landscape entropy within a 2-km fixed radius buffer around each site	Turkeys have been known to utilize mixed open/forested habitats (Niedzielski & Bowman 2016; Wright and Speake 1976; Donohoe & Mckibben 1970) therefore I hypothesize turkey occupancy will be positively correlated with entropy.
Distance to trails	Meters	781 (14.1-4549)	Distance to nearest feature labeled as a foot or bike path within a 2-km fixed radius buffer around each site, excluding sidewalks and roadway bike lanes	Areas of high human development have been categorized as poor-quality habitat for wild turkeys (Gustafson et al., 1994), therefore I hypothesize turkey occupancy will be negatively correlated with distance to trails due to consistent human use.
Precipitation	Centimeters	0.25 (0-2.9)	Average daily precipitation in any form recorded for the sampling period	Flooding has been known to impact turkey habitat use (Chamberlain 2013) and therefore may hamper our chances of detecting turkey.
Temperature	Centimeters	80 (29-90)	Average daily temperature recorded for the sampling period	Turkeys have been known to change activity patterns by season due to food availability (Niedzielski & Bowman 2016), therefore I chose temperature as a determinant in detecting turkeys on our cameras.

Season	Days	28.7 (0-51)	Number of days a camera was active at a site per season, across six sampling seasons	Increasing the number of sampling days will increase the probability of detection.

Methods

Study Area

This study was conducted within the Washington, D.C. metropolitan area, which includes Washington D.C., Prince George's and Montgomery Counties, Maryland, and the City of Alexandria, Arlington County, and Fairfax County, Virginia (Fig. 1). Washington, D.C. is the sixth largest metropolitan area within the U.S. (Smega et al. 2020) but maintains the highest proportion of parkland of any major U.S. city at 21.9% of total land cover (Cohen et al. 2017). My study area sits on the ancestral homeland of the Nacochtank (also called Anacostan) and Piscataway people (Tayac 2009). The climate is considered humid subtropical, experiencing all four seasons, and averaging 112 cm of precipitation annually (NOAA 2022). The Washington D.C. region contains a geological fall line, separating the city and surrounding lands into two distinct ecoregions, the Appalachian Piedmont region to the west, and the Mid-Atlantic coastal plain to the east (District of Columbia Department of Energy and Environment, 2015). This split diversifies the region's habitats including woody wetlands, coastal plain swamps, upland floodplain forests, ruderal grasslands, and open riverine habitats (District of Columbia Department of Energy and Environment, 2015).

Site Selection and Data Collection

An initial spatial grid of points was overlayed on our study area with each grid point 2-km apart. To identify sampling sites, seventy-five points were randomly chosen from the initial grid and the nearest greenspace to each selected point was identified as a sampling location (Fig. 1). Site selection was conducted using the sf (Pebesma 2018) and raster (Hijmans 2015) packages in R ver. 3.6.1 (R Core Team 2022). Greenspace types include public parks (n = 69), golf courses (n = 3), and cemeteries (n = 3). All public parks within this study area are utilized daily by people, many include or are adjacent to playgrounds, parking lots, biking trails, public restrooms, and athletic recreation areas. Within each selected greenspace, sampling sites were established at a location that maximized detection probability of wildlife species (e.g., heavily vegetated areas, animal trails, gravel roads, fence lines, etc.).

At each sampling site, I deployed one unbaited remotely triggered trail camera for approximately 30 days four times per year (January, April, July, and October) from July 2020-December 2021. Cameras were active an average of 28.7 days per sampling season. Three models of trail camera were used, Reconyx Hyperfire 2 (Reconyx, Holmen, WI, USA), Bushnell Trophy Cam HD Aggressor, and Bushnell Trophy Cam HD (Bushnell Corporation, Overland Park, KS, USA). Cameras were placed on trees approximately 1.5 m from the ground using nylon straps and were deployed at the same location each sampling season. Cameras were set to take one photo every trigger with a 15 second rest period between triggers. Photos were uploaded to the Urban Wildlife Information Network online database (Magle et al. 2019) and animals in photos were identified to the species level. Each photo was classified by two users, and a third independent user validated any discrepancies. For this analysis, only wild turkey observations were used.

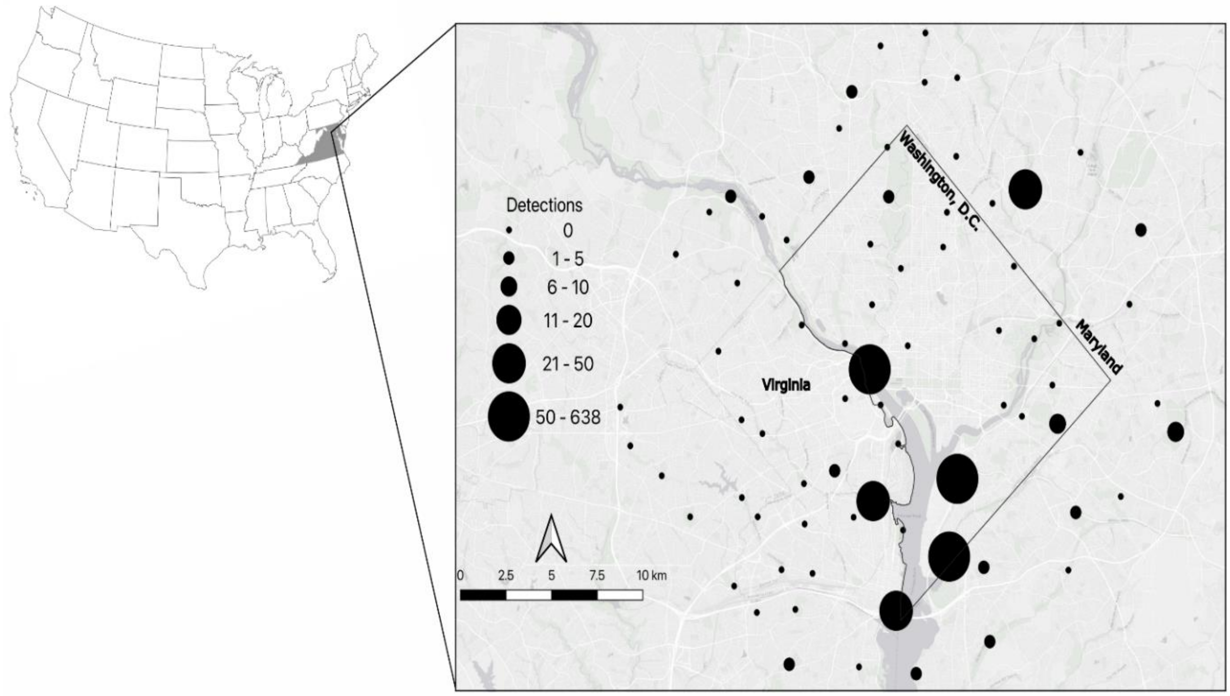

Figure 1. Seventy-five camera sites, denoted by black dots across the Washington, D.C., metropolitan region. Larger circles indicate areas where turkey detections were high, and smaller circles indicate sites with few to zero turkey detections.

Predictor Variables and Occupancy Model

Using a 2-km buffer around each sampling site I extracted the proportion of each: tree canopy cover (*TreeCover*), impervious surface (*Impervious*), low vegetation (*Veg*), and wetland (*Wetland*) land cover using the Chesapeake Bay Conservancy Land Cover 1-m resolution data set (Robinson et al. 2019; Table 1). A 2-km buffer was chosen to account for the estimated average home range of wild turkey near our study area (across winter/spring/summer seasons; Niedzielski and Bowman 2016; Holbrook et al., 1987).

Using the same land cover data, I calculated the distance from each sampling site to the nearest raster cell categorized as "water" (*dist2water*) and "road" (*dist2road*). Distance to nearest "hike and bike trail" (*dist2trails*) was calculated using the Open Street Map roadway layer (OpenStreetMap contributors 2022; Table 1). Here, I considered a trail to be any feature in the data layer categorized as a cycleway or footway and did not include bike lanes or sidewalk footpaths. Distance to hike and bike trails was chosen as an independent variable, because at the time of this study, high profile human-turkey interactions had taken place on hike and bike trails in our study area.

Mean elevation (*Elev*) within each 2-km buffer was extracted from a 1/3-arc second digital elevation model raster (United States Geological Service 2016), and I used the Global Ecosystem Dynamics Investigation (GEDI) LiDAR data set (Potapov et al. 2020) to extract average canopy height (*CanopyH*) within each 2-km buffer as tree height has been known to be an important habitat variable for turkey roost sites (Yarrow and Yarrow 1999). Mean human housing density (*log_hdens*) within each 2-km buffer was calculated using the U.S. Census Bureau (2010) data set (Radeloff et al. 2018; Table 1). The above analyses were conducted using the sf, terra (Hijimans et al. 2022), and raster packages in R.

To calculate a metric of habitat heterogeneity I used landscapemetrics (Hesselbarth et al. 2019) package in R (R Core Team 2022) and calculated landscape entropy (Nowosad and Stepinski 2019). Here I used only the tree cover and low vegetation land cover classifications to calculate a metric of heterogeneity between the two habitat types (*Entropy*). I also included season of sampling (*Season*) as a categorical variable within our models to account for the lack of closure between sampling seasons (Table 1).

Regarding detection probability, I hypothesized that the number of days a camera was active at a site during each sampling season (*CamDays*), average daily temperature (*Temp*), and average daily precipitation (*Precip*) would have an influence on the detection probability of wild turkeys. Temperature and precipitation data were obtained from the monitoring station at Reagan National Airport (National Oceanic and Atmospheric Administration 2020; Table 1) for each sampling day.

I fit single season occupancy models using the unmarked package (Fiske and Chandler 2011) in R to estimate the probability of occupancy (ψ) and probability of detection (p) of turkeys in our study area. Occupancy models are used to estimate the probability of occurrence of an organism while considering imperfect detection (MacKenzie et al., 2002). I used the dredge function (Barton and Barton 2011) in R to fit all combinations of predictor variables. Prior to developing the model set, variables were evaluated for multicollinearity using Pearson's Correlation Coefficient and variable pairs that had an $|r| > 0.70$ were not included in the same model (Ratner 2009; Boslaugh & Watters 2008). Canopy height and landscape entropy were correlated ($r = -0.72$), and therefore, were never included together in the same model. All covariate values were standardized by mean centering. Akaike information criterion corrected for small sample sizes (AICc) was used to compare models, and models within two ΔAICc values of the model with the lowest AICc value were considered top models (Anderson et al. 1994).

Results

From July 2020-Aug 2021 turkeys (including adults and poults) were detected 1575 times at 19 of 75 sampling sites ($\bar{x}$ = 315 detections per sampling season; Fig.

1). Of the nineteen sites where turkeys were detected, eighteen sites were in public parks and one site was a privately owned golf course. Turkeys were detected the most in winter 2021 (n = 572) followed by summer 2021 (n = 385) and then fall 2020 (n = 315), spring 2021 (n = 303), and summer 2020 (n = 0).

Model Results

Fitting all combinations of predictor variables resulted in a model set of 4065 models. Of these, twenty-five were within two ΔAICc of the top model (Table 2). The top model included mean canopy height, distance to roadway, and mean elevation as predictor variables on occupancy and the intercept only model on detection (Table 2). Regarding detection probability, the number of days cameras were active and average daily temperature (Table 2) were also present in our top model set, but the 95% confidence intervals of each model coefficient overlapped zero in all the models for which these variables were retained.

For the occupancy variables, elevation, entropy, canopy height, tree cover, distance to water, distance to road, and housing density were all retained in our top model set but were never significant (Table 2). Only distance to road and elevation had confidence intervals that did not overlap zero (Table 3), and both were found in our top model (Table 2). I chose not to conduct model averaging (Banner and Higgs 2017: Cade 2015) and instead only interpret the model coefficients and 95% confidence intervals for elevation and distance to road from our top model (Table 2).

Elevation was negatively correlated with turkey occupancy ((β = -0.000494, 95% CI= -0.038 – -0.10; Fig. 2A), and turkey were 48% more likely (odds ratio = 0.48) to use habitats that were at 1 sd lower in elevation than the mean elevation of our study area (54.9-25.8 m). I also found that turkey occupancy and distance to road were positively correlated (β = 0.000350, 95% CI= 0.003 – 0.012; Fig. 2B), and turkey were 77% more likely (odds ratio = 1.77) to use habitats that were 1 sd further from roads compared to the mean distance to roads (101.8–172.7 m).

