UNIVERSITY OF CALIFORNIA

Los Angeles

The Relationship Between Urban Built Form and Urban Biodiversity: An exploration of the influence of urban built form attributes on avian biodiversity in urban green spaces

A thesis submitted in partial satisfaction of the requirements for the degree Master of Urban and Regional Planning

by

Morgan Rogers

2021

ABSTRACT OF THE THESIS

The Relationship Between Urban Built Form and Urban Biodiversity: An exploration of the influence of urban built form attributes on avian biodiversity in urban green spaces

by

Morgan Rogers

Master of Urban and Regional Planning

University of California, Los Angeles, 2021

Professor V. Kelly Turner, Chair

Urban biodiversity plays an important role in ecological processes and ecosystem services within cities, making conservation a priority in many municipal sustainability plans. Urban green spaces (UGS) have been a key strategy for conservation by providing habitat for wildlife, including bird communities. While the ecological attributes necessary to enhance the habitability of UGS for bird communities are relatively well known, an understanding of how variation of surrounding urban built form influences avian richness outcomes in these spaces is less understood. As new urban areas continue to develop and UGS become increasingly important habitat areas, urban designers and planners will need a better understanding of the ways in which urban built form patterns influence avian biodiversity outcomes in UGS. To that end, this study investigates this relationship using high resolution land cover data, building LiDAR data, and twenty years of bird occurrence data from the eBird community science program in well-surveyed UGS in Los Angeles, California. Results confirm previous findings that an increase in UGS size is associated with more avian richness. Interestingly, both multivariate regression models, urban form metrics and site-level metrics, performed well in predicting avian richness. Moreover, an increase in the following urban built form metrics, Aggregation Index and Landscape Shape Index, were associated with higher levels of richness, suggesting that more compact and complex shape building patterns support better avian richness outcomes in UGS. These findings contribute to a more complete understanding of how urban built form patterns influence avian biodiversity outcomes and inform urban planning and design of new urban areas aiming to maximize ecological potential for avian biodiversity conservation.

The thesis of Morgan Rogers is approved.

Kirsten Schwarz

Paavo Monkkonen

Thomas Gillespie

V. Kelly Turner, Committee Chair

University of California, Los Angeles

2021

Table of Contents

List of Figures

List of Tables

Acknowledgments

I would like to express my sincere gratitude to my thesis advisor, Dr. V. Kelly Turner, for her continuous support and invaluable mentorship. Additionally, I would like to express gratitude to my thesis committee for their insightful comments and suggestions. Also, I would be remiss if I did not thank Zhiyuan Yao for her help troubleshooting R code for hours. I deeply appreciate Loyola Marymount University Center for Urban Resilience sharing their land cover and tree canopy assessment of Los Angeles. Finally, I would like to thank my family and friends for believing in me and for their unwavering encouragement.

I. Introduction

By 2100, the impact of land use change globally on biodiversity may be more significant than climate change as urban land is projected to triple between 2000 and 2030 (Haines-Young, 2009; McDonald et al., 2020). Conversion of previously undeveloped land to urban land uses causes biodiversity loss via fragmentation or loss of habitat at both local and regional scales (M. F. J. Aronson et al., 2014; Grimm et al., 2008a). Despite these negative impacts, urban centers in some cases support more species than natural reference systems, including endangered and threatened species (M. F. J. Aronson et al., 2014; Grimm et al., 2008b; Ikin et al., 2013; Ives et al., 2016). This not only makes urban areas important for biodiversity conservation, but demonstrates that urban form is not homogenous and that biodiversity loss from urbanization is not inevitable.

The pattern and form of new urban areas will have profound impacts on habitats around the world, but increased knowledge on the ways in which urban built form can support biodiversity will help shape more positive outcomes. Urban green spaces (UGS) have played a critical role in serving as habitat in urban areas and in creating landscape connectivity to natural reference systems (Amaya-Espinel et al., 2019; Beninde et al., 2015; Hostetler & Holling, 2000; La Sorte et al., 2020). The role of urban built form and how variation in its patterns may differentially impact avian communities in UGS at the intra-urban scale, however, is relatively unknown, posing a challenge for urban design and planning practitioners seeking to conserve avian communities.

This black box in urban design and planning is in part due to the complex interaction and feedbacks between social and ecological components of urban systems, making it challenging to disentangle the intricate relationship between urban form and avian biodiversity (Grimm et al., 2013; von der Lippe et al., 2020). How the form and pattern of cities modify urban avian biodiversity patterns is key to informing conservation efforts in cities. Urban design intervenes in different urban systems by structuring and shaping urban space through built form; therefore, urban designers seeking to conserve urban avian biodiversity need a greater understanding of how urban built form influences avian biodiversity. This study seeks to address this gap by contributing a more complete understanding of how urban built form influences avian biodiversity outcomes in UGS which are important habitat areas in cities.

Avian species are useful and critical to study for two reasons: there are more of them where there is an increase presence of vegetation and other taxonomic groups of diversity, making them an applicable indicator for biodiversity, and they allow for an understanding of large ecosystem health as they are particular sensitive to environmental changes and are more able to move across landscapes compared to other taxa (Alvarez et al., 2015; Blair, 1999; Kati et al., 2004; Mikusiński et al., 2001). UGS serve as critical habitat for avian communities in cities, providing space for breeding, shelter, and food (Amaya-Espinel et al., 2019; Ikin et al., 2013). A wealth of studies analyzing intra-urban biodiversity variation, including avian biodiversity, have generally found that increases in patch area and corridors that create landscape connectivity to natural reference systems have the most positive relationships with biodiversity in cities, followed by vegetation structure (Beninde et al., 2015). A recent study of avian species richness in New York's UGS confirmed that UGS size has the most significant impact on avian richness outcomes (La Sorte et al., 2020). Another study looking at both avian richness and abundance in UGS in Santiago, Chile, specifically the impacts of urban density, found that increased density negatively influenced bird community outcomes in UGS (Amaya-Espinel et al., 2019). Taken together, these findings suggest that UGS cannot be seen as a panacea, and considering surrounding urban form is critical to ensuring urban avian biodiversity conservation.

Analysis of urban form configuration has largely been measured in two-dimensional (2D) space due to challenges of data acquisition and storage (Wentz et al., 2018). New methods for quantifying three-dimensional (3D) urban form have been developed to examine both the dimensionality and spatial pattern of urban patterns. Urban ecosystem scholars suggest that configuration, both two and three dimensional space, are necessary to characterize urban form, otherwise the assumption is that cities exist on a homogenous flat plane (Wentz et al., 2018). Researchers from the Academy of Sciences in Shenyang, China, have developed a method for calculating landscape metrics for 3D urban built form pattern recognition, which they tested on seven cities in China with 87 percent classification accuracy, proving it possible to capture both 2D and 3D urban form and accurately characterize cities through these metrics (M. Liu et al., 2017). Additionally, researchers have demonstrated that quantifying urban form both from a 2D and 3D perspective has implications on urban flight corridors for migratory birds as well as acoustic transmission, which can impact ability to communicate, altering avian behaviour (Z. Liu et al., 2020; Warren et al., 2006). Arguably, capturing both 2D and 3D measures of urban built form surrounding UGS will provide greater insight into the influence of urban built form patterns on bird community outcomes in UGS.

This study investigates the influence of urban built form on bird communities in UGS using both 2D and 3D measures of urban form configuration, while controlling for variables that are typical drivers of avian biodiversity: size of UGS, vegetation structure, and landscape connectivity. Using high resolution satellite imagery, community science bird occurrence data, LiDAR derived building data, and the California Protected Areas Database GIS layer, urban built form, UGS, and avian richness metrics were quantified using geospatial and machine learning methods. These data were then compared using multivariate regression analysis to understand the level of influence of urban built form patterns on avian biodiversity outcomes in Los Angeles's UGS.

Study results indicate that the configuration of the urban built form surrounding UGS influences avian richness outcomes. High aggregation of buildings and shape complexity are positively associated with richness outcomes whereas higher variability in building heights are negatively correlated, meaning that more compact building configurations with complex shapes and less variability in height are associated with higher avian richness in UGS. I will explore the underlying mechanisms that explain this relationship in my next study and hypothesize that these urban built form attributes aid in increasing ecological connectivity via urban flight corridors. The results of this study contribute a better understanding of the influence of urban built form patterns on avian species richness in UGS, specifically that more compact urban built forms with complex shapes support avian richness in UGS.

II. Literature review

A. Urban biodiversity conservation: plans and policies

Municipal urban biodiversity conservation plans have proliferated in recent years. Biodiversity refers to the variety of ecological elements in an ecosystem and is a measure of variation at the genetic, species, and ecosystem level (i.e. different plants, animals and other species in a given habitat at a particular time). Biodiversity in cities consists of a mix of native, invasive, and introduced species. When looking at 135 plans from 40 cities globally, two of the most common attributes in these plans were habitat conservation and ecological connectivity (Nilon et al., 2017). There is an emphasis placed on protecting and creating a network of greenspaces that can provide habitat for species, while providing local ecosystem services to residents. Typically, conservation plans aim to conserve native biodiversity as they provide the most benefits to local ecosystems (I. T. Brown, 2019).

Many of these plans incorporate ecological principles to enhance habitat based on findings from socio-ecological scholarship. Biodiversity sensitive urban design has become an increasingly researched topic, providing numerous studies demonstrating positive biodiversity outcomes when urban design enhances green space and landscape connectivity (Beninde et al., 2015; Hostetler et al., 2011; Ikin et al., 2013). Socio-ecological scholars have put together frameworks for biodiversity sensitive design to help inform planners and policy makers on how to incorporate these ecological principles into actual design plans, particularly around urban green spaces (Garrard et al., 2018). Considering the robust findings on positive biodiversity outcomes in relation to enhanced green space and landscape connectivity and active efforts made by socio-ecological scholars to translate this knowledge into actionable plans, it makes sense that most urban biodiversity conservation plans focus on these principles in their management plans.

Urban biodiversity conservation has become more of a focus in sustainability plans in part due to a decadal decline in biodiversity. Research has shown that in some instances urban centers support more species than natural reference systems, including endangered and threatened species, making urban areas important in biodiversity conservation (M. F. J. Aronson et al., 2014; Grimm et al., 2008b; Ikin et al., 2013; Ives et al., 2016). However, conserving urban biodiversity is not just important for biodiversity conservation, it also provides multiple benefits for people.

Recent research has demonstrated the importance of urban biodiversity as one of the elements that supports human-well being and ecosystem services. This study does not examine human well-being and ecosystem services directly, but the biodiversity that underpin them. There is strong evidence that suggests biodiversity and ecosystem function are connected; in other words, a decrease in biodiversity can reduce ecosystem function and as a result, impact an ecosystem's ability to deliver services (Sandifer et al., 2015). This connection has been demonstrated in several studies, revealing positive links between biodiversity and ecosystem services, therefore making biodiverse urban systems important for urban sustainability (Schwarz et al., 2017; Tratalos et al., 2007). Ecosystem services provide us benefits such as oxygen, clean air and water, pollination of plants, pest control, and much more (Sandifer et al., 2015). Therefore, a reduction in urban biodiversity impacts human well-being (C. Brown & Grant, 2005; Fuller et al., 2010; Sandifer et al., 2015). Given the benefits provided by urban biodiversity and calls to conserve biodiversity, many municipalities have started to include urban biodiversity conservation plans in their sustainability plans.

