ECOLOGICAL DESIGN AND SMART LANDSCAPES: BOOSTING THE CONNECTION BETWEEN SCIENTIFIC FINDINGS AND DESIGN APPROACHES WITH SMART TECHNOLOGIES

BY

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DISSERTATION

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ABSTRACT

Landscape architecture is considered to be a highly comprehensive discipline that has been absorbing knowledge from many different disciplines since its establishment. Exponential growth in digitized information, a narrowing field of study, and a lack of interdisciplinary communication, however, are making it difficult for contemporary designers to keep up with the newest scientific concepts and technologies. Concepts in the life sciences such as biological control and ecosystem services, for example, have the potential to improve the performance of built environments, yet have not gained the full attention of landscape designers. Similarly, while the smart city has been a popular concept and environmental sensors have been installed in many parks and public spaces, we have yet to see how the data can help at the level of landscape design. Limitations inherent in scientific reductionism and an ability to analyze complex systems are also inhibiting designer interactions with complex bio-ecological information. To overcome these obstacles and connect ecological theories with landscape design, this dissertation aims to develop a "smart" system that integrates various data sources and scientific models. The purpose of the smart system is to ease the transition between scientific findings, design principles, and actions – in the form of landscape designs or plans.

The dissertation will present the following research projects that have applied smart tools to support landscape design, and how the conceptual framework of smart landscapes has evolved over the course of these studies.

- Assessing the Impacts of City Developments on Urban Green Spaces and Natural Areas Using the Landuse Evolution and Impact Assessment Model (LEAM) (Chapters 3 and 4). LEAM is a dynamic model and planning support system that forecasts future land use

changes based on various socioeconomic, sociophysical, and geographic variables. Projecting potential patterns of urban change in the near future, I was able to evaluate the stresses posed by these changes on green spaces and the impacts on the provisioning of ecosystem services. A comparative study was also conducted to investigate how the same data processing approach can be applied in different cities.

- Delivery of Ecosystem Services from Urban Parks and Green Spaces (Chapter 5): A Supply-and-Demand Analysis in Stockholm, SE. While urban parks can provide ecosystem services to the residents, the services can only be received when they are accessible to the residents. In this study, I combined a transportation model with multiple ecological indicators to evaluate the delivery of ecosystem services in the city of Stockholm, Sweden. The model is based on both the quality of ecosystems and their accessibility to the broader urban population. The work demonstrates the power of a combined ecological and transportation model in extracting information from fragmented data, and presents the results in easy-to-understand visual representations.
- Estimating Carbon Sequestration in Illinois: A Forest Structure Approach (Chapter 6). Forests make critically important contributions to the carbon pool. In Illinois, forests make up only 14.9% of all land cover in 2016, down from approximately 42.3% in the early 20th century. Still, understanding the sequestration potential of all lands and in particular, forested lands is vital to developing a carbon-neutral plan for the state. In this study, I analyzed forested structures in Illinois using data from the Critical Trends Assessment Program (CTAP). Carbon sequestration potential from the forests was estimated and projected based on the forest stand structure. One observation result of this study is that the state has been losing younger trees over the last two decades, meaning a

long-term reduction of sequestration is potential. Action is needed to ensure the survivability of the younger trees and preserve their sequestration potential. This study has highlighted the predictive power of the smart framework.

- Disease Vector Control with Landscape Integrated Pest Management: the Predatory Landscape (Chapter 7). This is a conceptual project that proposes a landscape design philosophy that issues landscape-based structure to reduce the threat of pest-borne disease vectors. A predatory landscape is proposed that creates habitats that accommodate the proliferation of natural predators of mosquitos. This project includes a literature review, a model simulation, and a case study to explore which of the landscape design features may most effectively reduce mosquito-borne threats. A data collection, data processing, and feedback framework is proposed based on the analysis.

With lessons and experiences learned from the above research, a more complex and sophisticated smart framework is introduced in Chapter 8. This framework is characterized by six features: **sentience, incisiveness, comprehensibleness, reasonableness, predictiveness, and adaptiveness**. It includes elements of data collection, information extraction, visualization and presentation, design/management decisions, and data feedback.