Table 2. All models that were within two ΔAICc of the top model used to estimate turkey occupancy and detection in the Washington, D.C. metropolitan area.

Model	k	ΔAIC_c	AIC_c Wt	Cuml. Wt
Ψ (CanopyH+Dist2Road+ Elev) p(.)	5	0	0.0142	0.014209
Ψ (CanopyH+Dist2Road+ Elev) p(Temp)	6	0.074	0.0136	0.027897
Ψ (Dist2Road+ Elev) p(.)	4	0.447	0.0113	0.025048
Ψ(CanopyH+Dist2Road+Elev) p (Camdays +Temp)	7	0.492	0.0111	0.022469
Ψ (Dist2Road +Elev) p(Temp)	5	0.508	0.0110	0.022128
Ψ(CanopyH+Dist2Road+Elev) p(Camdays)	6	0.745	0.0097	0.020806
Ψ (CanopyH+Dist2Road+Dist2Water+ Elev) p(.)	6	0.836	0.0094	0.019139
Ψ(Dist2Road+Dist2Water+Elev) p(.)	5	0.842	0.0093	0.018677
Ψ(Dist2Road+Elev) p (Temp +Camdays)	6	0.904	0.0090	0.018363
Ψ(Dist2Road+Dist2Water+Elev) p(Temp)	6	0.908	0.0090	0.018059
Ψ(CanopyH+Dist2Road+Dist2Water+Elev) p(Temp)	7	0.916	0.0089	0.018005
Ψ(Dist2Road+Elev) p(CamDays)	5	1.173	0.0079	0.016885
Ψ (Dist2Road+ Dist2Water+ Elev) p (CamDays +Temp)	7	1.309	0.0073	0.015283
Ψ(CanopyH+Dist2Road+Dist2Water+Elev) p (CamDays + Temp)	8	1.341	0.0072	0.014649
Ψ(CanopyH+Dist2Road+Elev+HousingDens) p(.)	6	1.568	0.0066	0.013752
Ψ(Dist2Road+Dist2Water+Elev) p(Camdays)	6	1.578	0.0065	0.012939
Ψ(CanopyH+Dist2Road+Dist2Water+Elev) p(Camdays)	7	1.591	0.0064	0.012865
Ψ (CanopyH+Dist2Road+ Elev + Season) p(.)	10	1.644	0.0063	0.012654
Ψ (CanopyH+Dist2Road+ Elev + HousingDens) p(Temp)	7	1.656	0.0062	0.01245

Ψ (Dist2Road+Dist2Water+ Elev + Entropy) p(.)	6	1.692	0.0061	0.012302
Ψ (Dist2Road+ Elev + Entropy) p(.)	5	1.171	0.0060	0.012111
Ψ (Dist2Road+Dist2Water + Elev + Entropy) p(Temp)	7	1.767	0.0058	0.011886
Ψ (Dist2Road + Elev + Entropy) p(Temp)	6	1.790	0.0057	0.011675
Ψ (CanopyH+Dist2Road+ Elev + Season) p(Temp)	11	1.799	0.0056	0.011584
Ψ (CanopyH+Dist2Road+ Elev + TreeCover) p(.)	6	1.957	0.0053	0.011119

Table 3. Model coefficients and 95% confidence intervals of landscape variables retained in the top twenty-five models for turkey occupancy. An * represents confidence intervals that do not overlap zero.

		95% Confidence Intervals	
Variable	Estimate	Lower	Upper
Entropy	-13.976	-22.796	5.156
Canopy Height	-0.069	-0.155	0.016
Elevation*	-0.024	-0.038	-0.010
Tree Cover	-0.050	-0.353	0.252
Distance to Water	-0.0005	-0.001	0.0003
Distance to Road*	0.008	0.003	0.012
Housing Density	0.224	-0.409	0.857

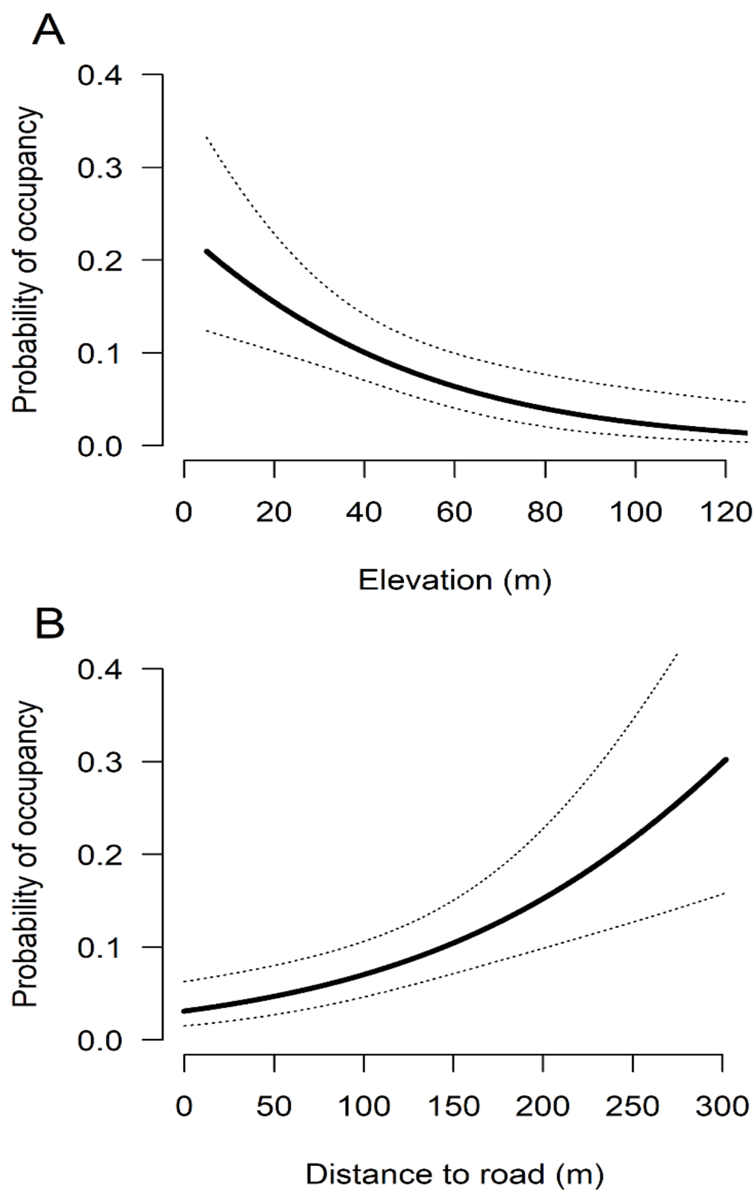

Figure 2. Landscape variables that significantly influenced wild turkey occupancy in our study. A) Predicted relationship between occupancy and elevation and B) the predicted relationship between occupancy and distance to roadways.

Discussion

To my knowledge this is one of the first studies to assess habitat use and occupancy of urban wild turkey (Tinsley 2014). Given that cities are expanding rapidly, these findings provide valuable insight into how wild turkey are using urban habitats. I estimated wild turkey occupancy across the Washington, D.C. region and found that sites further from roadways and at lower elevation had a higher probability of turkey presence. These findings support the hypothesis that turkey occupancy would be positively correlated with distance to roadways and negatively correlated with elevation. However, contrary to my initial hypotheses I did not find a significant positive correlation with canopy height and habitat heterogeneity, nor a significant negative correlation with human population density and distance to water.

Roadway right of ways are known to serve as habitat for wild turkey. Several studies have documented that the areas along roadways provide low vegetation edge habitats that are used by wild turkey for nesting (Lambert 1986) and foraging (Butler et al. 2005). However, in these studies, roadways were gravel or dirt roads with low traffic volume (Lambert 1986; Butler et al. 2005; Delahunt 2011). In this study, I found that sites further from roadways had higher probabilities of urban turkey presence (Fig. 2a). Roadways in the study area were multi-lane highways, primary, or secondary paved roads. Studies have researched the impacts of various road types on turkeys and found turkeys avoid primary and high traffic roads (Carl et al., 2023; Gerrits 2019).

Similarly, traffic noise might deter turkeys from occupying potential habitat near major roadways in our study area. Wild turkeys have keen eyesight and acute hearing (Thogmartin 1999) and may avoid high traffic roads due to traffic volume obstructing communication, predator-avoidance signals, and/or visual cues (Paris and Schneider 2009). Another explanation may be that turkeys are selecting larger habitat patches with less roads and not specifically avoiding roads. I did not include patch size as a variable because defining urban habitat patches is subjective and varies across publications. However, future research should focus on delineating what is an urban habitat patch to have a better understanding of habitat requirements for urban wildlife in general as well as using GPS data to track turkeys to gain an understanding of fine scale habitat selection.

Despite not finding a relationship between occupancy and the nearest land cover cell classified as water, I did find that the probability of turkey occupancy was higher at lower elevation sites. The lower elevations in the Washington D.C. region are along two major urban riverine systems, the Potomac, and the Anacostia rivers, which are surrounded by riparian and floodplain habitats (District of Columbia Department of Energy and Environment, 2015). Rural turkey broods select riparian areas for foraging and loafing since poults need almost constant foraging opportunities to maintain energy for survival (Healy 1985; Chamberlain et al. 2020). Additionally, lower elevation floodplains provide turkey flocks open edge habitats and nutrient rich soil accumulation, which can lead to high productivity rates of preferred plant species (Marks et al. 2020). Low elevation floodplains in our study could provide nutrient rich food resources for turkeys, while also allowing easy access to forest cover required for night roosting (Perlichek et al., 2009). Future landscape conservation and restoration efforts should focus on urban riparian ecosystems to maintain healthy habitat for turkeys and other native species.

Management Implications

More research is needed to better understand turkey-habitat relationships in urban ecosystems. A lack of published findings about urban turkey ecology leaves a substantial knowledge gap for the management and conservation of urban turkey populations. As large charismatic birds, Eastern wild turkeys can serve as a catalyst for understanding how urban wildlife use available habitat, and how urban areas may be providing habitats for animals declining in more natural landscapes. For example, I found a significant positive relationship between turkey occupancy and distance to roadways. Thus, management, restoration, and creation of urban greenspaces should 1) focus on restoring and managing habitat patches further away from roadways, 2) reduce roads or vehicular traffic in urban greenspaces, and/or 3) protect larger tracts of habitat with less roads.

Lower elevation habitats were important for urban turkey occupancy. Restoration efforts should focus on urban floodplain and riparian habitats to maintain healthy turkey habitat. Studies suggest that undeveloped riverine and riparian habitats can provide beneficial refuges from anthropogenic noise and infrastructure development (Donaldson et al. 2007; Crooks et al. 2004) for several species of urban wild birds. Thus, prioritizing low elevation riparian areas can provide conservation benefits across ecological communities.

These findings can also inform management to better position outreach efforts near areas where human-wildlife conflict may arise. For example, in 2021, multiple human-turkey conflicts occurred in urban Washington, D.C. near a hike and bike trail running along the Anacostia River (Washington Post 2022). These results provide additional insight to these human-turkey interactions, as turkeys are likely to occupy these low elevation riparian habitats within my study area. When trails and human access are encouraged in areas that intersect turkey habitat (even if turkeys have not been detected in the area), educational signage should be placed to mitigate human contact with turkeys and educate the public about the natural history of these animals. Data-driven educational campaigns such as these would increase the likelihood of coexistence between humans and urban wildlife.

Conclusion

Given the generalist nature of wild turkey, it is likely that turkey will continue to colonize and persist in urban landscapes. Therefore, it remains important to study how turkey use novel urban habitats. I quantified the relationship between landscape characteristics and turkey occupancy across the Washington, D.C. region. Our findings can aid managers in effectively identifying areas where habitat restoration initiatives can be targeted for conservation, as well as potential hotspots where turkeys could become problematic. A better understanding of how a species – once absent from the landscape – can reclaim urban habitats will lead to cities where humans and wildlife can coexist.