Many advances have been made in understanding ecological attributes that provide functional habitat in urban areas (Beninde et al., 2015; Threlfall et al., 2016). However, there is less insight into how urban built form configuration may differentially impact biodiversity outcomes in urban green spaces in part because it is difficult to isolate the various interactions between social and ecological components due to the complexity of urban systems. While great strides have been made to understand how urban built form, both composition and configuration, influences other ecosystem service outcomes like microclimate regulation, there is a dearth of empirical studies analyzing this relationship to biodiversity outcomes, especially when considering spatial configuration (Andersson & Colding, 2014; Connors et al., 2013; McPhearson et al., 2016; Zhou et al., 2011). Understanding how urban form, building patterns in particular, influence ecological attributes of a city, such as biodiversity, could help inform future sustainable designs.

B. Social-environmental systems scholarship: links between urban form and biodiversity outcomes

This study builds principally off two bodies of literature within what is typically referred to as social-environmental systems (SES) scholarship: land system science (LSS) and urban ecology (UE). Both fields study cities as SESs with complex interactions and feedback loops between social and environmental components within those systems (Grimm et al., 2013; Verburg et al., 2015). Both fields explore the nexus of urbanization and biodiversity with the goal of providing insights on paths forward for sustainable urban land stewardship and landcover change. However, while both fields are focused on the same nexus, they approach the topic from different sets of methods and at different scales. LSS uses remote sensing to monitor land characteristics and change at the regional scale. UE focuses on site level assessments of urban land conditions and the effectiveness of sustainable solutions. By using remote sensing and field survey approaches to compare regional and site level analyses, new insights can be gleaned from a multi-scalar analysis on what supports or doesn't support urban biodiversity, as it captures and differentiates interactions that are happening across urban scales.

According to several LSS studies, urban land-cover change has the greatest impact on biodiversity through the loss and fragmentation of habitats (Güneralp & Seto, 2013). To assess these patterns, LSS looks at the relationship between land use, land cover, biodiversity, and the output of ecosystems (Haines-Young, 2009). Land cover is the physical characteristics of the surface of the earth, which are produced through biotic and abiotic features (i.e. grass, asphalt, trees, bare ground, water, etc.). Land-use, on the other hand, is determined by activities and management of that land by people (i.e. agriculture, industrial, residential, etc.). Land and ecosystem function refers to the potential capacity of the land and ecosystems to carry out physicochemical and biological processes that maintain terrestrial life. Alternatively, ecosystem services are the final contributions that ecosystems make to human beings such as food, clean air, clean water, etc. Understanding the quantity, quality and spatial configuration of different aspects of land use and land cover has been the principal methodological approach to understanding impacts to biodiversity and ecosystem function.

One of the challenges the LSS field has faced in trying to understand the impacts of land-use and land cover change to biodiversity is that biodiversity can be measured in many ways. A multitude of indicators have been developed due to lack of consensus in the field, making it difficult to consistently analyze the relationship between land-use and cover to biodiversity (Haines-Young, 2009). However, vegetation and avian species have been typical indicators for biodiversity as they are correlated with other taxa. For example, Gillespie et al. (2008) highlighted several studies as examples of successfully using vegetation maps as proxies for the distribution of birds (Peterson et al., 2006), herpetofauna (Raxworthy et al., 2003), and insects (Luoto et al., 2002). In addition, a number of studies have revealed that presence of avian species is correlated with vegetation and other taxonomic groups' diversity and is often used as a proxy for biodiversity (Blair, 1999; Kati et al., 2004; Mikusiński et al., 2001). In short, avian species are a useful indicator of biodiversity as they are correlated with vegetation and the presence of other species.

Beyond choosing a consistent indicator for biodiversity, long term monitoring of changes can be difficult because more accurate surveys conducted on the ground are expensive and time intensive. Community science platforms such as iNaturalist and eBird can help fill in gaps in data. These data present their own challenges as they are unstructured, so both sampling sizes and time of capture can vary, making it difficult to assess population change trends (Bayraktarov et al., 2019). However, assessments of survey efforts and machine learning methods help address these limitations (Lobo et al., 2018; Strimas-Mackey, Hochachka, et al., 2020).

Regardless, remote sensing has been a helpful remedy to the limitations of ground survey methods by allowing for easier tracking of biodiversity health over long periods of time. Significant advances have been made in remote sensing by combining data from multi-passive and active sensors, allowing for better models of species richness, alpha diversity, and beta diversity (Gillespie et al., 2008). For example, a recent study was conducted using very high resolution satellite data and the Normalized Difference Vegetation Index (that allowed for the identification and classification of urban vegetation). However, there are limits to what sensors can tell us about biodiversity based on the spectral signatures (Hashim et al., 2019). Getting accurate readings of biodiversity via remote sensing has vastly improved, allowing for time series and comparative analysis, but is still limited at the urban scale due to limitations of sensors and the disparate indicators of biodiversity used across the field.

Nevertheless, improvements in remote sensing technology have allowed land system scientists to characterize the composition and spatial structure of the mosaic of land units, also known as land systems architecture, providing unique insights. LIDAR in particular, has enabled the improved characterization of land system architecture by characterizing vegetation structure and urban morphology. These modeling tools allow for the evaluation of tradeoffs between alternative composition and spatial structures of landscape (B. L. Turner et al., 2013). Recent studies using these methods have revealed that urban land is heterogeneous and produces varying ecosystem outcomes such as microclimate regulation due to this heterogeneity (Connors et al., 2013; Galletti et al., 2019). These findings suggest that alternate urban built forms may also produce variation in biodiversity outcomes since there is strong evidence suggesting that ecosystem function is connected to biodiversity (Sandifer et al., 2015). LSS provides a useful method for characterization of the composition and configuration of the built environment, but thus far it has been primarily used in microclimate studies at regional scales (McPhearson et al., 2016).

While the LSS field has used remote sensing as a primary method, urban ecology, on the other hand, integrates theory and methods from natural and social sciences, primarily conducting site level field surveys, though use of remote sensing methods has increased in recent years. UE studies patterns and processes in urban ecosystems typically at the intra-urban scale (Zhao & Wentz, 2020). Urban ecologists view cities as heterogeneous, dynamic landscapes and seek commonalities among city ecosystems to understand how context shapes the socio-ecological interactions within them (Grimm et al., 2013).

Urban ecologists, like land system scientists, have found that urbanization has negative impacts on biodiversity, mostly due to habitat loss and fragmentation (Elmqvist et al., 2016). Expansion of urban areas has a direct impact on native species dispersal through changes in habitat configuration and connectivity. As such, urban ecologists are interested in establishing management practices for biodiversity corridors in urbanizing regions (Forman, 2008). Many studies have gathered localized flora and fauna survey data as well as other biophysical data in order to understand urbanization processes and their impacts on biodiversity and ecosystem function. By understanding the biogeophysical context of an urban area, researchers can determine how biodiversity responds to rapid urbanization (Schewenius et al., 2014).

From decades of research, it is now well known that diversity, structure, and distribution of vegetation are impacted by fragmentation of natural patches (Brothers & Spingarn, 1992; LEVENSON, 1981) and that landscape connectivity can either support or limit the movement of resources and species through natural patches (M. Turner & Gardner, 1991; Walz & Syrbe, 2013). As urban built form continues to expand and change, it impacts connectivity and patch size directly through the modification of the landscape. Past urban-to-rural gradient studies, a common method of study used by urban ecologists in the past, found that species richness declines from the urban fringe towards the urban core (McKinney, 2002; Pickett et al., 2001). In addition, scholars have also discovered a trend of urban biotic homogenization due to land-uses homogenizing across cities, impacting both biodiversity and ecosystem service function at local, regional and continental scales (Carpenter et al., 2009; Elmqvist et al., 2016; Groffman et al., 2014).

Despite observed negative effects of urbanization, studies show that cities are typically built in areas with high species richness as well as fertile soil, making urban areas ripe with potential for biodiversity (Alberti, 2010). In some cases, urban areas may even have richer biodiversity compared to their rural counterparts due to human influence of increased irrigation, planting exotic plant species in urban areas, or supply of food (Schewenius et al., 2014). Indeed, loss of habitat and biotic homogenization have negative consequences for biodiversity, but the ways in which cities grow and their various forms are not entirely uniform. Urban areas continue to have heterogeneous land cover and land uses despite trends of land use homogenization. The heterogeneous nature of cities and observed biodiversity richness in some areas of cities indicates that not all urban areas are equal and can produce varied outcomes, some of which are positive.

In attempt to untangle the complexity of urban systems, scholars have emphasized the importance of adopting a multiscale and multi-site approach to understanding socio-ecological system dynamics, suggesting that biodiversity outcomes likely vary not only by alternate urban form features, but at scale with different interacting social drivers (Roy Chowdhury et al., 2011). A recent framework has filled a gap in the field of urban ecology by providing a framework that spatially addresses the interactions between social dynamics and ecological processes that impact biodiversity across scales (Andrade et al., 2020). An example of a study taking into account multiscalar impacts looked at urban butterfly abundance and richness patterns at 5m and 25m resolution. They found that spatial scale had a significant impact on the correlation of the model to biodiversity outcomes (Hazell & Rinner, 2020). The results showed that urban butterfly biodiversity was more highly correlated at the 25m resolution model, compared to the finer scale resolution model. Thus, it is important to analyze biodiversity outcomes and drivers of those outcomes at various scales, as these relationships may change based on local versus neighborhood versus landscape scale.

While urban ecologists have not been able to ascertain differential impacts of alternate built environment configurations on biodiversity outcomes, they have made significant progress in linking different features of urban form to biodiversity outcomes. A seminal study aggregating empirical studies on urban ecosystem function and biodiversity paired with a case study of the Puget Sound metropolitan region suggested that alternate urban development patterns produced differential ecological effects (Alberti, 2005). The paper proposed future empirical research to identify strategies to minimize urbanization impacts on ecosystems.

More than a decade later there are numerous empirical studies to draw from exploring different aspects of urban form and its relationship to urban biodiversity outcomes. An area of focus has been urban density and biodiversity outcomes, which has produced varied results. For example, one study of five UK cities found that while ecosystem service output declines with increasing urban density, there was a lot of variation, suggesting that a simple measure of density does not capture the full relationship between urban form and ecosystem service or biodiversity outcomes (Tratalos et al., 2007). The researchers concluded that there was potential to maximize ecological potential at any given density. Another study also concluded that using the percentage of built up surface area, a measure often used to calculate urban density, is a poor indicator of urban biodiversity potential (Brunbjerg et al., 2018). Thus, more nuanced measures to characterize the built environment are required to capture dynamic interactions between the built environment and ecological processes.

Although research on urban density has had mixed results, decades of research on green infrastructure or different land covers that provide habitat in cities is rather conclusive. The size of a patch area and the level of connectivity from these patches to natural reference systems provided by green corridors have the strongest positive effects on biodiversity, complemented by vegetation cover and structure (Beninde et al., 2015; Lynch, 2019; Plummer et al., 2020; Schütz & Schulze, 2015). There is a lack of empirical studies on the effects of urban built form configuration and composition on biodiversity outcomes and at the same time a lot of research demonstrating the importance of urban green spaces and vegetation cover for biodiversity (Norton, 2016). Given this imbalance in research, it makes sense that the majority of urban biodiversity conservation plans eschew a consideration of built form and instead focus on management of urban green spaces and the creation of ecological corridors. I intend to develop new insights on what supports or doesn't support urban biodiversity, specifically avian biodiversity, from the perspective of urban built form, by combining geospatial methods from LSS and site-level survey analysis methods from urban ecology.