To my family

for giving me courage to pursue my dreams

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CHAPTER 1: INTRODUCTION

Ecosystems are essential parts of outdoor environments. Nevertheless, many newer concepts of ecology such as ecosystem services are not very frequently seen in the field of landscape architecture. While contemporary landscape architects are fully aware of the importance of ecological processes in their designed outdoor spaces, the differences in the philosophies and methodologies between the two disciplines have prevented them from adapting the newest ecological principles. The common reductionism and narrowness of the study field in many scientific branches, including ecology, have resulted in highly fragmented pieces of knowledge (Creswell & Creswell, 2017). In addition, ecological studies are typically set in well-preserved ecosystems that are far from densely populated areas so they offer limited help to the typical anthropocentric landscape design project (M. van Lierop, 2011). As a result, landscape architects can find it difficult to access ecological knowledge, including principles and established methods that would enhance their professional day-to-day design processes. To promote a more effective integration of ecological approaches and design knowledge the science of ecological thinking needs a design-centric philosophy and a language that is easily understandable to landscape designers. Fortunately, new developments in data sciences and information and communication technologies (ICTs) can help to promote this transition from scientific knowledge to design practices.

In the last decade, we have witnessed the rapid emergence of ICTs and their application in many different fields including design and planning. Under the concept of smart cities, ICT-based smart tools have exhibited great potential for helping to optimize urban services planning and maintenance (Bifulco et al., 2016; Yeh, 2017). In the field of landscape architecture, however, the application of smart approaches is still a nascent idea. While there are examples such as sensors and other data collection devices that have been installed in urban open spaces (Barth, 2017; Ricaurte, 2021), how such data might help in landscape design is still unclear. One consideration is that design and planning can be highly divergent processes that emphasize different values and outcomes (Lierop et al., 2011). Moreover, the lack of awareness of planning support tools within the spatial planning community (G. Vonk et al., 2005) has also prevented these tools from being fully utilized in landscape design activities.

ICT-based tools can bring both opportunities and challenges to landscape design. These new technologies allow enormous amounts of data to be collected, stored, processed, and communicated with high efficiency (Sinaeepourfard et al., 2020; Tearle, 2003). As a result, it should be easier for landscape architects to access numerous and discrete scientific findings. On the other hand, without proper filtering and manipulation, they can also be easily overwhelmed by the torrents of information. This was a common problem in early attempts of smart city implementations, where the "smartness" of a city was measured by the amount of ICT infrastructure with little context awareness or connections to natural or social systems (Caragliu et al., 2013). Although vast amounts of data were generated, some of these systems simply failed to provide useful information to the actual decision-making process (Hollands, 2008). It ended up taking several years to establish an anthropocentric approach to smart cities that is centered on human needs and interactions (Caragliu et al., 2013). A sophisticated 'smart' system is more than mere data collection and processing (Begg, 2002). Instead, it needs to be "sentient", being aware of the context, of the application, and of the user (Deal et al., 2017).

Basic components of ecosystems, including both the organic elements (such as vegetation) and the inorganic elements (such as soil and water), makes essential design elements in landscape architecture. As a result, it is necessary for landscape architects to understand both these elements and their relationships within an ecosystem so they will be capable to design an outdoor element that is beneficial to both nature and human beings (Lovell & Johnston, 2009). Unfortunately, while most landscape architects are willing to participate in ecologically-oriented design practices, they usually end up finding their knowledge and skills in ecology are not enough to conduct such designs (Szenasy, 2002). To connect ecological principles with design implications, certain protocols and tools need to be developed to translate fragmented and abstract ecological knowledge into design suggestions that are easy for the designers to understand and adopt.

In this dissertation, I propose a concept of “smart landscapes”. Inspired by smart cities (Caragliu et al., 2013), planning support systems (Geertman & Stillwell, 2004; Pettit, Bakelmun, Lieske, Glackin, Thomson, et al., 2018), and responsive landscapes (Cantrell & Holzman, 2015), the smart landscape utilizes big data, quantitative models, and planning/design/decision support systems to form a smart system that accommodates the needs of landscape architects. The goal of the smart system is to translate the voluminous and fragmented data into useful information that can support design and management decisions. It is expected that the smart system will help to bridge the fragmented, sometimes abstract ecologically-based data into easy-to-understand information that landscape architects can use to bring forward reasonable design and management strategies at each of the study scales (**Figure. 1**).