Literature Cited

Anderson, D. R., K. P. Burnham, and G. C. White. AIC model selection in overdispersed capture-recapture data. Ecology 75, no. 6 (1994): 1780-1793.

Angel, S., J. Parent, and D. L. Civco (2012). The fragmentation of urban landscapes: Global evidence of a key attribute of the spatial structure of cities, 1990–2000. Environment and Urbanization, 24(1), 249–283.

Badyaev, A. V. (1995). Nesting habitat and nesting success of eastern wild turkeys in the Arkansas Ozark Highlands. The Condor, 97(1), 221-232.

Banner K.M., Higgs, M.D. (2017). Considerations for assessing model averaging of regression coefficients. Ecological Applications 27(1): 78-93.

Barton, K., and M. K Barton (2019). Package 'mumin.' R package version, 1(6).

Boslaugh, Sarah and Paul Andrew Watters. (2008). Statistics in a Nutshell: A Desktop Quick Reference, ch. 16. Sebastopol, CA: O'Reilly Media. ISBN-13: 978-0596510497.

Butler, M. J., M. C. Wallace, W. B. Ballard, S. J DeMaso, and R. D Applegate (2005). From the field: The relationship of Rio Grande wild turkey distributions to roads. Wildlife Society Bulletin, 33(2), 745–748.

Cade, B. S. (2015). Model averaging and muddled multimodel inferences. *Ecology*, *96*(9), 2370-2382.

Cardoza, J. E. (1993). The Wild Turkey: An Update. 21(5).

Carl, A. R., Londe, D. W., Elmore, R. D., Fuhlendorf, S. D., & Davis, C. A. (2023). Habitat Selection of Female Rio Grande Wild Turkey Influenced by Vegetation and Anthropogenic Structures. Rangeland Ecology & Management, 91, 44-54.

Chamberlain, M. J., Byrne, M. E., Stafford, N. J., Skow, K. L., & Collier, B. A. (2013). Wild turkey movements during flooding after opening of the Morganza Spillway, Louisiana. Southeastern Naturalist, 12(1), 93-98.

Chamberlain, M. J., B. S. Cohen, N. W Bakner, and B. A Collier (2020). Behavior and movement of wild turkey broods. The Journal of Wildlife Management, 84(6), 1139–1152.

Chamberlain, M. J., Hatfield, M., & Collier, B. A. (2022). Status and distribution of wild turkeys in the United States in 2019. Wildlife Society Bulletin, 46(2), e1287.

Chapagain, B. P., N. C. Poudyal, O. Joshi, C. Watkins, and R. D. Applegate. ()2020. Seasonal and Regional Differences in Economic Benefits of Turkey Hunting. Wildlife Society Bulletin 44:271-280

Cohen, D. A., B. Han, K. R. Evenson, C. Nagel, T. L. McKenzie, T. Marsh, S. Williamson, and P. Harnik (2017). The prevalence and use of walking loops in neighborhood parks: A national study. Environmental Health Perspectives, 125(2), 170–174.

Delahunt, K. S. (2011). Nesting ecology, survival, and home range of wild turkeys in an agricultural landscape. Southern Illinois University at Carbondale.

Delaney, K. S., S. P Riley, and R. N. Fisher (2010). A rapid, strong, and convergent genetic response to urban habitat fragmentation in four divergent and widespread vertebrates. Plos One, 5(9), e12767.

Donohoe, R. W., & McKibben, C. (1970). The wild turkey in Ohio (No. 3). Ohio Division of Wildlife.

Fiske, I., and R. Chandler (2011). Unmarked: An R package for fitting hierarchical models of wildlife occurrence and abundance. Journal of statistical software, 43, 1-23.

Gallo, T., Fidino, M., Gerber, B., Ahlers, A. A., Angstmann, J. L., Amaya, M., and Magle, S. B. (2022). Mammals adjust diel activity across gradients of urbanization. Elife, 11, e74756.

Gerrits, Alaina Pamela. (2019). Movement Ecology of Eastern Wild Turkeys (Meleagris gallopavo silvestris) and Turkey Hunters on Public Land in South Carolina, USA. LSU Master's Theses. 5000.

Gibb, H., and D. F. Hochuli (2002). Habitat fragmentation in an urban environment: large and small fragments support different arthropod assemblages. Biological Conservation, 106(1), 91–100.

Gustafson, E. J., Parker, G. R., and Backs, S. E. (1994). Evaluating spatial pattern of wildlife habitat: a case study of the wild turkey (Meleagris gallopavo). American Midland Naturalist, 24-33.

Hansen, C. P., A. W. Parsons, R. Kays, and J.J. Millspaugh (2020). Does Use of Backyard Resources Explain the Abundance of Urban Wildlife? Frontiers in Ecology and Evolution, 8, 570771.

Healy, W. M. (1985). Turkey poult feeding activity, invertebrate abundance, and vegetation structure. The Journal of Wildlife Management, 466–472.

Hesselbarth, M. H., M. Sciaini, K.A. With, K. Wiegand, and J. Nowosad (2019). landscapemetrics: an open-source R tool to calculate landscape metrics. Ecography, 42(10), 1648-1657.

Hijmans, R. J., J. Van Etten, J. Cheng, M. Mattiuzzi, M. Sumner, J.A. Greenberg, and M.R.J. Hijmans (2015). Package 'raster'. R package, 734, 473.

Holbrook, H. T., M.R. Vaughan, and P.T. Bromley (1987). Wild turkey habitat preferences and recruitment in intensively managed Piedmont forests. The Journal of Wildlife Management, 182–187.

Johnston, M. K., E.M. Hasle, K.R. Klinger, M.P. Lambruschi, A. Derby Lewis, D.F. Stotz, A.M. Winter, M.J. Bouman, and I. Redlinski (2019). Estimating Milkweed Abundance in Metropolitan Areas Under Existing and User-Defined Scenarios. Frontiers in Ecology and Evolution, 7. https://www.frontiersin.org/articles/10.3389/fevo.2019.00210

Kark, S., A. Iwaniuk, A. Schalimtzek, and E. Banker (2007). Living in the city: Can anyone become an 'urban exploiter'? Journal of Biogeography, 34(4), 638–651.

Kennamer, J. E., M. Kennamer, and R. Brenneman. (1992). History. Pages 6-17 in J. G. Dickson, editor. The wild turkey: biology and management. Stackpole, Harrisburg, Pennsylvania, USA

Kilpatrick, H.J., T.P. Husband, and C.A. Pringle. (1988). Winter-roost site characteristics of Eastern Wild Turkeys. The Journal of Wildlife Management 52:461-463

Lambert, E. P. (1986). Home range, movements, and habitat use of the eastern wild turkey in commercially managed pine forests of southeast Louisiana. Southeastern Louisiana University.

Luna, Á., P. Romero-Vidal, F. Hiraldo, and J.L. Tella (2018). Cities may save some threatened species but not their ecological functions. PeerJ, 6, e4908.

Luniak, M. (2004). Synurbization–adaptation of animal wildlife to urban development. In Proceedings 4th international urban wildlife symposium (pp. 50-55). Tucson: University of Arizona.

MacKenzie, D. I., J.D. Nichols, G.B. Lachman, S. Droege, J Royle. Andrew, and C.A. Langtimm (2002). Estimating site occupancy rates when detection probabilities are less than one. Ecology, 83(8), 2248–2255.

Magle, S. B., M. Fidino, E.W. Lehrer, T. Gallo, M.P. Mulligan, M.J. Ríos, A.A. Ahlers, J. Angstmann, A. Belaire, and B. Dugelby (2019). Advancing urban wildlife research through a multi-city collaboration. Frontiers in Ecology and the Environment, 17(4), 232–239.

McCance, E. C., D.J. Decker, A.M. Colturi, R.K. Baydack, W.F. Siemer, P.D. Curtis, and T. Eason (2017). Importance of Urban Wildlife Management in the United States and Canada. Mammal Study, 42(1), 1–16.

Mennechez, G., and Clergeau, P. (2006). Effect of urbanisation on habitat generalists: starlings not so flexible? Acta Oecologica, 30(2), 182-191.

Miller, J. E. (2018). United States Department of Agriculture. Wild Turkeys. Wildlife Damage Management Technical Series. 15.

National Oceanic and Atmospheric Administration [NOAA] (2022). Climate data online, daily summaries: Washington 5.1 NW, DC US; Washington 3.1 NNE, DC US; Washington 2.0 SSW, DC US. Available online at: https://www.ncdc.noaa. gov/cdo-web/datasets.

New York Times 2020. The Corner of N.Y.C. that's overrun by deer, turkeys, and feral cats. Retrieved from https://www.nytimes.com/2020/03/06/nyregion/Staten-island-deer-wild-turkeys.html

Niedzielski, B., and J. Bowman (2016). Home range and habitat selection of the female eastern wild turkey at its northern range edge. Wildlife Biology, 22(2), 55–63.

Nowosad, J., and T.F. Stepinski (2019). Information theory as a consistent framework for quantification and classification of landscape patterns. Landscape Ecology, 34, 2091-2101.

Ogden, L. E. (2015). Return of the native. New Scientist, 228(3049), 44–45.

Open Street Map contributors. (2022). OpenStreetMap database [PostgreSQL via API]. OpenStreetMap Foundation: Cambridge, UK; 202. Available under the Open Database Licence from: openstreetmap.org.

Otieno, E. N., and L. Frenette (2017). Stable isotope evidence shows key farmland structure features driving Eastern Wild Turkey food selection. Ornithological Science, 16(2), 121–129.

Pebesma, E. J. (2018). Simple features for R: standardized support for spatial vector data. R J., 10(1), 439.

Perlichek, K. B. (2005). Distribution, delineation, and characteristics of roosting habitat of wild turkey in Trans-Pecos, Texas. Sul Ross State University.

Porter, W. F. (1978). The ecology and behavior of the wild turkey (*Meleagris gallopavo*) in south-eastern Minnesota. Ph.D. Thesis, Univ. Minnesota, Minneapolis.

Potapov, P., X. Li, A. Hernandez-Serna, A. Tyukavina, M.C. Hansen, A. Kommareddy, and M. Hofton (2021). Mapping global forest canopy height through integration of GEDI and Landsat data. Remote Sensing of Environment, 253, 112165.

R Core Team (2022). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Radeloff, V. C., D.P. Helmers, H.A. Kramer, H.H. Mockrin, P.M. Alexandre, A. Bar-Massada, V. Butsic, T.J. Hawbaker, S. Martinuzzi, and A.D. Syphard (2018). Rapid growth of the US wildland-urban interface raises wildfire risk. Proceedings of the National Academy of Sciences, 115(13), 3314–3319.

Ratner, B. (2009). The correlation coefficient: Its values range between +1/−1, or do they? J Target Meas Anal Mark 17, 139–142. https://doi.org/10.1057/jt.2009.5

Robinson, C., L. Hou, K. Malkin, R. Soobitsky, J. Czawlytko, B. Dilkina, and N. Jojic (2019). Large scale high-resolution land cover mapping with multi-resolution data. In Proceedings of the IEEE/CVF Conference on Computer Vision and Pattern Recognition (pp. 12726-12735).

Rodewald, A. D., and L.J. Kearns (2011). Shifts in dominant nest predators along a rural-to-urban landscape gradient. The Condor, 113(4), 899–906.

Santini, L., M. González-Suárez, D. Russo, A. Gonzalez-Voyer, A. von Hardenberg, and L. Ancillotto (2019). One strategy does not fit all: Determinants of urban adaptation in mammals. Ecology Letters, 22(2), 365–376.

Semega, J., M. Kollar, E. Shrider, and J. Creamer (2020). U.S. Census Bureau, Current Population Reports, P60-270 (RV), Income and Poverty in the United States: 2019, U.S. Government Publishing Office, Washington, DC.

Tayac, G. (2009). IndiVisible: African-Native American Lives in the Americas. Smithsonian Books.

Thogmartin, W. E. (1999). Landscape attributes and nest-site selection in wild turkeys. The Auk, 116(4), 912–923.

Tinsley, K. A. (2014). Wild turkeys in the urban matrix: how an introduced species survives-and thrives-in a multifunctional landscape. Dissertation. University of Minnesota.