C. Urban avian biodiversity: impacts of green infrastructure and urban built form

According to current research, the world's cities are dominated by native avian species and have not homogenized despite overall density of species declining in urban regions (M. F. J. Aronson et al., 2014; Lepczyk et al., 2017). As avian species persist and even thrive in cities despite the multitude of disturbances and predators, many questions remain on how urban form and location influence avian richness (Marzluff, 2017). Several studies have linked various features of urban form, both built and green, to avian biodiversity outcomes (see Table 1 for a summary of studies). Most research is centered around enhancing green infrastructure and urban green spaces to create functional habitat and landscape connectivity via green corridors to natural reference systems.

Research on the benefits of urban green space and vegetation cover for avian biodiversity have demonstrated consistent positive outcomes. In general, increasing the size of green spaces, vegetation cover both in diversity and structural complexity, having more native vegetation, and enhancing green corridors to create landscape connectivity all have positive relationships to avian diversity (refer to table 1 for list of studies). One study did find that an increase in green infrastructure does not necessarily provide benefits for avian diversity unless it provides landscape connectivity (Strohbach et al., 2013). These studies suggest that avian biodiversity conservation in urban areas should focus on increasing habitat spaces and landscape connectivity as they are primary drivers.

However, these attributes are not the only drivers of avian biodiversity outcomes as suggested by studies analyzing various attributes of the built environment. The results have been mixed on the influence of various attributes of urban built form on avian biodiversity except for the impacts of roads, which consistently have a negative relationship to avian biodiversity outcomes (see table 1 for list of studies). There has been less of a focus on urban built form, especially from a 3D configuration perspective, whereby the dimensionality and spatial pattern of urban built form is taken into consideration. When studies do take into consideration 3D metrics it is usually only building height, which has had mixed results on the impacts of avian biodiversity. A simple measure of building height may not be sufficient to analyze its relationship to avian biodiversity outcomes as this metric is only one aspect of mapping urban flight corridors and understanding acoustic transmission, both of which can alter avian behavior (Z. Liu et al., 2020; Warren et al., 2006). Most studies that consider urban built from have focused on 2D measures of density, i.e. degree of urban built up area. These studies have found mixed results in terms of the direction of the relationship to avian richness, from positive, to negative, to mixed, and even neutral, suggesting that more nuanced measurements of urban built form may need to go beyond 2D measures and ranges of density.

Table 1: Summary of findings on the influence of different aspects of urban built form and urban green space on avian biodiversity outcomes. The relationship described: as "X" variable increases there is a "positive"(blue), "negative"(red), "mixed"(purple), or "neutral" (yellow) relationship to avian richness.

	Urban built form					Urban green space						
Study	Urban landcover/ land-use	Compact urban form	Sprawl urban form	Building height	Road density	UGS size	Green corridors	Green infrastructure	Vegetation cover	Vegetation diversity	Vegetation structural complexity	Native vegetation
(Amaya-Espinel et al., 2019)		*										
(Loss et al., 2014)												
(MacGregor-Fors et al., 2017)												
(Trollope et al., 2009)												
(White et al., 2005)												
(Beaugeard et al., 2020)												
(La Sorte et al., 2020)												
(Chowdhury & Sen, 2019)												
(Lindenmayer et al., 2020)												
(Kaushik et al., 2020)												
(Andersson & Colding, 2014)												
(Sushinsky et al., 2013)												
(Hostetler & Knowles-Yanez, 2003)												

(M. F. J. Aronson et al., 2014)												
(Marzluff, 2017)			**									
(Strohbach et al., 2013)								***				
(Ikin et al., 2013)												
(Dale, 2018)												

*Found compact urban form at the intraurban scale to have negative relationship to avian biodiversity, but a positive relationship at the regional scale

**Moderate levels of suburban and exurban development have a positive relationship to avian biodiversity

***Green infrastructure only has a positive effect on avian biodiversity if it enhances connectivity (i.e. creates a green corridor) to natural reference systems

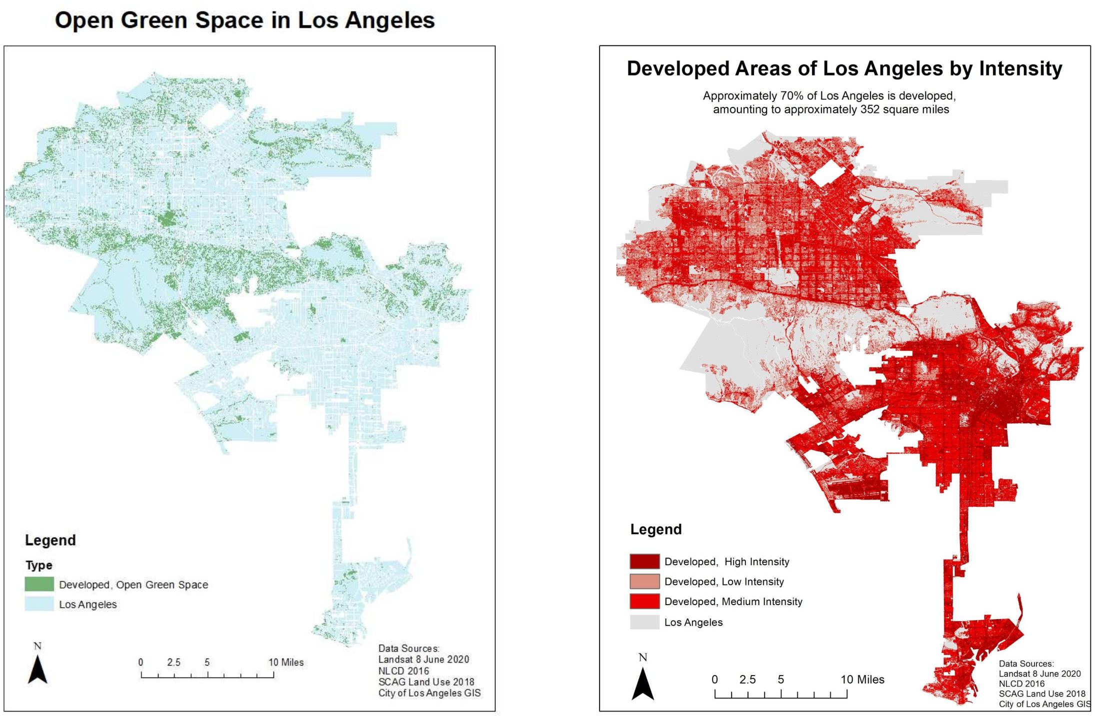
III. Methods

I used a combination of geospatial and machine learning methods to characterize urban built form and UGS site-level attributes, to model avian species distribution across UGS, and to analyze the relationship between urban built form patterns to avian richness outcomes within UGS in Los Angeles, California. High resolution satellite imagery, LiDAR building data, and twenty years of bird occurrence data from the eBird community-science program were used to estimate bird species richness within well-surveyed UGS and to analyze outcomes associated with urban built form variables. I calculated urban built form and landscape metrics using Fragstats, R, and ArcGIS software. Avian species richness was calculated using the software Modest R and the R package KnowBR. Ordinary least-squares regression was used to analyze the relationship between each metric and avian richness. Finally, I used multivariate regression models to compare the level of influence and predictive power of avian richness in UGS between urban built form metrics and site-level UGS metrics.

A. Study Area

This study examines avian richness in UGS in the City of Los Angeles, California. Los Angeles has approximately 11 percent open green space (Figure 1), and has 1,255 general green spaces listed by the California Protected Areas Database (CPAD) (GreenInfoNetwork, n.d.). Los Angeles is a Mediterranean city that is a designated “global biodiversity hotspot,” which means it has both high concentrations of biodiversity and highly threatened biodiversity (Brown, 2019). The City has more than 150 threatened or endangered species (City of Los Angeles, 2006). At the same time, it is home to 3.99 million residents with a total area of 130,276 hectares and varying degrees of development intensity (Figure 1). Los Angeles is an ideal city to study given the wide range of development intensities and range of highly modified green spaces to natural remnant habitats. Moreover, the Los Angeles City Council passed Biodiversity Motion 25A in 2017 that resulted in a custom biodiversity tracking index to conserve biodiversity and the 2015 Sustainable pLAn has a goal of no-net-biodiversity loss by 2035 (Brown, 2019). Responding to these efforts, a follow-up report was released in 2020 with an updated biodiversity index and management framework with local case studies demonstrating successful management of particular areas (Zaldivar, 2020). Additionally, Los Angeles has superior bird count coverage in the community science platform, eBird, leading all U.S. counties in number of bird count checklists (Los Angeles Leads All U.S. Counties in Nationwide Bird Count, 2015).

Fig. 1: Through a comparison of the right and left maps, it is clear that the majority of UGS tend to be on the periphery of urban development. However, there are also a lot of UGS embedded in highly developed urban areas. Furthermore, as the the map on the right indicates, there is a range of development intensities, allowing for a comparison of the influence of different development intensities on avian richness outcomes in UGS in Los Angeles



Data sources: National Land Cover Database (NLCD) 2016, Southern California Association of Governments Land Use dataset from 2018, California Protected Areas Database, and the City of Los Angeles GIS hub.

B. Data

I extracted Los Angeles UGS from the polygon GIS data layer, the CPAD (GreenInfoNetwork, n.d.), at the “super units” level, which captures the boundary of the whole space instead of individual subunits. I then filtered the layer to only include UGS that are classified as a park. This layer was used to capture the area of each selected UGS, longest distance of each selected UGS, distance to nearest green space, and to analyze site-level characteristics by clipping land cover data to UGS boundaries.

I extracted land cover and landscape metrics, such as percent tree canopy, percent grass, percent shrub, percent soil, percent water, percent building, percent road, percent paved, percent impervious cover, from a data layer generously shared by the Loyola Marymount

University Center for Urban Resilience. The GIS layer was derived from the 10cm resolution Los Angeles Region Imagery Acquisition Consortium (LARIAC 5) land cover and tree canopy assessment captured in 2017.

Both 2D and 3D architectural data for urban built form metrics were extracted from LARIAC 5 Buildings 2017 GIS layer provided by the Los Angeles County Open Data Hub. The layer captures over 3,000,000 building outlines in Los Angeles County, including building height and building area using stereo imagery methods.

I acquired avian occurrence data from the eBird community science dataset provided by The Cornell Lab of Ornithology (Sullivan et al., 2014). These data are structured around a checklist that represents observations from a single sampling event. Each checklist includes species observed, the number of each species seen, location and time of occurrence, survey protocol used, and measures of sampling effort used to collect data. Both the eBird Basic Dataset (EBD) and Sampling Event data (SED) were downloaded for all years combined, 2002 to 2021, for Los Angeles County (the program was initiated in 2002). Only complete checklists were selected, allowing for identification of species that were not detected, instead of not reported. The SED provides checklist data to capture sampling effort variables. All grouped checklists were combined into single checklists to remove any duplicates.