This dissertation focuses on the specific aspect of ecological design and its relationship to smart landscapes. The concept and framework of smart landscapes have been tested by five research projects in different scales and contexts. These projects are organized in a logical order. Research projects in the earlier chapters are connected more closely to well documented concepts (such as PSSs), while more original concepts and methodologies can be found in later chapters. Chapter 3 shows examples of how PSSs can be applied to support the ecological design of urban green spaces. Chapter 4 brings the approach one step further to a comparative study, highlighting the importance of context-specific, or “sentient” data processing. Chapter 5 combines ecological indicators with transportation models to evaluate the delivery of ecosystem services, providing an example of how hidden information can be revealed and presented by data processing. Chapter 6 applies a predictive model to forecast dynamics of forest stand structure, as well as corresponding changes of carbon sequestration potential in near future. Chapter 7 is a conceptual project that utilizes a full data processing framework to support the biological control of disease vectors.

In its initial state, the smart framework was as simple as using data processing to support design decision-making. After completing the research projects, the framework has been developed into a much more complex form that includes data collection, information extraction, visualization and presentation, design/management decisions, and data feedback (**Figure. 2**). A discussion of each Chapters contribution to the framework is made at the end of each chapter. A summary and discussion of the framework is elaborated in Chapter 8.

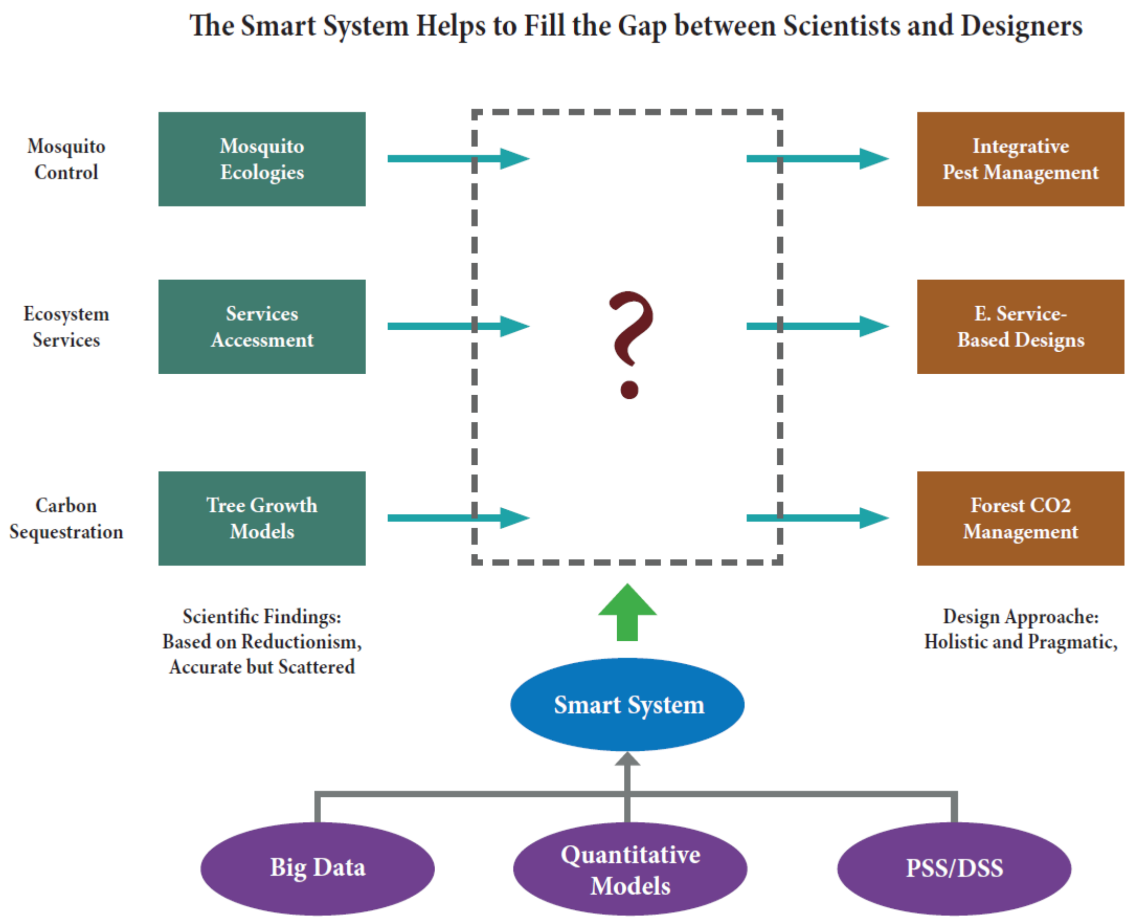


Figure 1 An illustration of the structure of this dissertation. The dissertation aims to establish the landscape-oriented smart system and apply it to three different ecological design projects.