United Nations, Department of Economic and Social Affairs, Population Division (2018). World Urbanization Prospects: The 2018 Revision (ST/ESA/SER.A/420). United Nations, New York.

United States Geological Survey (2016) 3D Elevation Program 1-Meter Resolution Digital Elevation Model. https://www.usgs.gov/the-national-map-data-delivery

Vander Haegen, W. M., Sayre, M. W., & Dodge, W. E. (1989). Winter use of agricultural habitats by wild turkeys in Massachusetts. The Journal of wildlife management, 30-33.

Washington Post 2022. Wild Turkey attacks people on Anacostia Riverwalk Trail. Retrieved from https://www.washingtonpost.com/dc-md-va/2022/05/19/wild-turkey-dc-anacostia-riverwalk-trail

Wheeler, R. J., Jr. (1948). The wild turkey in Alabama. Department of Conservation, Montgomery, Alabama, USA.

Wood, J. D., B.S. Cohen, T.J. Prebyl, L.M. Conner, B.A. Collier, and M.J. Chamberlain (2018). Time-since-fire and stand seral stage affect habitat selection of eastern wild turkeys in a managed longleaf pine ecosystem. Forest Ecology and Management, 411, 203–212.

Wright, G. A., D. W. Speake. (1976). Compatibility of the eastern wild turkey with the recreational activities at Land Between the Lakes, Kentucky. Proc. Annu. Conf. Southeast. Assoc. Game and Fish Comm. 29:578-584.

Yarrow, G.K. and D.T. Yarrow (1999). Managing Wildlife. Sweetwater Press. Birmingham, Alabama.

Chapter 4: How the Past Shapes the Present: Understanding how historic housing segregation has shaped mammal communities in the Nation's Capital

Introduction

Historic structural racism has shaped the economic, social, and ecological functions of cities in the United States (Rothstein, 2017). Where communities historically formed in cities was not decided by the individuals themselves, but rather by discriminatory land use and housing policies enacted at the federal, state, and private levels (Markley 2024). These decisions resulted in a mapping system known as "redlining," which codified the segregation of communities of color to resource impoverished areas (Rothstein, 2017). Despite discriminatory housing segregation being outlawed by the Fair Housing Act in 1968, the practice of redlining caused long-term segregation, creating a significant lack of generational wealth in communities of color resulting in social, environmental, and economic disparities that continue in cities today (Appel & Nickerson, 2016). Residents of these historically redlined neighborhoods are less likely to be college educated, have lower incomes, less access to health care, shorter overall lifespans and are at higher risk of violent crimes compared to other neighborhoods that were not redlined (Chandler 2020). Recent evidence has also found that redlined communities have less access to nature (Kephart 2022).

The Home Owners' Loan Corporation (HOLC) was established by the US government and the Federal Housing Administration (FHA) in 1933 to assist struggling Americans during the Great Depression by offering low interest housing loans (Hillier, 2003; Rothstein, 2017). To create criteria for low interest lending, the HOLC and FHA each created neighborhood ranking maps for large US Cities (Hillier, 2005). HOLC maps were color coded with red ("Hazardous") representing neighborhoods that were considered unsafe to invest in at one extreme, and green ("Best") being the safest neighborhoods for investments at the other extreme (Hillier, 2005). Red, the lowest ranked neighborhoods (i.e. redlined neighborhoods) were often majority Black and non-U.S. citizens, while green neighborhoods were predominantly affluent white communities (Crossney & Bartelt, 2005).

It is important to note that the HOLC never conducted any lending activity themselves (Hillier et al., 2003). However, there is evidence that the HOLC process of categorizing neighborhoods influenced similar efforts to assess property values by the Federal Housing Authority (FHA) (Woods 2012; Markley 2024). FHA maps categorized neighborhoods A-H, where categories A-D were white residents and E-H were communities of color or immigrants (Markley 2024). While these mapping efforts did not create segregation – they merely mapped existing segregation – these maps codified the practice of valuing neighborhoods based on class and race (Massey & Kanaiaupuni 1993).

Urban segregation and redlining has been heavily studied in regard to economics (Mentias et al., 2023; Aaronson et al., 2021) and social vulnerability (Noelke et al., 2022; Lynch et al., 2021), but only recently has research began to explore the connection between historic redlining and the environment (Estien et al., 2023; Locke et al., 2021). Recent studies such as Locke et al., (2021) found that tree canopy, an important asset to cities as trees help abate the urban heat island effect, was significantly lower (23%) in historic redlined neighborhoods compared to wealthy neighborhoods (43%). Hoffman et al., (2020) found that of 108 historically redlined cities, neighborhoods with the lowest ranking grades were up to five degrees hotter on average than historically wealthy neighborhoods, and subject to higher heat extremes during warm seasons.

The environmental changes created by historic redlining and city design have also affected urban biodiversity. In Los Angeles, California (USA), a higher diversity and abundance of forest dwelling birds was found in neighborhoods historically categorized as “Best” by HOLC (Wood et al., 2024). Whereas redlined areas were dominated by birds with generalist habitat requirements, due to increased urbanization (e.g., impervious surface) and a decrease in green infrastructure that provides wildlife habitat (Wood et al., 2024). Ellis-Soto et al., (2023) also found that historic records of bird species were lower in redlined neighborhoods across 195 US cities. These recent studies support the notion that redlined communities have less access to nature.

Here, I add to this growing body of research by studying the legacy effects of redlining on mammalian species richness in the Washington, D.C. metropolitan area. The HOLC did not map the Washington, D.C. region. However, FHA maps were drawn for Washington, D.C. and FHA lending contributed to the displacement and segregation of thousands of people by ensuring that in Washington D.C. 98 percent of housing loans went to white borrowers from 1934-1962 (Blank et al., 2005; Jackson, 1980). These decisions supported the accumulation of property and wealth by white families, and has shaped the neighborhood makeup, wealth distribution, environment, and ecology of Washington D.C. (Blank et al., 2005; Rothstein, 2017). Our objective was to assess whether there was a difference in mammalian species richness and community composition between historically redlined neighborhoods and non-redlined neighborhoods using field collected wildlife observations and historical FHA maps.

The luxury-effect hypothesis states that more affluent areas tend to have more green spaces and higher vegetation diversity; thus, more overall space for wildlife (Hoffman et al., 2020; Wood & Esaian, 2020; Schell et al., 2020). Following the legacy-effect hypothesis I hypothesized that species richness (α-diversity) and habitat use of mammals (occupancy) would be lower in redlined communities compared to non-redlined communities regardless of the level of urbanization of the respective neighborhood. I also hypothesized that community composition (β-diversity) would be different among neighborhood classification due to potential differences in habitat availability (Wood et al., 2024; Honda et al., 2019). Understanding the lasting effects of urban planning practices can inform justice-centered greenspace planning and restoration and better inform urban conservation and policy initiatives in cities.

Methods

Study Area

Our study was conducted within the Washington, D.C. metropolitan area, which includes Washington D.C., Prince George’s and Montgomery Counties, Maryland, and the City of Alexandria, Arlington County, and Fairfax County, Virginia. Washington, D.C. is the sixth largest metropolitan area within the U.S. (Smega et al. 2020). Washington, D.C. still has segregated neighborhoods as a byproduct of the assignment of historic racial covenants and neighborhood grading by the FHA starting in the 1930s, with these neighborhoods being economically impacted as well resulting in generational poverty (Chandler 2020). Sixty-seven percent of neighborhoods were still considered racially segregated in 2017, including most Black neighborhoods being located east and northeast of the Anacostia River, and white dominant neighborhoods in Northwest D.C. into Montgomery County, Maryland (Chandler 2020).

Ecologically, the climate in the Washington D.C. region experiences all four seasons, and averages 112 cm of precipitation annually (NOAA 2022). The Washington D.C. region contains a geological fall line, separating the city and surrounding lands into two distinct ecoregions, the Appalachian Piedmont region to the east, and the Mid-Atlantic coastal plain to the west (District of Columbia Department of Energy and Environment, 2015). This geological split diversifies the regions habitats including woody wetlands, coastal plain swamps, upland floodplain forests, ruderal grasslands, and open riverine habitats (District of Columbia Department of Energy and Environment, 2015). Our study area lies in the ancestral homeland of the Nacochtank (also called Anacostan) and Piscataway people (Tayac 2009).

Wildlife Data Collection

Our data comes from a spatial grid of remotely triggered trail cameras that have been established in the Washington D.C. region since 2020 (Fig. 1; see Collins et al. in review. for specific information on study design). Cameras are placed in greenspaces, including but not limited to public parks, golf courses, and cemeteries. Within each selected greenspace, a permanent sampling site was established at a location that maximized detection probability of wildlife species (e.g., animal trails, gravel roads, fence lines, etc.).

At each sampling site, one unbaited remotely triggered trail camera was deployed for approximately 30 days four times per year (January, April, July, and October) from January 2021-February 2023 (Fig. 1). Cameras were active an average of 29.4 days per season. Three models of trail camera were used, Reconyx Hyperfire 2 (Reconyx, Holmen, WI, USA), Bushnell Trophy Cam HD Aggressor, and Bushnell Trophy Cam HD (Bushnell Corporation, Overland Park, KS, USA). Cameras were placed on trees approximately 1.5 m from the ground using nylon straps and were deployed at the same location each sampling season. Cameras were set to take one photo every trigger with a 15 second rest period between triggers. Photos were uploaded to the Urban Wildlife Information Network online database (Magle et al. 2019) and animals in photos were identified to the species level. The photo identification process used a triple authentication system where species in each photo were identified by two users, and a third independent user validated any discrepancies.

Neighborhood Classifications

Washington, D.C. was categorized by the FHA wherein neighborhoods were categorized as A-H with A being the best neighborhoods to invest in and H considered areas to avoid investments (Table 1; Weimer 1937). I collapsed the FHA categorization into two categories for our analyses based on grade descriptions: I categorized E-H as “red” neighborhoods and A-D as “green” neighborhoods (Weimer 1937; Table 1).

Digital maps of the historic FHA maps for the Washington, DC metropolitan was provided by the Prologue DC and Mapping Segregation Washington, D.C. project (Prologue DC/Mapping Segregation, 2020). Using these maps, I divided our 75 sampling sites into “green” (n = 22) or “red” (n = 20) based on the FHA categorization of the neighborhood in which each sampling site was located within (Fig. 1). Camera sites that did not fall within an FHA categorized neighborhood were categorized as “U” or unassigned (n = 33). I used the unassigned category as a reference category, as these neighborhoods were not historically categorized due to low population density at the time (Mapping Inequality 2016). However, at the time of this study, they have high population densities and would be considered urban sections of the Washington, D.C. region.