C. Survey completeness and avian richness

I pre-processed avian occurrence data using the R package “AUK” developed by The Cornell Lab of Ornithology to easily subset EBD data into a manageable size for analysis in R (Strimas-Mackey, Miller, et al., 2020). The AUK package was used to subset occurrence data falling within the City of Los Angeles extent, to combine grouped checklists into single checklists, and to combine occurrence and sampling effort data, per best practices (Strimas-Mackey, Hochachka, et al., 2020). I further filtered the dataset to only include occurrences with sampling events falling within boundaries of UGS using a spatial intersect join function from the

R package SF (Pebesma, 2021). All incidental and stationary sampling protocol records were kept. Per methods used by La Sorte et al (2020), area sampling protocol records were further filtered to only include records where the sampling effort area did not exceed the maximum area of the UGS, and travel sampling protocol records were filtered to only include records where sampling effort distance did not exceed the maximum length of the UGS. This resulted in 23,754 EBD records and 75 UGS.

In order to select only well surveyed UGS for avian richness modeling and analysis, I calculated survey completeness for each UGS. Survey completeness is the percentage of observed species richness captured by predicted species richness (Lobo et al., 2018). This calculation is an estimate of sampling events' ability to capture all species expected to be present at a given time and location (Colwell et al., 1994; La Sorte et al., 2020). To prepare the data for survey completeness analysis, I used ModestR to create a species occurrence format that takes semi-structured data, like EBD, and gives the number of records per species within a set of spatial unit parameters (*ModestR Software for Species Distribution Data Management*, n.d.). I used the R package, KnowBR, to calculate survey completeness for each UGS by estimating the species accumulation curve using the estimator of (Ugland et al., 2003) and the slope between number of actual observed species and number of database records (*KnowBR*, n.d.). Per methods used by La Sorte et al. (2020) that were defined by Lobo et al 2018, poorly surveyed areas were removed. Poorly surveyed areas meet the following three parameters: (1) the ratio between number of database records and observed species is <3, (2) the species accumulation curve slope is >0.3, and (3) survey completeness is <50 (La Sorte et al., 2020; Lobo et al., 2018). After taking these factors into account, 24 UGS were considered to be well surveyed. Avian species richness was calculated for each UGS. Heidelberg had the lowest species count at 25 and Ken Malloy Harbor Regional Park had the highest species richness at 145 different species.

D. Data analysis

I extracted site-level metrics for UGS from a data layer provided by Loyola Marymount University Center for Urban Resilience, which is a GIS layer derived from the 10cm resolution LARIAC 5 land cover and tree canopy assessment conducted in 2017. Size of the UGS was extracted from the California Protected Areas Database GIS layer. Distance to nearest green space was used as a proxy for landscape connectivity and calculated using the Near Analysis ArcGIS tool (Amaya-Espinel et al., 2019). All calculations were conducted in ArcGIS. A description of each of the 10 metrics used to quantify UGS site-level metrics is provided in Table 2.

To quantify the patterns of surrounding urban built form, a 200 meter buffer was created around each UGS. This buffer size was selected as it is the average territory range of the most common avian species found in Southern California; and therefore, where urban built form is likely to have more of an influence on avian community in UGS (Amaya-Espinel et al., 2019). I extracted building data falling within the buffer surrounding each park from the LARIAC 5 Buildings 2017 GIS layer. To capture the pattern of urban built form, the configuration, both the dimensionality and spatial pattern, of buildings was quantified using 12 landscape metrics developed to accurately depict 2D and 3D aspects of architectural patterns (M. Liu et al., 2017). In order to capture the diversity in building height, buildings were divided into six categories by size: Bungalow (0-4m), Low Building (4 to 10m), Multi-story building (10 to 19m), Middle-height building (19 to 30m), Tall building (30 to 100m), and Super tall building (>100m) (M. Liu et al., 2017). I calculated all of the metrics in ArcGIS except for Landscape Shape Index, Patch Density, Aggregation Index, and Shannon's Diversity Index. In order to calculate these landscape metrics, the GIS layer was rasterized to 1m resolution and exported to the software Fragstats 4.1 for analysis whereby each building was considered as one patch (McGarigal & Marks, 1995). For a description of each of the 12 metrics used to quantify urban built form refer to Table 1.

Finally, I quantified urban built form patterns for each 200 meter buffer surrounding a UGS and compared them to site-level characteristics quantified for each UGS to understand the direction and strength of influence of each on avian richness. I used Ordinary Least Squares (OLS) Regression to analyze the relationship between each metric and annual avian richness outcomes. Two multivariate linear regressions were used as a comparison of avian richness predictive power between urban built form metrics and site-level UGS metrics.

Table 2: Description of metrics used to quantify urban built form and site-level characteristics of each UGS. Formulas and descriptions for urban built form metrics are from Liu et al. 2017.

Level	Variable measured	Description
Urban built form (200m buffer)	Number of buildings (NB)	Total buildings NB = N
	Number of tall buildings (NTB)	Total buildings exceeding 24m NTB = TB
	Tall building ratio (TBR)	Ratio between buildings exceeding 24m and total buildings HBR = N_h/N N_t: number of tall buildings N: number of total buildings
	Mean architecture projection area (MAPA)	Mean area of architecture projected vertically to floor MAPA = TAPA/N TAPA: total architecture projection area N: number of buildings
	Mean architecture height (MAH)	The average height of all buildings in study area MAH = $(\Sigma^{n}_{i=1} H_i) / n$ MAH: Mean architecture height
	Architecture height standard deviation (AHSD)	Variation degree of the buildings in study area AHSD = $\sqrt{(\Sigma \; ^{n}{}_{i=1}(H_i - MAH)^2)/n}$
	Building coverage ratio (BCR)	The degree of building diversity BCR = Σ^{F} / A F: the land area of building taken A: study area
	Floor area ratio (FAR)	Building area unit land area FAR = $(\Sigma^{n}_{i=1} (C \; x \; F)) / A$ C: number of floors F: area of building A: study area
	Landscape shape index (LSI)	The degree of landscape shape complexity LSI = E / min E E: total length of edge in landscape in terms of number of cell surfaces; includes all landscape boundary and background edge segments. min E: minimum total length of edge in landscape in terms of number of cell surfaces.

	Patch density (PD)	PD equals the number of patches in the landscape (100 hectares), reflects density of patches PD = $(N/A)(10{,}000)(100)$ N = total number of patches in the landscape. A = total landscape area (m^2)
	Aggregation Index (AI)	AI equals the number of like adjacencies involving the corresponding class, divided by the maximum possible number AI = $\left[\Sigma\ {}^{m}{}_{i=1} \left(g_{ii}/max \rightarrow g_{ii}\right)P_i\right](100)$ g_{ii} = number of like adjacencies (joins) between pixels of patch type (class) i based on the single- count method max-g_{ii} = maximum number of like adjacencies (joins) between pixels of patch type (class) i (see below) based on the single-count method P_i = proportion of landscape comprised of patch type (class) i
	Shannon's Diversity Index (SHDI)	SHDI equals minus the sum, across all patch types, of the proportional abundance of each patch type multiplied by that proportion SHDI = $-\Sigma^{n}_{i=1} P_i \times \ln P_i$ P_i = proportion of the landscape occupied by patch type (class) i
Urban green space (site-level)	Patch size (PS)	Patch area in hectares
	Distance to nearest green space (DNGS)	Distance in meters to nearest green space using Near Analysis tool in ArcGIS
	% Tree canopy (TC)	Percentage of land cover type in patch area %LCC = $(A_{LCC}/T_p)(100)$ A_{LCC} = Area covered by land cover class T_p = Total patch area
	% Grass (G)	
	% Soil (S)	
	% Water (W)	
	% Road (R)	
	% Paved (P)	
	% Shrub (SH)	
	% Pervious (PE)	
	% Impervious (IMP)	

IV. Results

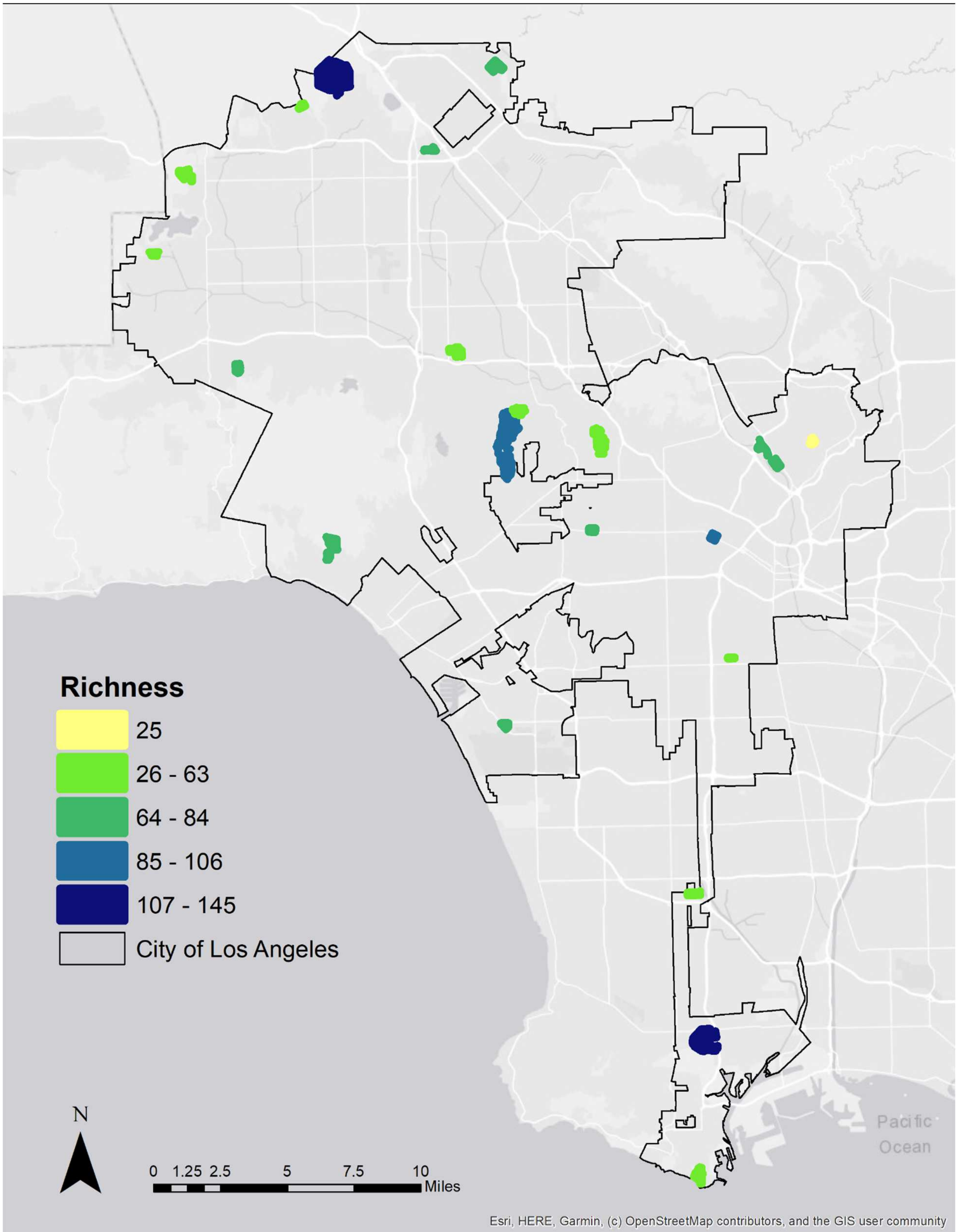
A. Predicted avian species richness

Out of the 75 UGS with EBD records, 24 UGS were categorized as well surveyed. The location of these UGS spanned North (6 UGS), South (5 UGS, West (5 UGS), and East (5 UGS) with a relatively even spread (Figure 2). Two parks (Cheviot Hills Park and Recreation Center and Woodley Avenue Park) needed to be removed due to lack of sufficient land cover data. The average species richness count was 73 with a standard deviation of 27.51 for the 22 UGS, indicating that the sample of UGS had a lot of variation in avian richness outcomes. Avian richness across UGS ranged from 25 to 145 different species with higher levels of richness associated with larger UGS (see Figure 2 and Table 3).