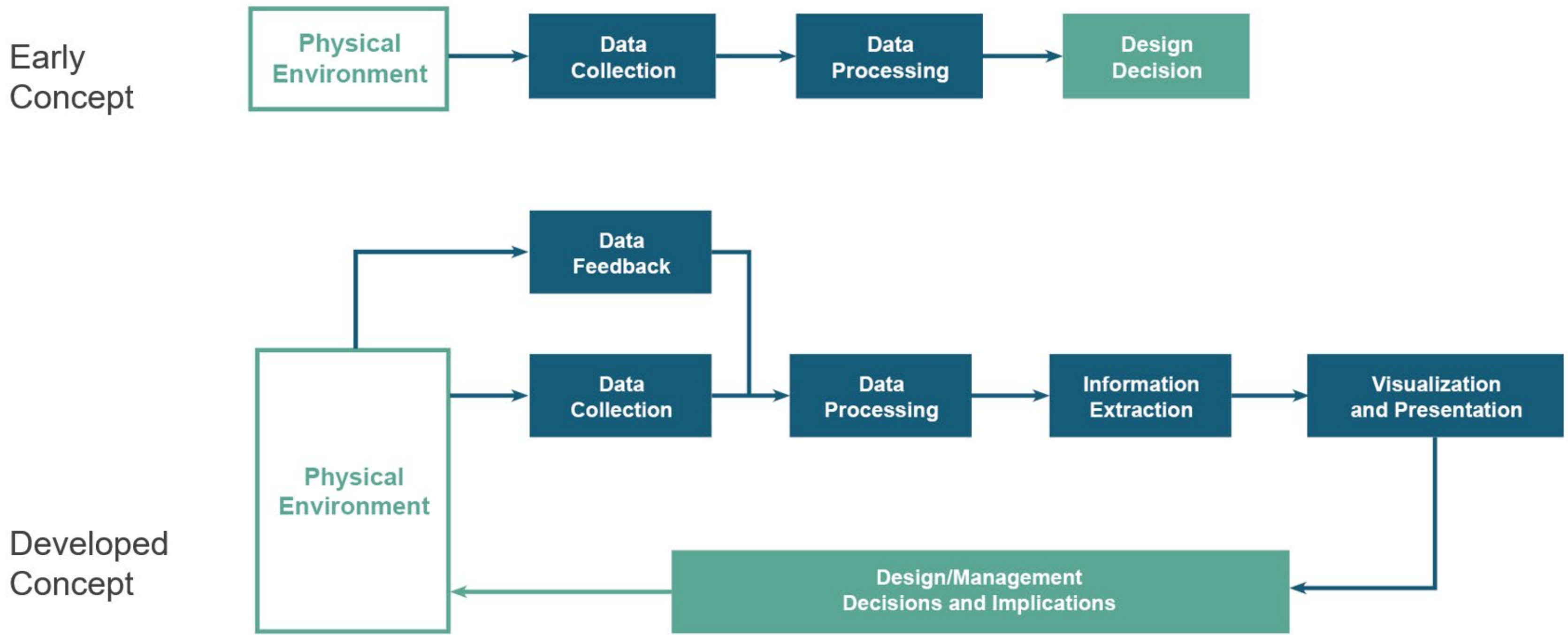


Figure 2 The evolution of the smart landscape framework, before and after completing the 5 research projects.

CHAPTER 2: LITERATURE REVIEW

In this chapter, I cover existing concepts, knowledge, and projects related to ecological design and smart landscapes. Section 2.1 will introduce the important inspirations that have helped me to develop the concepts. Section 2.2 - 2.5 will summarize the background knowledge and existing studies that are related to the individual research projects in Chapter 3 through Chapter 7. The literature review will delineate why the topic is important, what are the current problems we are facing, as well as how they might be studied or addressed.