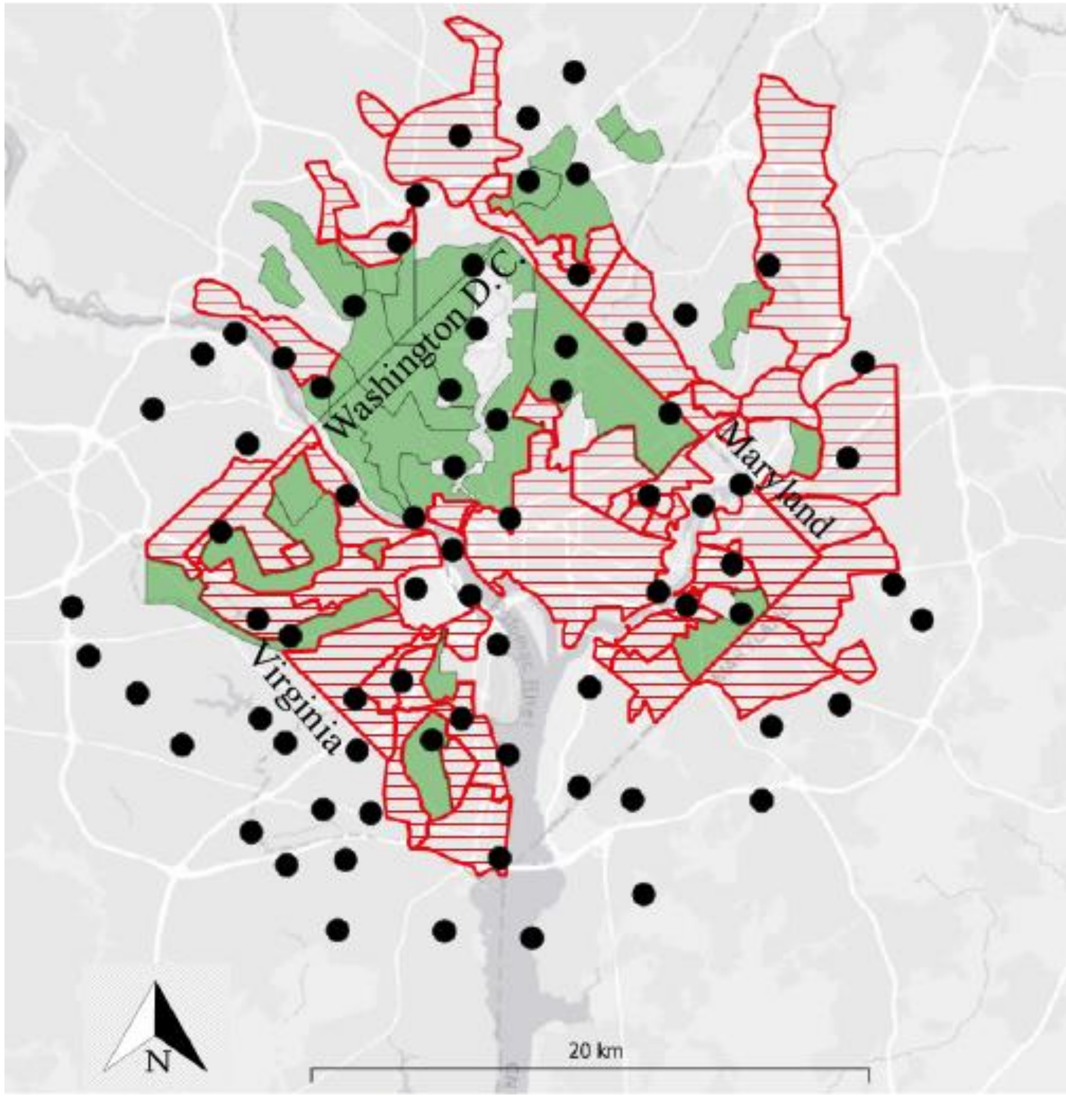

Figure 1. A map of seventy-five sampling sites (black dots) located in red (red hatched polygon), green (green polygon), and unclassified neighborhoods used to study mammal richness, diversity, and habitat use in the Washington, D.C. region.

Table 1. Neighborhood grade descriptions by the Federal Housing Administration, Divisions of Economics and Statistic, created in 1937 to categorize loan and investment risk across the Washington, D.C. metropolitan area. A-D categories were considered minimal risk for investment due to being white residency areas, and E-H categories were considered highest risk for investment due to being occupied by Black communities.

Green		Red	
A	White Residency, Upper Class, High Income, Property value of 15K+	E	Middle Class, mixed racial, lower income, property values at 5K or less
B	White Residency, Upper Class, High Income, Property value of 10K+	F	Black residency, residential homes declining to slums, low business investment areas
C	White Residency, Upper Middle Class, Property value of 7K+	G	Black residency, poor streets, little resources, property values $700
D	White Use, Transitional with some properties being converted from family homes to businesses	H	Areas meant for Black people only, no resources nearby, most homes equivalent to temporary shacks

Urban Index

To control for the varying levels of urbanization around each sampling site, I calculated an index of urbanization around each site. To do this, I extracted mean percent canopy cover, mean percent impervious surface cover, and mean housing density with a 1km buffer around each site. To calculate tree and impervious cover I used the Chesapeake Bay Conservancy Land Cover 1-m resolution data (Robinson et al. 2019; See Chapter 3). To calculate housing density, I used the U.S. Census Bureau (2020) data set for human housing density (Radeloff et al. 2018; See Chapter 3). I used a principal component analysis (PCA) to reduce the dimensionality of these three variables and used the first principal component as our index of urbanization. The first principal component accounted for 77.92% of data variation. PCA results indicated that negative values of the first principal component equate to a higher rate of urbanization (greater impervious surface and greater housing density) and positive values equate to lower urbanization (higher percentage of tree canopy). For easier interpretations, I multiplied the first principal component by -1 so that higher values indicated higher urbanization, and lower values indicated less urbanization.

Multi-Species Occupancy Model

To estimate species richness while also accounting for imperfect detection I formulated a Bayesian multi-species dynamic occupancy model using data augmentation (Mackenzie et al., 2006; Dorazio et al., 2010). This model considers the detection probability of each species and helps account for species that may have been present but were completely undetected in our study period. Dynamic occupancy models allow you to estimate initial occupancy (the probability that a site is occupied in the first time period of sampling), colonization (the probability that a site is occupied in time period t given it was unoccupied in time period t-1), and persistence (the probability a site is occupied in time period t given it was previously occupied in time period t-1) as a function of covariates. From this model formulation I was also able to derive γ-diversity (regional species pool) and α-diversity (site-level species richness) of medium-sized mammal species in the Washington D.C. area. Using our estimated latent state (that a site was occupied by each species) I also calculated β-diversity (community composition) between the different neighborhood housing grades using Jaccard's similarity index (Real & Vargas 1996). A Jaccard's index of one indicates mammal communities are the same, and an index of zero indicates no shared species at sites.

Model Formulation and Estimation

I kept the formulation of this model simple and for each species, I included the categorical variable of neighborhood classification and the urban index as covariates on initial occupancy, colonization, and persistence. Posterior distributions of model parameters were estimated using a Markov chain Monte Carlo (MCMC) algorithm in JAGS with the package runjags in R ver. 4.3.1 (Denwood 2016; R Studio Team 2022). Eight parallel chains were run from randomized starting values for 75,000 iterations with a thinning rate of ten. The first 25,000 model iterations were discarded as burn in. Thus, I retained 50,000 samples. Model convergence was evaluated by checking that the Gelman-Rubin statistic for each parameter was < 1.1 (Gelman and Rubin 1992) and by visual inspection of all trace plots. I considered model parameters to be significant if the 95% credible intervals did not overlap or if the 95% credible intervals of regression coefficients did not overlap 0.

Results

Between January 2021-February 2023, 198,700 photos of twenty mammal species, not including humans or species marked as “unknown mammal”, were captured across nine sampling seasons. In our model we included detection data for 10 species: coyote (*Canis latrans*), domestic cat (*Felis catus*), Eastern chipmunk (*Tamias striatus*), Eastern cottontail (*Sylvilagus floridanus*), Eastern gray squirrel (*Sciurus carolinensis*), raccoon (*Procyon lotor*), red fox (*Vulpes vulpes*), Virginia opossum (*Didelphis virginiana*), white-tailed deer (*Odocoileus virginianus*), and woodchuck (*Marmota monax*). I augmented our model with an additional 10 species (Royle et al., 2007) to account for a species pool that I either 1) observed but did not have enough detections to include in our model or 2) did not observe but could potentially be present in our study area (e.g., bobcat (*Lynx rufus*)).

Mammal Diversity and Habitat Use between Historic Neighborhood Grades

In this study there was no significant variation in α-diversity across neighborhood categories. The model estimated that red and green neighborhoods were most likely to have one to two species present 15-20 percent of the time (sites and sampling seasons), with a low probability of having all ten species present (5%). Uncategorized neighborhoods had an 18% chance of only having one species present, and an equally low chance (5%) of having more than one species (Fig. 2).

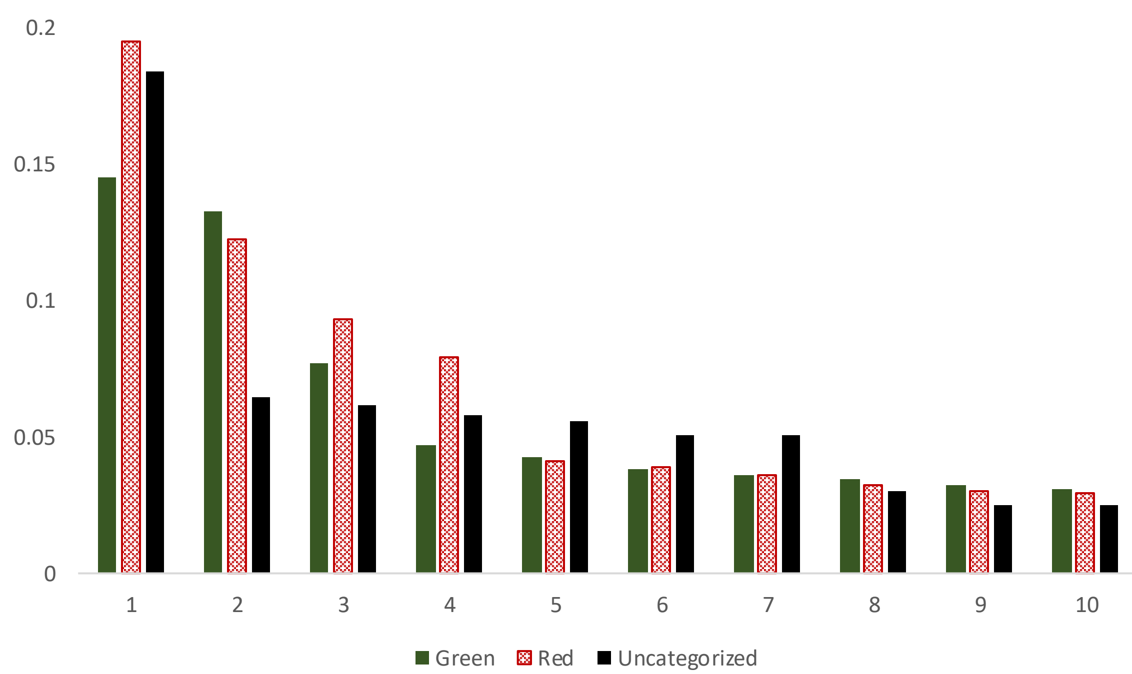

Figure 2. Histograms of median values of species richness across historic neighborhood grades in Washington, D.C. metropolitan area. Bars represent the

estimated frequency of observing the respective number of species across all sites and sampling seasons.

Similarly, there was no significant variation in β-diversity between categorizations across the nine seasons of our sampling period. Jaccard's Similarity Indices between both unclassified neighborhoods and green neighborhoods and redlined neighborhoods to green neighborhoods were consistently >0.5 indicating sites shared more than half the number of species (Fig. 3).

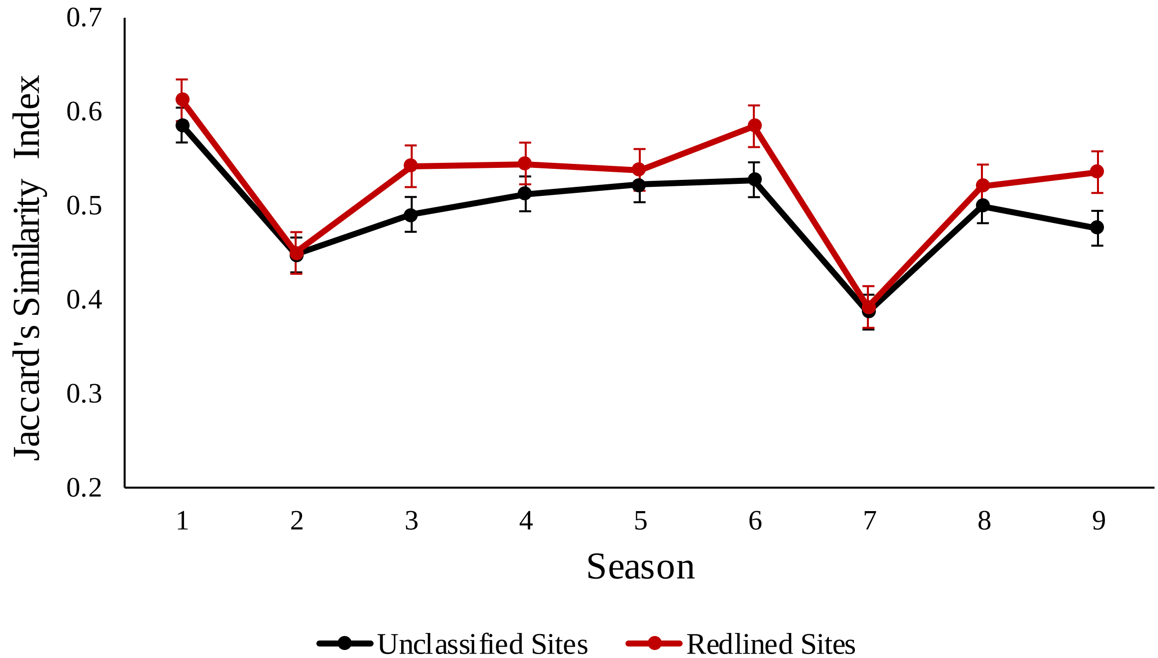

Figure 3. Jaccard's similarity indices comparing unclassified sites and sites in historic redlined neighborhoods to sites within green neighborhoods in the Washington, D.C. metropolitan area. Data is from across nine sampling seasons of camera trap data. An index of one indicates mammal communities are the same, and an index of zero indicates no shares species at sites.