Table 3: KnowBR survey completeness and predicted species richness for the 24 UGS that met all three parameters: (1) the ratio between number of database records and observed species is <3, (2) the species accumulation curve slope is >0.3, and (3) survey completeness is <50 (La Sorte et al., 2020; Lobo et al., 2018). For output figures of all parks from KnowBR analysis refer to the Appendix.

UGS	Records	Observed Richness	Predicted Richness	Slope	Completeness	Ratio
Franklin Canyon Park	775	90	106.1	0.03	84.82	8.6
O'Melveny Park	12572	134	137.9	0	97.16	93.82
Ken Malloy Harbor Regional Park	608	113	144.8	0.06	78.04	5.38
Heidelberg Park	65	19	25.4	0.09	74.72	3.42
Angels Gate Park	500	43	47.7	0.01	90.2	11.63
Orcutt Ranch Horticultural Center Park	215	41	48.2	0.04	85.04	5.24
Roosevelt Memorial Park	113	37	51.5	0.06	71.84	3.05
South Los Angeles Wetlands Pocket Park	375	45	51.8	0.02	86.84	8.33
Coldwater Canyon Park	176	43	56.3	0.07	76.41	4.09
Van Nuys-Sherman Oaks War Memorial Park	197	45	58.2	0.07	77.35	4.38
Porter Ridge Park	198	46	59.5	0.08	77.25	4.3
Chatsworth Park South	142	43	59.9	0.09	71.76	3.3
Runyon Canyon Park	220	49	63.1	0.08	77.64	4.49
Brand Park	443	57	68.1	0.03	83.73	7.77
El Cariso Community Regional Park	236	54	68.6	0.08	78.72	4.37
Westchester Recreation Center	321	56	69.0	0.05	81.15	5.73
Temescal Gateway Park	397	60	71.0	0.02	84.55	6.62
Serrania Avenue Park	998	68	72.7	0.02	93.49	14.68
Rio de Los Angeles State Park	238	58	75.0	0.08	77.29	4.1
Hancock Park	419	60	79.9	0.06	75.12	6.98
MacArthur Park	1704	86	97.02	0.01	88.64	19.81
Lewis MacAdams Riverfront Park	279	65	83.7	0.06	77.69	4.29

Figure 2: Avian species richness by urban green space



B. Urban built form and landscape metrics for sample UGS

The average patch size of the 22 UGS was 43.23 hectares with a standard deviation of 68.06, showing a lot of variation across sample UGS being tested (Table 4). This allowed for an analysis of the impact of UGS size on avian richness. The top three most avian biodiverse UGS were on average 199.79 hectares and the least diverse were 15.29 hectares on average, suggesting that larger UGS are correlated with higher levels of avian richness. On average the UGS had 31.93 percent tree cover, but there was a lot of variation among UGS. Interestingly, the top three most diverse UGS had an average of 32.77 percent tree canopy coverage whereas the three least diverse had 46.83, suggesting that tree canopy coverage isn't necessarily a strong predictor of richness in UGS. In general, there was a lot of variation in metrics among UGS except for the ratio between the total number of buildings and tall buildings. There were not many tall buildings at sample UGS sites and the average building height was 6.5 meters, which is typical of Los Angeles.

Table 4: The mean and standard deviation of each variable

Level	Variable measured	Average and standard deviation (metric)
Urban built form	Number of buildings (NB)	424.56 ± 209.96
	Number of tall buildings (NTB)	1.55 ± 3.83
	Tall building ratio (TBR)	0.01 ± 0.02
	Mean architecture projection area (MAPA)	326.61 ± 213.90
	Mean architecture height (MAH)	6.50 ± 1.81
	Architecture height standard deviation (AHSD)	67.72 ± 2.68
	Building coverage ratio (BCR)	19.29 ± 10.20
	Floor area ratio (FAR)	44.53 ± 48.63
	Landscape shape index (LSI)	12.74 ± 3.92
	Patch density (PD)	957.78 ± 1317.75
	Aggregation Index (AI)	97.00 ± 2.07
	Shannon's Diversity Index (SHDI)	0.57 ± 0.27
Urban green space	Patch size (hectares)	43.23 ± 68.06
	Distance to nearest green space (meters)	1046.45 ± 1375.50

	% Tree canopy	31.93 ± 20.53
	% Grass	21.23 ± 18.80
	% Soil	23.62 ± 17.05
	% Water	2.19 ± 5.87
	% Road	1.26 ± 1.88
	% Building	3.07 ± 5.54
	% Paved	10.45 ± 9.67
	% Shrub	6.26 ± 8.21
	% Pervious	85.22 ± 14.47
	% Impervious	14.78 ± 14.47

C. OLS and multivariate regression modeling

Only two metrics were statistically significant when assessing each metric's relationship to annual avian richness outcomes in UGS using OLS regressions, suggesting that these two metrics were significant drivers of avian richness outcomes (see Table 5). Patch size (P = 0.0003719) was positively associated with richness, indicating that as patch size increases it has a positive influence on avian richness outcomes. Also, water cover had a statistically significant relationship (P= 0.02631), suggesting that presence of water cover supports waterfowl, increasing avian richness in UGS.

Table 5: Results from OLS regression analysis. Statistically significant P values are in bold.

Variable	**Coef.**	**t**	**P**	**R2**
Number of Buildings (NB)	-0.003054	-0.048	0.9622	0.0001154
Number of Tall Buildings (NTB)	0.4372	1.455	0.1612	0.09571
Tall building ratio (TBR)	5.0244	1.154	0.2619	0.06247
Mean architecture projection area (MAPA)	0.09101	1.394	0.1787	0.08852
Mean architecture height (MAH)	0.8304	0.802	0.4318	0.03118
Architecture height standard deviation (AHSD)	0.5158	0.882	0.3882	0.03746
Building coverage ratio (BCR)	-2.129	-0.724	0.4773	0.02556
Floor area ratio (FAR)	0.3512	0.281	0.7816	0.003933

Landscape shape index (LSI)	0.1087	0.175	0.863	0.001526
Patch density (PD)	-0.02488	-1.092	0.2878	0.05627
Aggregation Index (AI)	1.975	0.616	0.5449	0.01861
Shannon's Diversity Index (SHDI)	-1.556	-0.887	0.3857	0.03784
Patch Size (PS)	0.2811	4.273	**0.0003719**	0.4772
Distance to Nearest Green Space (DNGS)	-0.001475	-0.094	0.9258	0.0004441
Tree Canopy (TC)	-0.1546	-0.812	0.4263	0.03193
Grass (G)	-0.002864	-0.019	0.9851	1.794
Soil (S)	-0.03779	-0.195	0.8475	0.001894
Water (W)	0.5531	2.399	**0.02631**	0.2234
Road (R)	0.1350	0.298	0.769	0.004411
Paved (P)	0.05259	0.253	0.8031	0.003182
Building (B)	-0.1228	-0.461	0.6497	0.01052
Pervious (PE)	0.01986	0.047	0.9626	0.0001124
Shrub (SH)	0.2207	1.263	0.2213	0.07382
Impervious (IMP)	0.01712	0.096	0.9242	0.000464

When comparing two multivariate regression models for avian richness predictive power, the model with the urban built form metrics was statistically significant (P=0.04408) and had high explanatory power (R^2 = 0.8106), suggesting that urban built form patterns do influence avian richness outcomes in UGS. The Aggregation Index, patch density, landscape shape index, and architecture height standard deviation were all statistically significant. The Aggregation Index, patch density, and the landscape shape index all had positive relationships with avian richness, whereas the architecture height standard deviation had a negative relationship. These relationships suggest that more compact urban built forms with complex shapes, but less variability in building heights better support avian richness in UGS. The model with site-level UGS metrics performed better in terms of explanatory power (R^2 = 0.8437) and statistical significance (P =0.006091). However, patch size was the only statistically significant variable (P = .000119), suggesting that patch size is the driving factor for avian richness. See Table 6 for all coefficients and P values.

Table 6: Multivariate regression models

Model 1: urban built form metrics			**Model 2: UGS metrics**		
Dependent variable: predicted avian richness			Dependent variable: predicted avian richness		
Independent variables: NB, NTB, TBR, MAPA, MAH, AHSD, BCR, FAR, PD, LSI, SHDI, AI			Independent variables: TC, G, S, W, B, R,SH, PS, DNGS, IMP, PE, SH		
Variable	**Coefficient**	**P value**	**Variable**	**Coefficient**	**P value**
Number of Buildings (NB)	-1.01	0.7394	Tree Canopy (TC)	5.82	0.59774
Number of Tall Buildings (NTB)	-4.55	0.6554	Grass (G)	5.92	0.59224
Tall building ratio (TBR)	9.54	0.4869	Soil (S)	5.8	0.59880
Mean architecture projection area (MAPA)	-8.65	0.1460	Water (W)	6.04	0.58435
Mean architecture height (MAH)	1.58	0.1664	Distance to Nearest Green Space (DNGS)	-8.57	0.27106
Architecture height standard deviation (AHSD)	-4.99	**0.0191****	Road (R)	5.84	0.86
Building coverage ratio (BCR)	-6.85	0.1530	Paved (P)	5.80	0.59799
Floor area ratio (FAR)	3.74	0.1108	Patch Size (PS)	2.96	**0.00051*****
Patch density (PD)	1.31	**0.0122****	Building (B)	5.87	0.59539
Landscape shape index (LSI)	1.31	**0.0582***	Shrub (SH)	5.92	0.59104
Shannon's Diversity Index (SHDI)	1.49	0.1244			
Aggregation Index (AI)	1.1	**0.0110****			
Statistically significant: 0.01**, 0.05* R2:0.8106 **p-value: 0.04408***			Statistically significant: 0.01**, 0.001*** R2: 0.8473 **p-value: 0.006091****		

V. Discussion

The main aim of this study was to quantify urban built form surrounding UGS to characterize the configuration pattern in order to assess the level of influence of these patterns on avian richness outcomes. Previous studies have shown mixed results on certain aspects of urban built form such as building height and density, but in general have tended to find that an increase in urban land cover leads to a decline in avian richness (M. F. J. Aronson et al., 2014; Chowdhury & Sen, 2019) . Meanwhile, the vast majority of studies that have focused on the composition and structure of greenspace have resoundly shown that an increase in patch size is strongly associated with avian richness followed by level of connectivity (Amaya-Espinel et al., 2019; Beaugeard et al., 2020; Beninde et al., 2015; La Sorte et al., 2020). The analysis of this exploratory study confirms previous assertions that patch size is the strongest correlate to avian richness, but presents a more complex story when considering urban built form pattern and level of connectivity. The multivariate regression model with urban built form pattern metrics performed well in predicting avian richness in UGS and revealed some interesting insights, mainly that high aggregation of buildings and increased complexity in shape (i.e. AI and LSI metrics) were positively correlated with avian richness regardless of level of density captured by the building coverage ratio metric. This finding suggests that urban built form patterns have more complex interactions that go beyond the typical 2D density measures used and that urban built form can be maximized for ecological potential at various densities.