2.1 PLANNING AND DECISION SUPPORT SYSTEMS AND RELATED CONCEPTS

A Planning or Decision Support System (PSS/DSS) is a combination of information systems, developed to support the decision-making process in planning, business, management, or other practices (Geertman & Stillwell, 2004; Pettit et al., 2018; Vonk, 2006). Usually, this type of system gathers data from different sources. The data is then interpreted through a series of modes and turned into advice to the decision makers (Brail & Klosterman, 2001). The results are visualized and presented through an interface (Power, 2002; G. Vonk & Geertman, 2008). While less frequently seen among designers, similar approaches do exist as Design Support Tools or Systems (Han et al., 2018; Kobayashi et al., 2005; Rose et al., 2016). An early example by (Jindo et al., 1995) evaluated the relationship between users' reflection and design elements of office chairs using a semantic differential method. The system was able to select preferred elements and represent the chair design in 3D models. Another example by (Tokumaru et al., 2002) supported product designers by generating color schemes automatically. The interface allowed the user to input one color and several keywords describing the impression of the color scheme. The system would then select colors that fit the entered requirements from databases and combine them under the rule of color harmony.

One primary inspiration of this dissertation is the Landuse Evolution and Impact Assessment Model (LEAM) PSS (Deal & Pallathucheril, 2009; Deal et al., 2015). The LEAM is a planning support system built around land use models, which is capable to predict future urban development based on a wide range of biophysical and socio-economic factors (Deal & Pallathucheril, 2009; Pan et al., 2020; Pan, Zhang, et al., 2019) Basically, the model works by delineating these factors (such as homes, employments, public services, or urban green spaces) as a set of "attractors". The influence of each attractor is represented by an attraction weight. For example, the attraction of a residential area is determined by its population, while the attraction of a park is determined by its quality of service. The accessibility to the attractors is then calculated using a travel-cost approach based on the transportation network (Pan, Zhang, et al., 2019). Multiple travel modes including walking, biking, driving, and public transit can be assessed (Cong et al., 2022). The attraction weight and travel cost are then combined to create an "attraction map", in which a higher attraction value means better accessibility to resources such as housing, employment, or recreational opportunities. The probability of future development is determined by these attraction values. A modular approach enables the stringing of various attractors and cellular automata based decision models (Deal & Pallathucheril, 2009). A final probability surface is used to allocate various cell types.

One advantage of LEAM is that the spatial relationship is represented by attraction value, an indicator that reflects both the accessibility to an attractor and the importance of the attractor itself at the same time (Zhang & Deal, 2020). In addition, the accessibility is calculated by network-based travel time instead of Euclidean distance. This enables the complexity of the urban fabric to be more accurately reflected in the probability surfaces generated without oversimplification (Tanser et al., 2006). Moreover, the model is very flexible in terms of input variables, any biophysical or socio-economic factors that may affect future development can be considered as an attractor. As a result, it can be utilized under many different conditions without extensive modification (Pan et al., 2018). Last but not least, it provides an online API that allows users without professional knowledge to use the system and read the results with ease (Zhang & Deal, 2020). Advanced users can also modify and customize the model to meet their needs. This type of dynamic planning and design support will be important for providing accessible information that designers might find useful in daily landscape design activities.

2.1.1 Related Concepts in Landscape Design

A frequently used information-driven design process in landscape architecture is GeoDesign. Being an evolving concept, the definition of GeoDesign varies among designers, but there are generally some common features such as the full awareness of geographical context; the evolvement of specialist from different professions and interdisciplinary communications; the emphasis on models, algorithms, and computational technologies; the science-, data-, and evidence-driven decision-making process; as well as timely feedback about the outcomes of the design implication (Batty, 2013; Ervin, 2013; Miller, 2012; Steinitz, 2012). From the aspect of planners, GeoDesign allows designers to benefit from state-of-the-art planning support system information. The key is that designers are able to understand whether their design decisions are viable and desirable from a planning perspective (Warren-Kretzschmar et al., 2012). The system thinking and interdisciplinary collaboration of GeoDesign can help the landscape architects to navigate through the complexity of ecological principles, allowing more concepts in ecology, environmental sciences, and sustainability to be adapted in landscape design (Gu et al., 2018).