Finally, the most common mammal community was similar across the three categories. The top mammal community in historically redlined neighborhoods and uncategorized neighborhoods were white-tailed deer, red fox, and Virginia opossum. In green neighborhoods white-tailed deer, red fox, and woodchuck were the most common species across all seasons. Species-specific initial occupancy, colonization, nor persistence differed between housing grades for any species (Table 2).

Table 2. Median model coefficients and 95% credible intervals for the effects of historic housing grade on species-specific initial occupancy, colonization, and persistence across the Washington, D.C. metropolitan area. No significant relationship between housing grade, occupancy, persistence, or colonization was found.

Initial Occupancy Redlined			
Species	Lower	Median	Upper
Coyote	0.1484689	0.4763604	0.8106186
Domestic cat	0.1925920	0.5345381	0.8444973
Eastern chipmunk	0.1081119	0.4830471	0.8444464
Eastern cottontail rabbit	0.1822709	0.5135569	0.8286656
Eastern gray squirrel	0.2180272	0.5944537	0.8975064
Raccoon	0.1829377	0.5160654	0.8404901
Red fox	0.2543257	0.6537959	0.9337163
Virginia opossum	0.2262680	0.5697415	0.8621051
White-tailed deer	0.3235071	0.6960762	0.9324570
Woodchuck	0.2865834	0.6939456	0.9449172
Initial Occupancy Green			
Coyote	0.1584719	0.4755699	0.8162880
Domestic cat	0.1355919	0.4418047	0.7962721
Eastern chipmunk	0.08763346	0.40246043	0.80523990
Eastern cottontail rabbit	0.1590501	0.4643616	0.7981198
Eastern gray squirrel	0.1557064	0.4803535	0.8385032
Raccoon	0.1927787	0.5219417	0.8497077
Red fox	0.1763655	0.5349898	0.8913416
Virginia opossum	0.09867797	0.35627679	0.71704482
White-tailed deer	0.1065056	0.3505763	0.7103949
Woodchuck	0.07550619	0.37352979	0.78461585
Initial Occupancy Undefined Land Category			
Coyote	0.1118274	0.3506054	0.6987399
Domestic cat	0.09942089	0.33022060	0.67895637

Eastern chipmunk	0.05802577	0.30405193	0.69919829
Eastern cottontail rabbit	0.1219331	0.3661842	0.7046715
Eastern gray squirrel	0.1912216	0.5031754	0.8368799
Raccoon	0.1872163	0.4775277	0.8017593
Red fox	0.1443029	0.4415205	0.8018407
Virginia opossum	0.1573805	0.4278778	0.7590074
White-tailed deer	0.1767694	0.4637543	0.7914938
Woodchuck	0.04828404	0.25999364	0.64374508
Colonization Redlined			
Species	Lower	Median	Upper
Coyote	0.1082148	0.3194624	0.6447802
Domestic cat	0.1460364	0.4058773	0.7353621
Eastern chipmunk	0.06026648	0.21686205	0.52084362
Eastern cottontail rabbit	0.07523617	0.24427479	0.54920106
Eastern gray squirrel	0.0817611	0.2856825	0.6456870
Raccoon	0.08247606	0.27333847	0.61384077
Red fox	0.09156592	0.29527426	0.64704950
Virginia opossum	0.07063288	0.23367466	0.53930658
White-tailed deer	0.07078481	0.27375282	0.65169250
Woodchuck	0.06050694	0.22996501	0.55932768
Colonization Green			
Coyote	0.09366425	0.28596847	0.21454972
Domestic cat	0.0817371	0.2548870	0.18163325
Eastern chipmunk	0.09083022	0.28714598	0.61007117
Eastern cottontail rabbit	0.1202381	0.3477686	0.6749889
Eastern gray squirrel	0.09006527	0.32289974	0.69990972
Raccoon	0.1451477	0.4142494	0.25760716
Red fox	0.1243988	0.3835704	0.7510141
Virginia opossum	0.1480588	0.3852942	0.6978899
White-tailed deer	0.09850831	0.33802236	0.71351116
Woodchuck	0.1212421	0.3493429	0.6767896
Colonization Undefined Land Category			
Coyote	0.6447802	0.59856947	0.49406897
Domestic cat	0.05429605	0.18163325	0.44774621
Eastern chipmunk	0.09568973	0.28228117	0.59355061
Eastern cottontail rabbit	0.07297905	0.22387473	0.50764713
Eastern gray squirrel	0.1158038	0.3509526	0.7102900
Raccoon	0.08507428	0.25760716	0.57003063
Red fox	0.09132652	0.27466042	0.59559484
Virginia opossum	0.07830408	0.23345601	0.51765417
White-tailed deer	0.0866305	0.2774405	0.6170332

Woodchuck	0.06652993	0.21691015	0.50917184
	Persistence Redlined		
Species	Lower	Median	Upper
Coyote	0.2555981	0.6290845	0.8819056
Domestic cat	0.4274807	0.7555594	0.9283090
Eastern chipmunk	0.3107013	0.7059683	0.9300224
Eastern cottontail rabbit	0.3096816	0.6646395	0.8948822
Eastern gray squirrel	0.3091313	0.6393756	0.8749499
Raccoon	0.3847497	0.7074620	0.9035436
Red fox	0.2828657	0.6062351	0.8542405
Virginia opossum	0.5671522	0.8466825	0.9623899
White-tailed deer	0.4195896	0.7593852	0.9384769
Woodchuck	0.3375209	0.7144594	0.9218559
	Persistence Green		
Coyote	0.2515501	0.6561637	0.9001088
Domestic cat	0.3575405	0.7157447	0.9145017
Eastern chipmunk	0.3452054	0.7320861	0.9300852
Eastern cottontail rabbit	0.4893193	0.8011319	0.9457616
Eastern gray squirrel	0.5722647	0.8413254	0.9585227
Raccoon	0.5841143	0.8448002	0.9576543
Red fox	0.5450360	0.8227785	0.9484209
Virginia opossum	0.3889867	0.7255536	0.9157075
White-tailed deer	0.4135436	0.7651559	0.9403676
Woodchuck	0.2158761	0.6198856	0.8852560
	Persistence Undefined Land Category		
Coyote	0.4586099	0.7816936	0.9390441
Domestic cat	0.3002266	0.6471828	0.8770909
Eastern chipmunk	0.3350926	0.6996547	0.9120505
Eastern cottontail rabbit	0.3541400	0.6856228	0.8939674
Eastern gray squirrel	0.4650772	0.7608645	0.9232991
Raccoon	0.3562818	0.6703223	0.8809156
Red fox	0.4894369	0.7750813	0.9256068
Virginia opossum	0.2329022	0.5454044	0.8183897
White-tailed deer	0.4614792	0.7852694	0.9478069
Woodchuck	0.4019719	0.7542592	0.9363210

Urbanization

Urbanization had a negative effect on the probability of initial occupancy of Virginia opossum (β = -0.53, 95% CI = -1.06 – -0.07; Table 3) but did not have a significant effect on initial occupancy of any other species. Urbanization had a negative effect on Raccoon persistence (B = -0.59, 95% CI = -0.94 – -0.30; Table 3) but did not have a significant effect on the probability of persistence for any other species (Table 3). I found no effect of urbanization on colonization probability of any species in our study.

Table 3. Median model coefficients and 95% credible intervals for the effects of urbanization on species-specific initial occupancy, colonization, and persistence across the Washington, D.C. metropolitan area. ** Indicates 95% CI did not overlap zero.

Initial Occupancy			
Species	Lower	Median	Upper
Coyote	-0.69806	-0.15857	0.345437
Domestic cat	-1.06161	-0.49547	0.009552
Eastern chipmunk	-1.42732	-0.10906	1.241217
Eastern cottontail rabbit	-0.48368	-0.04277	0.385606
Eastern gray squirrel	-0.06969	0.528608	1.203991
Raccoon	-0.41478	0.042237	0.531353
Red fox	-0.53081	0.177439	0.940371
Virginia opossum**	-1.06111	-0.52661	-0.07773
White-tailed deer	-0.67374	-0.19932	0.265959
Woodchuck	-1.77244	-0.54138	0.52161
Colonization			
Species	Lower	Median	Upper
Coyote	-0.28932	0.03695	0.358382
Domestic cat	-0.26728	0.076201	0.432352
Eastern chipmunk	-0.70946	-0.27779	0.108389
Eastern cottontail rabbit	-0.50724	-0.143	0.191313
Eastern gray squirrel	-0.12751	0.319835	0.865899

Raccoon	-0.27907	0.101524	0.490282
Red fox	-0.14195	0.285925	0.768812
Virginia opossum	-0.39273	-0.08412	0.217258
White-tailed deer	-0.48825	0.225336	0.986496
Woodchuck	-0.40983	0.005271	0.392485
	Persistence		
Species	Lower	Median	Upper
Coyote	-0.95822	-0.29298	0.339495
Domestic cat	-0.93752	-0.40695	0.089562
Eastern chipmunk	-1.10077	-0.41975	0.194897
Eastern cottontail rabbit	-0.35484	0.070113	0.520433
Eastern gray squirrel	-0.56589	-0.25616	0.044364
Raccoon**	-0.93542	-0.59901	-0.29748
Red fox	-0.45433	-0.18423	0.083608
Virginia opossum	-0.50417	-0.11135	0.285137
White-tailed deer	-0.10238	0.420848	1.046344
Woodchuck	-0.61139	0.06759	0.655618

Discussion

Urban ecology research is just beginning to explore the effects of historic urban planning decisions on wildlife communities. Here, I analyzed camera trap data across historic Federal Housing Administration neighborhood grades in the Washington, DC metropolitan area to understand if the racist practice of redlining influenced modern-day assemblages of medium-sized mammals. Contrary to our hypothesis, I found no correlation between historic redlining and modern-day mammal species richness and no difference among mammal communities across the different neighborhood categorization. Eistein et al. (2023) found that species diversity, abundance, and detection across historic HOLC categories varied by city. My results suggest that more single-city studies are needed to understand how city-specific nuances of historic land use practices correlate to modern day species assemblages.

Species Richness and Community Composition

Species richness and community composition across the three housing categorizations in this study area were similar and I did not find a relationship between historic housing grade and mammal habitat use. The most often observed mammal community in historically redlined and uncategorized neighborhoods was white-tailed deer, red fox, and Virginia opossum. Similarly, in green categorized neighborhoods the most observed mammal community was white-tailed deer, red fox, and woodchuck. One explanation for these similarities is that the mammal communities in our study area have already been filtered by the urban landscape to only include generalist species (Aronson et al., 2016), and more specialized species requiring specific habitats or dietary needs are not as common in these urban environments regardless of historic land use (McCleery 2010; Ducatez et al., 2018; Santini et al., 2019).