This study confirms previous assertions that the size of the UGS is the strongest correlate to avian richness, but did not find a similarly strong relationship with level of connectivity (Amaya-Espinel et al., 2019; Beaugeard et al., 2020; Beninde et al., 2015; Dale, 2018; Ikin et al., 2013; La Sorte et al., 2020; Lynch, 2019; Strohbach et al., 2013). This might be in part due to the proxy measure used for connectivity, nearest distance to green space; however, this measure is a fairly common proxy (Amaya-Espinel et al., 2019). Level of connectivity is a complex metric as there is structural and functional connectivity, the former being the easier of the two to capture (Lynch, 2019). Scholars have developed a methodology for mapping ecological connectivity utilizing principles of resistance and flow from electrical engineering whereby landscapes are treated as resistive surfaces (I. T. Brown, 2019; *TNC*

Omniscape Connectivity, n.d.). Additionally, avian species are unique in that they can fly, meaning that what might be considered a barrier to movement for one species is not the same for an avian species. Scholars have shown that urban flight corridors are an important factor in mapping connectivity, which requires assigning resistive surface scores with 3D building landscapes in mind (Z. Liu et al., 2020). Additionally, it is important to note that the configuration of the built environment has implications for acoustic transmission, which may alter avian flight paths due to an ability or inability to communicate (Warren et al., 2006), which arguably should be considered in resistive surface scoring as this aspect can modify avian behaviour. In the next stage of this study I will use this methodology to map urban flight corridors and will factor in measures of acoustic transmission that may modify these corridors. I hypothesize that this method of measuring connectivity will be strongly associated with avian richness.

When analyzing the influence of urban built form metrics, both the Aggregation Index (AI) and Patch Density (PD) were statistically significant and revealed positive associations with avian richness, suggesting that building configuration patterns influence avian richness outcomes in UGS. The fact that both AI and PD were statistically significant is not surprising in that AI and PD tend to be correlated (M. Liu et al., 2017). What is interesting is that an increase in AI, which quantifies spatial patterns of landscapes, thus representing the aggregation of buildings in each study area in this case, is strongly correlated with an increase in avian richness within UGS (He et al., 2000; M. Liu et al., 2017).

Furthermore, AI was not correlated with the building cover ratio (BCR) metric, a common measure used for density, suggesting that regardless of level of density, higher aggregation patterns of buildings seem to support avian richness in UGS. This confirms studies suggesting that urban areas can maximize ecological potential at any given density (Hostetler & Knowles-Yanez, 2003; Tratalos et al., 2007). Additionally, an increase in LSI was positively correlated with an increase in avian richness with UGS, suggesting that increased complexity in building shape patterns better supports avian biodiversity. Why a higher AI and LSI is correlated with avian richness in UGS remains an open question. In the next stage of research, I will assess whether AI and LSI are correlated with larger UGS or enhanced connectivity via increased vegetation cover or flight corridors, with a consideration of acoustic transmission factored into mapping flight corridors.

The impact of building height on avian richness has had mixed results in research to date. In some cases an increase in building height has been associated with avian richness (Amaya-Espinel et al., 2019), in other studies it has had a negative effect (Loss et al., 2014), and in some it has presented mixed results depending on which avian species is being assessed (MacGregor-Fors et al., 2017). In this analysis, variation in building height, captured by the architecture height standard deviation metric (AHSD), was negatively associated with avian richness in UGS. According to previous studies, AHSD tends to be correlated with a tall building presence, which is true of this study as well (M. Liu et al., 2017). This would suggest that an increase in building height is negatively associated with avian richness in UGS. However, given that this metric represents variation in building height and not outright tall buildings, it is not clear whether tall buildings are fully to blame for this negative effect. One hypothesis is that variation in building height may affect flight paths and acoustic transmission, thus impacting ecological connectivity via flight corridors and acoustic transmission capabilities.

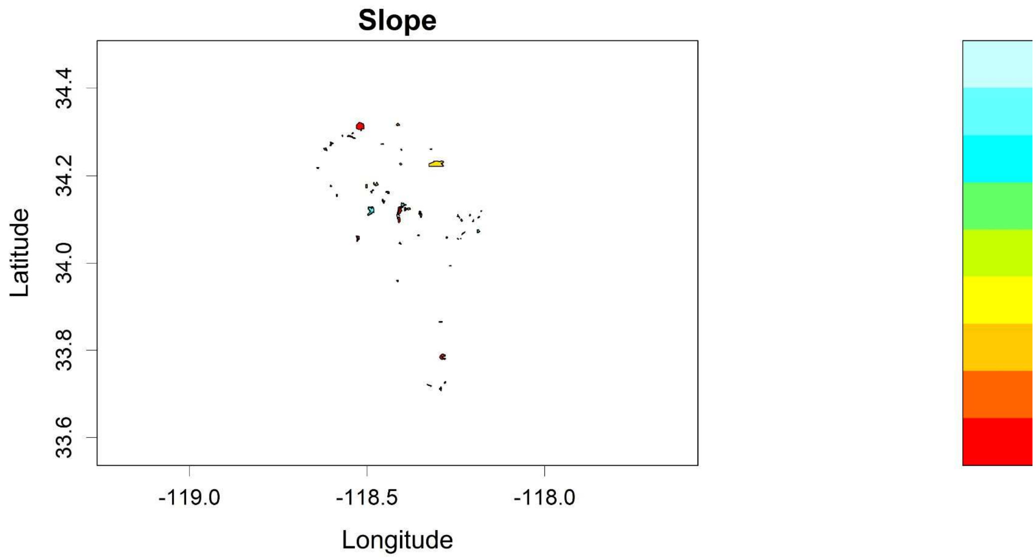
It is important to note that while this exploratory study focused on characterizing patterns of the urban built environment, it will be germane to integrate social attributes of the surrounding area of each UGS in a future study. Cities are complex social-environmental systems, necessitating an investigation of both social and environmental components in order to more fully capture the complex interactions in urban systems (Grimm et al., 2008b; Verburg et al., 2015). Furthermore, socioeconomic and cultural dynamics can be determinants in UGS management decisions, making an analysis of social drivers of avian richness in UGS paramount in future studies (M. F. Aronson et al., 2017). The next stage of this study will investigate demographic variables surrounding each UGS to characterize population and socioeconomic patterns.

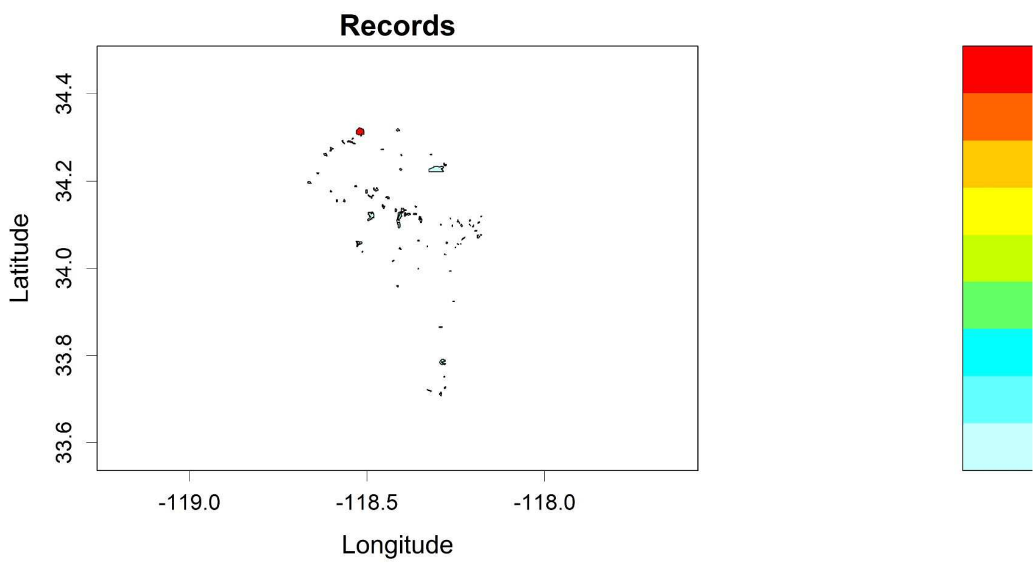
VI. Conclusion

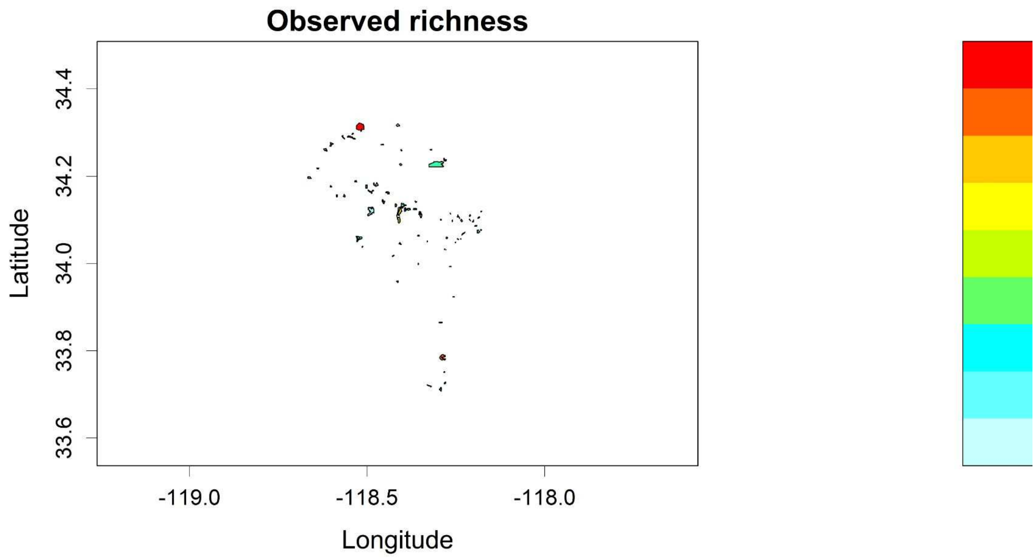
The implications of this exploratory study for urban avian biodiversity conservation suggests that urban design and planning professionals should aim to maximize urban green space size and the compactness and complexity of shape of urban built form as new urban areas develop. More research needs to be conducted to understand the underlying mechanisms that explain why more compact urban forms with complex shape support avian richness in UGS. In addition, an analysis of social drivers of avian richness in UGSs is a critical component to incorporate given that UGS management decisions can be determinants of socioeconomic and cultural dynamics. In a future study, I intend to explore demographic variables surrounding each UGS as well as the relationships between AI and LSI and vegetation cover and landscape connectivity. Furthermore, instead of using distance to nearest green space as a proxy of connectivity, I will map 2D and 3D ecological connectivity utilizing principles of resistance and flow from electrical engineering whereby landscapes are treated as resistive surfaces, factoring in acoustic transmission, to assess urban flight corridors. I hypothesize that higher AI and LSI are associated with enhanced connectivity via increased vegetation cover and urban flight corridors with greater acoustic transmission.