To make it easier for designers, managers, and advocates to incorporate the newest technologies into the parks, UCLA Luskin Center for Innovation (2017) put forward a concept of

"SMART Parks", defined as "*a park that uses technology (environmental, digital, and materials) to achieve a series of values: equitable access, community fit, enhanced health, safety, resilience, water and energy efficiency, and effective operations and maintenance*". While not focusing on models and data-driven processes, this concept agreed with one of the important ideas of this dissertation, that the "smartness" of a landscape should be measured on the system's ability to effect on-the-ground design and landscape scaled decision making, instead of the mere amount of technologies implied. Cantrell & Holzman (2015) proposed a concept of "responsive landscape". The concept argues that landscapes are dynamic and temporal mediums by their nature, and designers should respect this quality. To be responsive, a landscape should constantly engage in the process of feedback and conversation. They suggest that the design of processes, logic, and protocols in addition to physical objects, and the design interventions should evolve throughout a project's lifespan. While not as sophisticated as smart cities, these concepts have provided inspiring insights into innovative landscape design processes.

2.2 BIOLOGICAL CONTROL OF VECTOR-BORNE DISEASES

In their early ages, city planning and public health shared many common concerns and evolved together. Rapid industrialization and poor living condition in the late-19th century had caused devastating outbreaks of infectious diseases such as cholera (Smith, 2002) and typhoid (Vanderslott et al., 2019). During the time, planners and health specialists worked mutually to develop better drainage and ventilation systems, treat hazardous wastes, and improve the poor living condition (Greenberg et al., 1994). While the cooperation had achieved great success, it did not last long into the 20th century. Since vaccination and antibiotics had been proven effective in eliminating pathogens, urban infrastructure has become less prominent in the control of infectious diseases. Ironically, the advanced medical technologies not only failed to encourage further cooperation between urban planning and public health, but eventually caused a disconnection between them (Corburn, 2004).

Modern urban planners are not unaware of this disconnection, and there has been an increasing concern of the fact that the potential of urban planning in promoting the health of the residents has been underestimated (Corburn, 2004; Hancock, 1996). While the living condition in modern cities is much better comparing with that in the 19^{th} century, health problems still raise as results of pollution, stress, lack of accessibility to natural areas (Sullivan & Chang, 2017), and overdependence on automobiles (Jackson & Kochtitzky, 2001). Evidences have shown that sufficient exposure to parks or even street trees will have positive effects on both physical and mental health (Frumkin et al., 2017; D. Li & Sullivan, 2016). A well-designed, walking- and biking-friendly outdoor environment will encourage the residents to exercise more and thus reduce the risk of chronic health problems such as obesity, heart disease, diabetes, or asthma (Jackson & Kochtitzky, 2001). A recent study also discovered that a higher ratio of green spaces was significantly associated with a lower racial disparity in the SARS-CoV-2 infection rate, suggesting the possible relationship between the availability of green spaces and health equity (Y. Lu et al., 2020). All these findings indicate that urban planning and design plays a conspicuous role in public health.

2.2.1 The Increasing Threat from Vector-Borne Diseases in Near Future

The term vector-borne diseases refers to human illnesses that can be transmitted between humans, or from animals to humans, by living vectors, usually bloodsucking insects (World Health Organization, 2020). While considered to be less threatening in countries with better health services, vector-borne diseases remains a major reason for deaths in the developing world (Mbacham et al., 2019). Moreover, evidences have shown growing threats from vector-borne diseases since the late 20th century, with the range of the diseases spreading geographically and the number of cases increasing. This included both the emergence of newly confirmed diseases such as the dengue hemorrhagic fever, as well as the resurgence of long-known diseases such as malaria that were thought to be under effective control (Gratz, 1999).

Climate change has been considered to be one of the most important reasons for the spreading of vector-borne diseases (Caminade et al., 2019; J. M. Medlock & Leach, 2015; Semenza & Suk, 2018). Most disease vectors such as mosquitos, ticks, and fleas are poikilotherm and are sensitive to temperature. The warming climate may accommodate their survival. Lyme disease, a tick-borne disease, has been found extending its range to Scandinavia and Canada (Bouchard et al., 2015; Ostfeld & Brunner, 2015). While ticks are vulnerable to cold weather, the increasingly moderate and humid winters, as well as reduced seasonal temperature variation have facilitated their spreading to cold regions (Bouchard et al., 2015). Another example is the West Nile Virus which is transmitted by *Culex* sp. mosquitos. Temperature anomalies above average have been found to be associated with the outbreaks (Paz et al., 2013). It was also discovered that the mosquito population and the transmission of West Nile Virus were also related to other environmental factors such as vegetation, water, and bird migration (Marcantonio et al., 2015). While these factors can also be affected by the climate, the mechanism that mosquito population is impacted by climate change can be a highly complex process.