Vertebrate species that have more general habitat and dietary needs and can exhibit behavioral plasticity are able to exist near humans and these tend to be the species that persist in cities (Ducatez et al., 2018). White-tailed deer, red fox, and Virginia opossum – three of the most common species in our study – benefit from the supplemental resources such as food (Batemen & Fleming, 2012) and den sites (Wright et al., 2012) that humans directly or indirectly provide. These species also often lack natural predators in cities (Blanchong et al., 2013), and can therefore thrive in urban environments. It is likely that these common mammal species occur across all neighborhood grades due to plastic biological and behavioral traits and not due to historic land use differences between neighborhoods. Further, the mammals in this study have varying dietary needs. While some, like woodchucks, depend primarily on vegetation, others, like red fox have much more omnivorous diets. These dietary strategies may decouple mammal species from vegetative age and diversity that have been correlated with redlined neighborhoods in similar studies on birds (Estein et al., 2023; Wood et al., 2024).

It is not clear why woodchucks are a common species in neighborhoods categorized as green, but not redlined or uncategorized neighborhoods. Little research has been conducted on urban woodchuck habitat use, but one study found a negative relationship between woodchuck body condition and urbanization (Hellgren & Polnaszek 2011). Woodchuck prefers habitat along forest edges and are known to use manicured fields and residential lawns (Hellgren & Polnaszek 2011; Armitage 2003). Nardone et al. (2021) found that redlined neighborhoods had lower values of Normalized Vegetative Index (NDVI) indicating that these neighborhoods have less overall vegetation than non-redlined areas. Our results could be evidence of a luxury effect (Honda et al., 2018) where more affluent neighborhoods have more vegetation and therefore more habitat for woodchuck (Clark et al., 2013; Aznarez et al., 2023). However, more information would be needed to better understand woodchuck habitat use in cities before inferring a luxury effect.

Urbanization Effects on Mammal Occupancy, Colonization, and Persistence

While multi-city studies suggest that urbanization can affect colonization or persistence of mammal species in the US (Magle et al., 2021), I did not find this trend in Washington, D.C. The only species in our study that showed a correlation between persistence and urbanization was raccoon, which had a higher probability of persistence at less urban sites, and opossum, which had lower probability of initial occupancy at more urban sites.

Mesopredator response to urbanization and human development is complex and can result in a variation of occupancy and behavioral patterns (Veon et al., 2023). Raccoons are known to be adaptable urban exploiters that can live in highly urban areas (Rodriguez et al., 2021). However, Washington, D.C. maintains the highest proportion of parkland of any major U.S. city at 21.9% of total land cover (Cohen et al. 2017). In our specific study area, racoons may persist less in the urban core since higher quality habitat is available across the city.

Virginia opossums are also adaptable urban exploiters (Veon et al., 2023; Fidino et al., 2016). Veon et al., (2023) found that opossums in a multi-city study were more likely to occur in areas with higher anthropogenic sound and human density. The lower probability of initial occupancy in more urban areas of Washington, D.C. may be a result of nomadic behavior associated with opossum as this species tends to move den sites nightly (Gillette 1980). No effect on persistence or colonization may translate to the generalist nature of opossums seen in other urban species, including the exploitation of human provided resources such as food, water, and denning sites regardless of urbanization levels (Veon et al., 2023; Larson et al., 2020).

Future Work and Management Implications

There have been several multi-city studies published on the relationship between historic urban planning, wealth, and biodiversity in cities (Ellis-Soto et al., 2023; Estein et al., 2023; Magle et al., 2023) but few have focused on this relationship in a single city (however, see Wood et al., 2024). These multi-city studies have created a baseline of generalizations to help guide research. However, as cities across the U.S. can vary by native species composition, vegetation regimes, climate, green space quality, and intensity of human development it is crucial – from a conservation and management perspective – to conduct single city studies to understand the context of these variables at the individual city level. The results of this study demonstrate the importance of single city studies, as Washington, D.C. does not follow the general trends of animal community relationships with legacy effects of historic urban planning decisions that have been seen in multi-city analyses (Ellis-Soto et al., 2023; Magle et al., 2021; Eistein et al., 2023). Thus, single city studies like ours can provide context specific management recommendations to urban planners and natural resource practitioners.

While this study did not find evidence of a strong relationship between historic housing grades and mammal diversity in Washington, DC, I did not directly analyze habitat connectivity or species movement – key factors in wildlife conservation and management. Unlike birds, mammals have a much more limited capacity to move through the urban matrix without adequately connected habitat (Bierwagen, 2007). Gallo et al., (in-review) suggests that historically redlined areas of cities have less overall green space, and less connectivity between available green spaces compared to green categorized areas. This lack of connectivity may affect a mammal's ability to move between neighborhoods and can cause genetic isolation (Schmidt & Garroway 2022). Thus, future studies on the relationship between historic housing and mammal communities should take landscape connectivity and movement corridors into account.

Mammal habitat use within different housing grades was similar, but I did not study wildlife population health. Schmidt and Garroway (2021) found that overall genetic diversity of species was lower in minority neighborhoods across 268 locations in US cities, suggesting that environmental conditions in historically redlined neighborhoods may not be conducive to wildlife health. Given known differences in human health outcomes in redlined communities (Nardone et al., 2020), it is possible that mammal health could also be affected. Mammals could therefore serve as environmental sentinels, or "proxies" for human health in these communities as they do in other landscapes (García-Fernández et al., 2020). Future studies should take a One Health approach when studying the relationships between historic urban planning and biodiversity.

Conclusion

Managing urban wildlife to account for human well-being as well as conservation can be challenging. Thus, it is paramount to create a comprehensive understanding of how historic urban planning, including social inequities, have shaped native biodiversity distribution and habitat use (Schmidt & Garroway, 2021). Management decisions to achieve conservation initiatives and improve human well-being will therefore need to be made by diverse interdisciplinary teams that consider current and past environmental justice issues, and social inequities, while also promoting healthy wildlife populations and habitat of native species (Schell et al., 2020). More single city studies on the intersection of historic city planning and wildlife, including incorporating data on discriminatory housing practices like redlining, will help inform managers in those cities about the most effective strategies to promote species persistence and create just land use strategies that benefit people and biodiversity simultaneously.

Literature Cited

Aaronson, D., Faber, J., Hartley, D., Mazumder, B., & Sharkey, P. (2021). The long-run effects of the 1930s HOLC “redlining” maps on place-based measures of economic opportunity and socioeconomic success. Regional Science and Urban Economics, 86, 103622.

Appel, Ian, and Jordan Nickerson. "Pockets of poverty: The long-term effects of redlining." Available at SSRN 2852856 (2016).

Armitage, K. B. (2003). Marmots, p. 188-210. In: G. A. Feldhamer, B. C. Thompson, and J. A. Ch (eds.). Wild mammals of North America: biology, conservation, and management. Se edition. Johns Hopkins University Press, Baltimore, Ma

Aronson, M. F., Nilon, C. H., Lepczyk, C. A., Parker, T. S., Warren, P. S., Cilliers, S. S., ... & Zipperer, W. (2016). Hierarchical filters determine community assembly of urban species pools. Ecology, 97(11), 2952-2963.

Aznarez, C., Svenning, J. C., Pacheco, J. P., Have Kallesøe, F., Baró, F., & Pascual, U. (2023). Luxury and legacy effects urban biodiversity, vegetation cover and ecosystem services. npj Urban Sustainability, 3(1), 47.

Bateman, P. W., & Fleming, P. A. (2012). Big city life: carnivores in urban environments. Journal of Zoology, 287(1), 1-23.

Bierwagen, B. G. (2007). Connectivity in urbanizing landscapes: The importance of habitat configuration, urban area size, and dispersal. Urban Ecosystems, 10, 29-42.

Blanchong, J. A., Sorin, A. B., & Scribner, K. T. (2013). Genetic diversity and population structure in urban white-tailed deer. The Journal of Wildlife Management, 77(4), 855-862.

Blank, E. C., Venkatachalam, P., McNeil, L., & Green, R. D. (2005). Racial Discrimination in Mortgage Lending in Washington, DC: A Mixed Methods Approach1. The Review of Black Political Economy, 33(2), 9-30.

Chandler, A. (2021). Tearing down" No Section 8" Signs: The Disparate Racial Impact of Source-of-Income Discrimination and the Validity of Louisville's New Law against It. U. Louisville L. Rev., 60, 127.

Clarke, L. W., Jenerette, G. D., & Davila, A. (2013). The luxury of vegetation and the legacy of tree biodiversity in Los Angeles, CA. Landscape and Urban Planning, 116, 48-59.

Cohen, D. A., B. Han, K. R. Evenson, C. Nagel, T. L. McKenzie, T. Marsh, S. Williamson, and P. Harnik (2017). The prevalence and use of walking loops in neighborhood parks: A national study. Environmental Health Perspectives, 125(2), 170–174.

Crossney, K. B., & Bartelt, D. W. (2005). The legacy of the homeowners' loan corporation. Housing Policy Debate, 16(3-4), 547-574.

Denwood, M. J. (2016). runjags: An R package providing interface utilities, model templates, parallel computing methods and additional distributions for MCMC models in JAGS. Journal of Statistical Software, 71, 1-25.

Dorazio, R. M., Kéry, M., Royle, J. A., & Plattner, M. (2010). Models for inference in dynamic metacommunity systems. Ecology, 91(8), 2466-2475.

Ducatez, S., Sayol, F., Sol, D., & Lefebvre, L. (2018). Are urban vertebrate city specialists, artificial habitat exploiters, or environmental generalists? Integrative and Comparative Biology, 58(5), 929-938.

Ellis-Soto, D., Chapman, M., & Locke, D. H. (2023). Historical redlining is associated with increasing geographical disparities in bird biodiversity sampling in the United States. Nature Human Behaviour, 7(11), 1869-1877.

Estien, C. O., Fidino, M., Wilkinson, C. E., Morello-Frosch, R., & Schell, C. J. (2023). Historical redlining impacts wildlife biodiversity across California.

Fidino, M. A., Lehrer, E. W., & Magle, S. B. (2016). Habitat dynamics of the Virginia opossum in a highly urban landscape. The American Midland Naturalist, 175(2), 155-167.

García-Fernández, A. J., Espín, S., Gómez-Ramírez, P., Martínez-López, E., & Navas, I. (2020). Wildlife sentinels for human and environmental health hazards in ecotoxicological risk assessment. Ecotoxicological QSARs, 77-94.

Gelman, A., & Rubin, D. B. (1992). Inference from iterative simulation using multiple sequences. Statistical Science, 7(4), 457-472.

Gillette, L. N. (1980). Movement patterns of radio-tagged opossums in Wisconsin. American Midland Naturalist, 1-12.

Hellgren, E. C., & Polnaszek, T. J. (2011). Survival, habitat selection, and body condition of the woodchuck (*Marmota monax*) across an urban-rural gradient. The American Midland Naturalist, 165(1), 150-161.

Hillier, A. E. (2003). Redlining and the homeowners' loan corporation. Journal of Urban History, 29(4), 394-420.

Hillier, A. (2005). Searching for red lines: Spatial analysis of lending patterns in Philadelphia, 1940–1960. Pennsylvania History: A Journal of Mid-Atlantic Studies, 72(1), 25-47.

Hoffman, J. S., Shandas, V., & Pendleton, N. (2020). The effects of historical housing policies on resident exposure to intra-urban heat: a study of 108 US urban areas. Climate, 8(1), 12.

Honda T, Iijima H, Tsuboi J, Uchida K (2018). A review of urban wildlife management from the animal personality perspective: the case of urban deer. Sci Total Environ 644:576–582.

Jackson KT (1980). Race, ethnicity, and real estate appraisal: The Homeowners Loan Corporation and the Federal Housing Administration. Journal of Urban History 6(4): 419–452.

Kephart, L. (2022). How racial residential segregation structures access and exposure to greenness and green space: A review. Environmental Justice, 15(4), 204-213.

Larson, R. N., Brown, J. L., Karels, T., & Riley, S. P. D. (2020). Effects of urbanization on resource use and individual specialization in coyotes (*Canis latrans*) in southern California. PLoS One, 15, e0228881.

Locke, D. H., Hall, B., Grove, J. M., Pickett, S. T., Ogden, L. A., Aoki, C., ... & O'Neil-Dunne, J. P. (2021). Residential housing segregation and urban tree canopy in 37 US Cities. npj urban sustainability, 1(1), 15.

Lynch, E. E., Malcoe, L. H., Laurent, S. E., Richardson, J., Mitchell, B. C., & Meier, H. C. (2021). The legacy of structural racism: associations between historic redlining, current mortgage lending, and health. SSM-population health, 14, 100793.