VII. Appendix

KnowBR Analysis Outputs for slope, records, and richness






Observed richness
Latitude
34.4
34.2
34.0
33.8
33.6
-119.0
-118.5
-118.0
Longitude

VIII. References

Alberti, M. (2005). The Effects of Urban Patterns on Ecosystem Function. *International Regional Science Review*, *28*(2), 168–192. https://doi.org/10.1177/0160017605275160

Alvarez, D., Kang, C., Lin, D., Tran, J., & Wu, T. (2015). *Developing Biodiversity Indicators for Los Angeles County*. 60.

Amaya-Espinel, J. D., Hostetler, M., Henríquez, C., & Bonacic, C. (2019). The influence of building density on Neotropical bird communities found in small urban parks. *Landscape and Urban Planning*, *190*, 103578. https://doi.org/10.1016/j.landurbplan.2019.05.009

Andersson, E., & Colding, J. (2014). Understanding how built urban form influences biodiversity. *Urban Forestry & Urban Greening*, *13*(2), 221–226. https://doi.org/10.1016/j.ufug.2013.11.002

Andrade, R., Franklin, J., Larson, K. L., Swan, C. M., Lerman, S. B., Bateman, H. L., Warren, P. S., & York, A. (2020). Predicting the assembly of novel communities in urban ecosystems. *Landscape Ecology*. https://doi.org/10.1007/s10980-020-01142-1

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J. L., Kühn, I., MacGregor-Fors, I., McDonnell, M., Mörtberg, U., … Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B: Biological Sciences*, *281*(1780), 20133330. https://doi.org/10.1098/rspb.2013.3330

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), 189–196. https://doi.org/10.1002/fee.1480

Beaugeard, E., Brischoux, F., & Angelier, F. (2020). Green infrastructures and ecological corridors shape avian biodiversity in a small French city. *Urban Ecosystems*. https://doi.org/10.1007/s11252-020-01062-7

Beninde, J., Veith, M., & Hochkirch, A. (2015). Biodiversity in cities needs space: A meta-analysis of factors determining intra-urban biodiversity variation. *Ecology Letters*, *18*(6), 581–592. https://doi.org/10.1111/ele.12427

Blair, R. B. (1999). Birds and Butterflies Along an Urban Gradient: Surrogate Taxa for Assessing Biodiversity? *Ecological Applications*, *9*(1), 164–170. https://doi.org/10.1890/1051-0761(1999)009[0164:BABAAU]2.0.CO;2

Brown, C., & Grant, M. (2005). Biodiversity and Human Health: What Role for Nature in Healthy Urban Planning? *Built Environment*, *31*(4), 326–338. https://doi.org/10.2148/benv.2005.31.4.326

Brown, I. T. (2019). *Managing Cities as Urban Ecosystems: Analysis Tools for Biodiversity Stewardship in Los Angeles* [UCLA]. https://escholarship.org/uc/item/4c81w4nr#metrics

Brunbjerg, A. K., Hale, J. D., Bates, A. J., Fowler, R. E., Rosenfeld, E. J., & Sadler, J. P. (2018). Can patterns of urban biodiversity be predicted using simple measures of green infrastructure? *Urban Forestry & Urban Greening*, *32*, 143–153. https://doi.org/10.1016/j.ufug.2018.03.015

Carpenter, S. R., Mooney, H. A., Agard, J., Capistrano, D., DeFries, R. S., Díaz, S., Dietz, T., Duraiappah, A. K., Oteng-Yeboah, A., Pereira, H. M., Perrings, C., Reid, W. V., Sarukhan, J., Scholes, R. J., & Whyte, A. (2009). Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. *Proceedings of the National Academy of Sciences*, *106*(5), 1305–1312. https://doi.org/10.1073/pnas.0808772106

Chowdhury, K., & Sen, S. (2019). *A Comparative Study of Avian Biodiversity in Two Sites Viz. Urban and Suburban and in Two Seasons Viz. Winter and Spring* (SSRN Scholarly Paper ID 3633781). Social Science Research Network. https://papers.ssrn.com/abstract=3633781

Colwell, R. K., Coddington, J. A., & Hawksworth, D. L. (1994). Estimating terrestrial biodiversity through extrapolation. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences*, *345*(1311), 101–118. https://doi.org/10.1098/rstb.1994.0091

Connors, J. P., Galletti, C. S., & Chow, W. T. L. (2013). Landscape configuration and urban heat island effects: Assessing the relationship between landscape characteristics and land surface temperature in Phoenix, Arizona. *Landscape Ecology*, *28*(2), 271–283. https://doi.org/10.1007/s10980-012-9833-1

Dale, S. (2018). Urban bird community composition influenced by size of urban green spaces, presence of native forest, and urbanization. *Urban Ecosystems*, *21*(1), 1–14. https://doi.org/10.1007/s11252-017-0706-x

Elmqvist, T., Zipperer, W., & Güneralp, B. (2016). Urbanization, habitat loss, biodiversity decline: Solution pathways to break the cycle. *In, Seta, Karen; Solecki, William D.; Griffith, Corrie A. (Eds.). Routledge Handbook of Urbanization and Global Environmental Change. London and New York:Routledge.*, *2016*, 139–151. https://www.fs.usda.gov/treesearch/pubs/52798

Fuller, R., Irvine, K., & Gaston, K. J. (2010). Interactions between people and nature in urban environments. In *Urban Ecology* (pp. 134–171). Cambridge University Press.

Garrard, G. E., Williams, N. S. G., Mata, L., Thomas, J., & Bekessy, S. A. (2018). Biodiversity Sensitive Urban Design. *Conservation Letters*, *11*(2), e12411. https://doi.org/10.1111/conl.12411

GreenInfoNetwork. (n.d.). *California Protected Areas Database*. Retrieved February 2, 2021, from https://www.calands.org

Grimm, N. B., Chapin, F. S., Bierwagen, B., Gonzalez, P., Groffman, P. M., Luo, Y., Melton, F., Nadelhoffer, K., Pairis, A., Raymond, P. A., Schimel, J., & Williamson, C. E. (2013). The impacts of climate change on ecosystem structure and function. *Frontiers in Ecology and the Environment*, *11*(9), 474–482. https://doi.org/10.1890/120282

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M. (2008a). Global Change and the Ecology of Cities. *Science*, *319*(5864), 756–760. https://doi.org/10.1126/science.1150195

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M. (2008b). Global Change and the Ecology of Cities. *Science*, *319*(5864), 756–760. https://doi.org/10.1126/science.1150195

Groffman, P. M., Cavender-Bares, J., Bettez, N. D., Grove, J. M., Hall, S. J., Heffernan, J. B., Hobbie, S. E., Larson, K. L., Morse, J. L., Neill, C., Nelson, K., O'Neil-Dunne, J., Ogden, L., Pataki, D. E., Polsky, C., Chowdhury, R. R., & Steele, M. K. (2014). Ecological homogenization of urban USA. *Frontiers in Ecology and the Environment*, *12*(1), 74–81. https://doi.org/10.1890/120374

Haines-Young, R. (2009). Land use and biodiversity relationships. *Land Use Policy*, *26*, S178–S186. https://doi.org/10.1016/j.landusepol.2009.08.009

Hazell, E. C., & Rinner, C. (2020). The impact of spatial scale: Exploring urban butterfly abundance and richness patterns using multi-criteria decision analysis and principal component analysis. *International Journal of Geographical Information Science*, *34*(8), 1648–1681. https://doi.org/10.1080/13658816.2019.1675072

He, H. S., DeZonia, B. E., & Mladenoff, D. J. (2000). An aggregation index (AI) to quantify spatial patterns of landscapes. *Landscape Ecology*, *15*(7), 591–601. https://doi.org/10.1023/A:1008102521322

Hostetler, M., Allen, W., & Meurk, C. (2011). Conserving urban biodiversity? Creating green infrastructure is only the first step. *Landscape and Urban Planning*, *100*(4), 369–371. https://doi.org/10.1016/j.landurbplan.2011.01.011

Hostetler, M., & Holling, C. S. (2000). Detecting the scales at which birds respond to structure in urban landscapes. *Urban Ecosystems*, *4*(1), 25–54. https://doi.org/10.1023/A:1009587719462

Hostetler, M., & Knowles-Yanez, K. (2003). Land use, scale, and bird distributions in the Phoenix metropolitan area. *Landscape and Urban Planning*, *62*(2), 55–68. https://doi.org/10.1016/S0169-2046(02)00096-8

Ikin, K., Beaty, R. M., Lindenmayer, D. B., Knight, E., Fischer, J., & Manning, A. D. (2013). Pocket parks in a compact city: How do birds respond to increasing residential density? *Landscape Ecology*, *28*(1), 45–56. https://doi.org/10.1007/s10980-012-9811-7

Ives, C. D., Lentini, P. E., Threlfall, C. G., Ikin, K., Shanahan, D. F., Garrard, G. E., Bekessy, S. A., Fuller, R. A., Mumaw, L., Rayner, L., Rowe, R., Valentine, L. E., & Kendal, D. (2016). Cities are hotspots for threatened species. *Global Ecology and Biogeography*, *25*(1), 117–126. https://doi.org/10.1111/geb.12404

Kati, V., Devillers, P., Dufrêne, M., Legakis, A., Vokou, D., & Lebrun, P. (2004). Testing the Value of Six Taxonomic Groups as Biodiversity Indicators at a Local Scale. *Conservation Biology*, *18*(3), 667–675. https://doi.org/10.1111/j.1523-1739.2004.00465.x

Kaushik, M., Tiwari, S., & Manisha, K. (2020). Habitat patch size and tree species richness shape the bird community in urban green spaces of rapidly urbanizing region of India. *BioRxiv*, 2020.10.23.348391. https://doi.org/10.1101/2020.10.23.348391

KnowBR: Discriminating Well Surveyed Spatial Units from Exhaustive Biodiversity Databases. (n.d.). Comprehensive R Archive Network (CRAN). Retrieved May 21, 2021, from https://CRAN.R-project.org/package=KnowBR

La Sorte, F. A., Aronson, M. F. J., Lepczyk, C. A., & Horton, K. G. (2020). Area is the primary correlate of annual and seasonal patterns of avian species richness in urban green spaces. *Landscape and Urban Planning*, *203*, 103892. https://doi.org/10.1016/j.landurbplan.2020.103892

Lepczyk, C. A., La Sorte, F. A., Aronson, M. F. J., Goddard, M. A., MacGregor-Fors, I., Nilon, C. H., & Warren, P. S. (2017). Global Patterns and Drivers of Urban Bird Diversity. In E. Murgui & M. Hedblom (Eds.), *Ecology and Conservation of Birds in Urban Environments* (pp. 13–33). Springer International Publishing. https://doi.org/10.1007/978-3-319-43314-1_2

Lindenmayer, D. B., Blanchard, W., Foster, C. N., Scheele, B. C., Westgate, M. J., Stein, J., Crane, M., & Florance, D. (2020). Habitat amount versus connectivity: An empirical study of bird responses. *Biological Conservation*, *241*, 108377. https://doi.org/10.1016/j.biocon.2019.108377