Mackenzie, D. I. (2006). Modeling the probability of resource use: the effect of, and dealing with, detecting a species imperfectly. The Journal of Wildlife Management, 70(2), 367-374.

Magle, S. B., M. Fidino, E.W. Lehrer, T. Gallo, M.P. Mulligan, M.J. Ríos, A.A. Ahlers, J. Angstmann, A. Belaire, and B. Dugelby (2019). Advancing urban wildlife research through a multi-city collaboration. Frontiers in Ecology and the Environment, 17(4), 232–239.

Magle, S. B., Fidino, M., Sander, H. A., Rohnke, A. T., Larson, K. L., Gallo, T., ... & Schell, C. J. (2021). Wealth and urbanization shape medium and large terrestrial mammal communities. Global change biology, 27(21), 5446-5459.

Mapping Inequality. (2016). United States. Retrieved from the Library of Congress, https://www.loc.gov/item/lcwaN0026813/.

Markley, S. (2024). Federal ‘redlining’maps: A critical reappraisal. Urban Studies, 61(2), 195-213.

Massey, D. S. & Kanaiaupuni, S. M. Public Housing and the Concentration of Poverty. Social Science Quarterly 74, 109–122 (1993).

McCleery, R. (2010). Urban mammals. Urban Ecosystem Ecology, 55, 87-102.

Mentias, A., Mujahid, M. S., Sumarsono, A., Nelson, R. K., Madron, J. M., Powell-Wiley, T. M., & Pandey, A. (2023). Historical redlining, socioeconomic distress, and risk of heart failure among Medicare beneficiaries. Circulation, 148(3), 210-219.

Nardone, A., Rudolph, K. E., Morello-Frosch, R., & Casey, J. A. (2021). Redlines and greenspace: the relationship between historical redlining and 2010 greenspace across the United States. Environmental Health Perspectives, 129(1), 017006.

National Oceanic and Atmospheric Administration [NOAA] (2022). Climate data online, daily summaries: Washington 5.1 NW, DC US; Washington 3.1 NNE, DC US; Washington 2.0 SSW, DC US. Available online at: https://www.ncdc.noaa. gov/cdo-web/datasets.

Noelke, C., Outrich, M., Baek, M., Reece, J., Osypuk, T. L., McArdle, N., & Acevedo-Garcia, D. (2022). Connecting past to present examining different approaches to linking historical redlining to present day health inequities. PloS One, 17(5), e0267606.

Prange, S., Gehrt, S. D., & Wiggers, E. P. (2004). Influences of anthropogenic resources on raccoon (*Procyon lotor*) movements and spatial distribution. Journal of Mammalogy, 85(3), 483-490.

R Core Team (2022). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Radeloff, V. C., D.P. Helmers, H.A. Kramer, H.H. Mockrin, P.M. Alexandre, A. Bar-Massada, V. Butsic, T.J. Hawbaker, S. Martinuzzi, and A.D. Syphard (2018). Rapid growth of the US wildland-urban interface raises wildfire risk. Proceedings of the National Academy of Sciences, 115(13), 3314–3319.

Real, R., & Vargas, J. M. (1996). The probabilistic basis of Jaccard's index of similarity. Systematic Biology, 45(3), 380-385.

Robinson, C., L. Hou, K. Malkin, R. Soobitsky, J. Czawlytko, B. Dilkina, and N. Jojic (2019). Large scale high-resolution land cover mapping with multi-resolution data. In

Proceedings of the IEEE/CVF Conference on Computer Vision and Pattern Recognition (pp. 12726-12735).

Rodriguez, J. T., Lesmeister, D. B., & Levi, T. (2021). Mesocarnivore landscape use along a gradient of urban, rural, and forest cover. PeerJ, 9, e11083.

Rothstein, R. (2017). The color of law: A forgotten history of how our government segregated America. Liveright Publishing.

Royle, J. A., Dorazio, R. M., & Link, W. A. (2007). Analysis of multinomial models with unknown index using data augmentation. Journal of Computational and Graphical Statistics, 16(1), 67-85.

Santini, L., González-Suárez, M., Russo, D., Gonzalez-Voyer, A., von Hardenberg, A., & Ancillotto, L. (2019). One strategy does not fit all: determinants of urban adaptation in mammals. Ecology Letters, 22(2), 365-376.

Schell, C. J., Dyson, K., Fuentes, T. L., Des Roches, S., Harris, N. C., Miller, D. S., ... & Lambert, M. R. (2020). The ecological and evolutionary consequences of systemic racism in urban environments. Science, 369(6510), eaay4497.

Schmidt, C., Dray, S., & Garroway, C. J. (2022). Genetic and species-level biodiversity patterns are linked by demography and ecological opportunity. Evolution, 76(1), 86-100.

Semega, J., M. Kollar, E. Shrider, and J. Creamer (2020). U.S. Census Bureau, Current Population Reports, P60-270 (RV), Income and Poverty in the United States: 2019, U.S. Government Publishing Office, Washington, DC.

Tayac, G. (2009). IndiVisible: African-Native American Lives in the Americas. Smithsonian Books.

Veon, J. T., Lassiter, E. V., Johansson, E., Shaw, M., McTigue, L., Massey, A., ... & DeGregorio, B. A. (2023). Influence of human development and predators on patterns of Virginia opossum occupancy, abundance, and activity. Journal of Zoology, 321(4), 278-288.

Weimer, A. M. (1937). The Work of the Federal Housing Administration. Journal of Political Economy, 45(4), 466-483.

Wood, E. M., & Esaian, S. (2020). The importance of street trees to urban avifauna. Ecological Applications, 30(7), e02149.

Wood, E. M., Esaian, S., Benitez, C., Ethington, P. J., Longcore, T., & Pomara, L. Y. (2024). Historical racial redlining and contemporary patterns of income inequality negatively affect birds, their habitat, and people in Los Angeles, California. Ornithological Applications, 126(1), duad044.

Woods, L. L. (2012). The Federal Home Loan Bank Board, redlining, and the national proliferation of racial lending discrimination, 1921–1950. Journal of Urban History, 38(6), 1036-1059.

Wright, J. D., Burt, M. S., & Jackson, V. L. (2012). Influences of an urban environment on home range and body mass of Virginia opossums (*Didelphis virginiana*). Northeastern Naturalist, 19(1), 77-86.

Chapter 5: Conclusion

A central theme in this research has been incorporating ecological past, present, and future into research questions and methodologies on urban wildlife. Across this dissertation these themes are emphasized in each chapter, and with this in mind, we now ask “what now?” How do we incorporate thinking about these themes into the future of this discipline?

There have been many lessons learned across the four years it took to produce the scientific research in this document. One major lesson learned is that no single profession can accomplish effective science alone. We see this theme emerge across all chapters, for instance, social sciences moving to prominence within the published urban wildlife literature over the last decade (Collins et al., 2021). Or the need for single city research across disciplines to understand the legacy effect of housing segregation on biodiversity (Wood et al., 2024).

Incorporating interdisciplinary work is happening in conservation, but individual scientists, leaders in the field, and institutions need to do more to support and promote transdisciplinary research that transcends disciplines. For example, urban wildlife research continues to be conducted by mostly academics (Magle et al., 2012; Collins et al., 2021). Aronson et al. (2017) found that a major challenge in urban conservation is access to scientific research for state, federal, and municipal natural resource managers. However, there is no impetus for academics to work directly with practitioners and managers (Rupp 2012), nor is there a reward structure for academic research to be used in management and policy decisions (Rupp 2012; Bury 2006).

Therefore, I suggest that an easy path forward is to include practitioners and managers – those doing direct on the ground conservation – directly in the research project as I have done (Chapter 2. Partnering universities with NGO and natural resource managers at state or federal levels will strengthen the direct application of science coming from academia, and benefit managers who might not have the time or resources to commit leading research projects. Developing strong relationships between on the ground practitioners is paramount to successful urban biodiversity conservation.

Another conclusion that can be drawn from this dissertation is that while most wildlife research focuses on the conservation of wildlife, people are an integral part of all ecosystems (Schell et al., 2020). With so many people calling cities home, and urban areas continuing to grow in spatial extent, it is impossible to ignore that we will need to place a greater emphasis on conserving urban species to protect species globally (Aronson et al., 2017). We can see this in Chapter 3, wherein turkeys, that were once extirpated from the landscape, have now returned to a very different ecosystem. While this species is experiencing declines in the more natural areas of its range, it appears to be doing well in many cities.

Therefore, we can no longer focus most, if not all, financial resources into researching and conserving "pristine" environments. Urban ecosystems may not be the first thing that comes to mind as an important ecosystem for many, but urban ecosystems will soon dominate earth's landscape (United Nations 2018). Therefore, more funding needs to be available for urban wildlife studies to create healthier cities for people and biodiversity.

Stemming from the last conclusion, to create healthy cities for people and wildlife, the field of urban ecology needs to connect the history of people and landscapes, with the current distribution of wildlife and their habitats and reconsider how we refer to ecology in cities (Noelke et al., 2022). For example, in Chapter 4 I found that there is little research on the intersection of ecology, biodiversity, and historic segregation and resource distributions in cities. Recent research has shown that the configuration of urban landscapes, and therefore the green spaces and habitats within cities (Locke et al., 2021), result from historic policies and practices that were often discriminatory and unjust (Schell et al., 2020). Scientists should work to better understand how to mitigate the legacy effects of past injustices, especially in cities, where many people are still affected by decisions that were made decades ago.

Considering the intertwined influences of the past and present on future ecological must also consider how we can use this information to create more equitable, just, and healthy landscapes for people to live alongside wildlife. Justice-centered urban ecology will take an interdisciplinary approach to ecological problem solving, adequate resource distribution of funding for ecological research, and a consideration for people just as much as wildlife across systems. This is especially true for cities, which can be host equally important habitats for species in areas designed for humans. Taking this all-encompassing approach to ecology research will allow effective application of research findings that can benefit all.

Literature Cited

Aronson, M.F., Lepczyk, C.A., Evans, K.L., Goddard, M.A., Lerman, S.B., MacIvor, J.S., Nilon, C.H. & Vargo, T. (2017). Biodiversity in the city: key challenges for urban green space management. Frontiers in Ecology and the Environment, 15(4).

Bury, R. B. (2006). Natural history, field ecology, conservation biology and wildlife management: time to connect the dots. Herpetological Conservation and Biology, 1(1), 56-61.

Collins, M. K., Magle, S. B., & Gallo, T. (2021). Global trends in urban wildlife ecology and conservation. Biological Conservation, 261, 109236.

Locke, D. H., Hall, B., Grove, J. M., Pickett, S. T., Ogden, L. A., Aoki, C., ... & O'Neil-Dunne, J. P. (2021). Residential housing segregation and urban tree canopy in 37 US Cities. npj urban sustainability, 1(1), 15.

Magle, S. B., Hunt, V. M., Vernon, M., & Crooks, K. R. (2012). Urban wildlife research: past, present, and future. Biological Conservation, 155, 23-32.

Noelke, C., Outrich, M., Baek, M., Reece, J., Osypuk, T. L., McArdle, N., & Acevedo-Garcia, D. (2022). Connecting past to present examining different approaches to linking historical redlining to present day health inequities. PloS One, 17(5), e0267606.

Rupp, S. P. (2012). Practical dissonance: Bridging the gap between academia and agencies. Wildlife Society Bulletin, 36(4), 796-801.

Schell, C. J., Dyson, K., Fuentes, T. L., Des Roches, S., Harris, N. C., Miller, D. S., ... & Lambert, M. R. (2020). The ecological and evolutionary consequences of systemic racism in urban environments. Science, 369(6510), eaay4497.

United Nations, Department of Economic and Social Affairs, Population Division (2018). World Urbanization Prospects: The 2018 Revision (ST/ESA/SER.A/420). New York: United Nations.

Wood, E. M., Esaian, S., Benitez, C., Ethington, P. J., Longcore, T., & Pomara, L. Y. (2024). Historical racial redlining and contemporary patterns of income inequality negatively affect birds, their habitat, and people in Los Angeles, California. Ornithological Applications, 126(1), duad044.