Liu, M., Hu, Y.-M., & Li, C.-L. (2017). Landscape metrics for three-dimensional urban building pattern recognition. *Applied Geography*, *87*, 66–72. https://doi.org/10.1016/j.apgeog.2017.07.011

Liu, Z., Huang, Q., & Tang, G. (2020). Identification of urban flight corridors for migratory birds in the coastal regions of Shenzhen city based on three-dimensional landscapes. *Landscape Ecology*. https://doi.org/10.1007/s10980-020-01032-6

Lobo, J. M., Hortal, J., Yela, J. L., Millán, A., Sánchez-Fernández, D., García-Roselló, E., González-Dacosta, J., Heine, J., González-Vilas, L., & Guisande, C. (2018). KnowBR: An application to map the geographical variation of survey effort and identify well-surveyed areas from biodiversity databases. *Ecological Indicators*, *91*, 241–248. https://doi.org/10.1016/j.ecolind.2018.03.077

Loss, S. R., Will, T., Loss, S. S., & Marra, P. P. (2014). Bird–building collisions in the United States: Estimates of annual mortality and species vulnerability. *The Condor*, *116*(1), 8–23. https://doi.org/10.1650/CONDOR-13-090.1

Lynch, A. J. (2019). Creating Effective Urban Greenways and Stepping-stones: Four Critical Gaps in Habitat Connectivity Planning Research. *Journal of Planning Literature*, *34*(2), 131–155. https://doi.org/10.1177/0885412218798334

MacGregor-Fors, I., Quesada, J., Lee, J., & Yeh, P. (2017). Space invaders: House Sparrow densities along three urban-agricultural landscapes. *Avian Conservation and Ecology*, *12*(2). https://doi.org/10.5751/ACE-01082-120211

Marzluff, J. M. (2017). A decadal review of urban ornithology and a prospectus for the future. *Ibis*, *159*(1), 1–13. https://doi.org/10.1111/ibi.12430

McDonald, R. I., Mansur, A. V., Ascensão, F., Colbert, M., Crossman, K., Elmqvist, T., Gonzalez, A., Güneralp, B., Haase, D., Hamann, M., Hillel, O., Huang, K., Kahnt, B., Maddox, D., Pacheco, A., Pereira, H. M., Seto, K. C., Simkin, R., Walsh, B., … Ziter, C. (2020). Research gaps in knowledge of the impact of urban growth on biodiversity. *Nature Sustainability*, *3*(1), 16–24. https://doi.org/10.1038/s41893-019-0436-6

McGarigal, K., & Marks, B. J. (1995). FRAGSTATS: Spatial pattern analysis program for quantifying landscape structure. *Gen. Tech. Rep. PNW-GTR-351. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 122 p*, *351*. https://doi.org/10.2737/PNW-GTR-351

McPhearson, T., Pickett, S. T. A., Grimm, N. B., Niemelä, J., Alberti, M., Elmqvist, T., Weber, C., Haase, D., Breuste, J., & Qureshi, S. (2016). Advancing Urban Ecology toward a Science of Cities. *BioScience*, *66*(3), 198–212. https://doi.org/10.1093/biosci/biw002

Mikusiński, G., Gromadzki, M., & Chylarecki, P. (2001). Woodpeckers as Indicators of Forest Bird Diversity. *Conservation Biology*, *15*(1), 208–217. https://doi.org/10.1111/j.1523-1739.2001.99236.x

ModestR software for species distribution data management. (n.d.). Retrieved May 21, 2021, from http://www.ipez.es/modestr/

Nilon, C. H., Aronson, M. F. J., Cilliers, S. S., Dobbs, C., Frazee, L. J., Goddard, M. A., O'Neill, K. M., Roberts, D., Stander, E. K., Werner, P., Winter, M., & Yocom, K. P. (2017). Planning for the Future of Urban Biodiversity: A Global Review of City-Scale Initiatives. *BioScience*, *67*(4), 332–342. https://doi.org/10.1093/biosci/bix012

Norton, B. A. (2016). *Urban Biodiversity and Landscape Ecology: Patterns, Processes and Planning*.

Pebesma, E. (2021, March 17). *Simple Features for R [R package sf version 0.9-8]*.

Comprehensive R Archive Network (CRAN). https://CRAN.R-project.org/package=sf

Plummer, K. E., Gillings, S., & Siriwardena, G. M. (2020). Evaluating the potential for bird-habitat models to support biodiversity-friendly urban planning. *Journal of Applied Ecology*, *57*(10), 1902–1914. https://doi.org/10.1111/1365-2664.13703

Roy Chowdhury, R., Larson, K., Grove, M., Polsky, C., Cook, E., Onsted, J., & Ogden, L. (2011). A Multi-Scalar Approach to Theorizing Socio-Ecological Dynamics of Urban Residential Landscapes. *Cities and the Environment (CATE)*, *4*(1). https://digitalcommons.lmu.edu/cate/vol4/iss1/6

Sandifer, P. A., Sutton-Grier, A. E., & Ward, B. P. (2015). Exploring connections among nature, biodiversity, ecosystem services, and human health and well-being: Opportunities to enhance health and biodiversity conservation. *Ecosystem Services*, *12*, 1–15. https://doi.org/10.1016/j.ecoser.2014.12.007

Schütz, C., & Schulze, C. H. (2015). Functional diversity of urban bird communities: Effects of landscape composition, green space area and vegetation cover. *Ecology and Evolution*, *5*(22), 5230–5239. https://doi.org/10.1002/ece3.1778

Schwarz, N., Moretti, M., Bugalho, M. N., Davies, Z. G., Haase, D., Hack, J., Hof, A., Melero, Y., Pett, T. J., & Knapp, S. (2017). Understanding biodiversity-ecosystem service relationships in urban areas: A comprehensive literature review. *Ecosystem Services*, *27*, 161–171. https://doi.org/10.1016/j.ecoser.2017.08.014

Strimas-Mackey, M., Hochachka, W. M., Ruiz-Gutierrez, V., Robinson, O. J., Miller, E. T., Auer, T., Kelling, S., Fink, D., & Johnston, A. (2020). *Best Practices for Using eBird Data v1.0* (v1.0) [Computer software]. Zenodo. https://doi.org/10.5281/zenodo.3620739

Strimas-Mackey, M., Miller, E., Hochachka, W., & Ornithology, C. L. of. (2020). *auk: EBird Data Extraction and Processing in R* (0.4.3) [Computer software]. https://CRAN.R-project.org/package=auk

Strohbach, M. W., Lerman, S. B., & Warren, P. S. (2013). Are small greening areas enhancing bird diversity? Insights from community-driven greening projects in Boston. *Landscape and Urban Planning*, *114*, 69–79. https://doi.org/10.1016/j.landurbplan.2013.02.007

Sullivan, B. L., Aycrigg, J. L., Barry, J. H., Bonney, R. E., Bruns, N., Cooper, C. B., Damoulas, T., Dhondt, A. A., Dietterich, T., Farnsworth, A., Fink, D., Fitzpatrick, J. W., Fredericks, T., Gerbracht, J., Gomes, C., Hochachka, W. M., Iliff, M. J., Lagoze, C., La Sorte, F. A., … Kelling, S. (2014). The eBird enterprise: An integrated approach to development and application of citizen science. *Biological Conservation*, *169*, 31–40. https://doi.org/10.1016/j.biocon.2013.11.003

Sushinsky, J. R., Rhodes, J. R., Possingham, H. P., Gill, T. K., & Fuller, R. A. (2013). How should we grow cities to minimize their biodiversity impacts? *Global Change Biology*, *19*(2), 401–410. https://doi.org/10.1111/gcb.12055

Threlfall, C. G., Williams, N. S. G., Hahs, A. K., & Livesley, S. J. (2016). Approaches to urban vegetation management and the impacts on urban bird and bat assemblages. *Landscape and Urban Planning*, *153*, 28–39. https://doi.org/10.1016/j.landurbplan.2016.04.011

TNC Omniscape Connectivity. (n.d.). Retrieved May 23, 2021, from https://tnc.maps.arcgis.com/apps/webappviewer/index.html?id=3cbb9454372e43ffac44b9dda07b5551

Tratalos, J., Fuller, R. A., Warren, P. H., Davies, R. G., & Gaston, K. J. (2007). Urban form, biodiversity potential and ecosystem services. *Landscape and Urban Planning*, *83*(4), 308–317. https://doi.org/10.1016/j.landurbplan.2007.05.003

Trollope, S. T., White, J. G., & Cooke, R. (2009). The response of ground and bark foraging insectivorous birds across an urban–forest gradient. *Landscape and Urban Planning*, *93*(2), 142–150. https://doi.org/10.1016/j.landurbplan.2009.06.013

Ugland, K. I., Gray, J. S., & Ellingsen, K. E. (2003). The species–accumulation curve and estimation of species richness. *Journal of Animal Ecology*, *72*(5), 888–897.

https://doi.org/10.1046/j.1365-2656.2003.00748.x

Verburg, P. H., Crossman, N., Ellis, E. C., Heinimann, A., Hostert, P., Mertz, O., Nagendra, H., Sikor, T., Erb, K.-H., Golubiewski, N., Grau, R., Grove, M., Konaté, S., Meyfroidt, P., Parker, D. C., Chowdhury, R. R., Shibata, H., Thomson, A., & Zhen, L. (2015). Land system science and sustainable development of the earth system: A global land project perspective. *Anthropocene*, *12*, 29–41. https://doi.org/10.1016/j.ancene.2015.09.004

von der Lippe, M., Buchholz, S., Hiller, A., Seitz, B., & Kowarik, I. (2020). CityScapeLab Berlin: A Research Platform for Untangling Urbanization Effects on Biodiversity. *Sustainability*, *12*(6), 2565. https://doi.org/10.3390/su12062565

Warren, P. S., Katti, M., Ermann, M., & Brazel, A. (2006). Urban bioacoustics: It's not just noise. *Animal Behaviour*, *71*(3), 491–502. https://doi.org/10.1016/j.anbehav.2005.07.014

Wentz, E. A., York, A. M., Alberti, M., Conrow, L., Fischer, H., Inostroza, L., Jantz, C., Pickett, S. T. A., Seto, K. C., & Taubenböck, H. (2018). Six fundamental aspects for conceptualizing multidimensional urban form: A spatial mapping perspective. *Landscape and Urban Planning*, *179*, 55–62. https://doi.org/10.1016/j.landurbplan.2018.07.007

White, J. G., Antos, M. J., Fitzsimons, J. A., & Palmer, G. C. (2005). Non-uniform bird assemblages in urban environments: The influence of streetscape vegetation. *Landscape and Urban Planning*, *71*(2), 123–135. https://doi.org/10.1016/j.landurbplan.2004.02.006

Zaldivar, E. C. (2020). *2020 Biodiversity Report City of Los Angeles* (p. 110). LA Sanitation. https://www.lacitysan.org/san/sandocview?docname=cnt052553

Zhou, W., Huang, G., & Cadenasso, M. L. (2011). Does spatial configuration matter? Understanding the effects of land cover pattern on land surface temperature in urban landscapes. *Landscape and Urban Planning*, *102*(1), 54–63. https://doi.org/10.1016/j.landurbplan.2011.03.009