Another reason contributing to the increasing risk of vector-borne diseases is rapid urbanization (Zayed, 2020). While large cities can usually provide better infrastructure and healthcare, this is not always the case in newly urbanized areas, especially in developing countries. The construction of infrastructure and healthcare facilities may fall behind the expansion of urban areas, resulting in poor living conditions (Neiderud, 2015). Moreover, sprawling urban areas can also shatter existing wildlife habits. Habitat fragmentation not only disturbs the ecosystem but also creates more opportunities for humans to come into contact with disease hosts in close range (Hassell et al., 2017). Both climate change and urbanization have put up challenges to existing disease control, demanding a system that is able to respond to the rapidly changing environment.

2.2.2 Integrated Pest Management (IPM)

The term biological control refers to the suppression of the population of pests or weeds using living organisms, usually their natural enemies or predators (Heimpel & Mills, 2017). While being considered as an “environmentally-friendly” alternative to insecticides, biological control has not been adopted universally (Thomas, 2018). In many cases, there is a lack of evidence of their effectiveness (Lazaro et al., 2015; Walshe et al., 2017). A more common concern is that the introduced control agent can become invasive and cause havoc in the ecosystems (Ayala et al., 2007; Messing & Wright, 2006). Because of the complexity of ecosystems, the effects of biological control need to be analyzed in a broader context (Barratt et al., 2018). As a result, modern efforts of biological control are usually amalgamated into a more comprehensive approach names Integrated Pest Management (IPM) (Flint & Van den Bosch, 2012).

Developed to reduce the usage of broad-spectrum insecticides in agricultural production, the IPM is an ecologically-based pest control approach that aims to maximize the effects of every natural mortality factor (Flint & Van den Bosch, 2012). The term ‘integrated’ implies that the level of natural enemies is taken into consideration during the decision-making process. Instead of simply introducing the biocontrol agents, IPM uses compatible, non-disruptive methods to increase the population of natural enemies (Ehler, 2006). While the usage of pesticides is not prohibited in an IPM approach, they are applied selectively and timely to increase effectiveness and reduce collateral damages (Barzman et al., 2015). An IPM is expected to provide economic savings and protect both the environment and human health (Ehler, 2006). One example is the control of corn earworms in sorghum production in Australia (Franzmann et al., 2008). By creating a pest-resistant hybrid sorghum cultivar and adapting the nucleopolyhedrovirus as a biopesticide, the usage of synthetic pesticides had been reduced greatly, leading to significant economic and environmental benefits. An IPM requires thorough understanding of the ecology and continuous monitoring of the population of both the pest and its natural enemies. The decision-making process needs to be able to respond quickly when pest level reaches certain thresholds (Barzman et al., 2015). Therefore, a sophisticated data collection, processing, and decision support framework can provide significant help to IPM approaches.

2.3 ECOSYSTEM SERVICES IN LANDSCAPE DESIGN

The term ecosystem services refers to any benefits humans can gain from a healthy, functioning ecosystem (De Groot et al., 2002). Typically, ecosystem services are classified into three categories: 1) provisioning: material goods that humans can gain from ecosystems such as fruit and firewood; 2) regulatory: ecosystems act as regulators of natural processes, such as water purification and storm protection, and 3) cultural: ecosystems used as places of cultural, spiritual and recreational activities (Reid et al., 2005). While ecosystems may be less robust in cities, their services such as shading, climate regulation, stormwater mitigation, as well as aesthetic experiences and recreational opportunities are still essential in the lives of urban residents. Despite not usually captured in the market, the economic value of ecosystem services can be significant (Daily, 2013). An estimation by (Costanza et al., 1997) indicated the total value of the world’s ecosystem service could be US$16-54 trillion per year, higher than the total gross national product which was around US$18 trillion per year at that time. Compromising ecosystems in city development can lead to long-term economic disadvantages as man-made services need to be provided to make up for the loss of the services typically provided by healthy ecosystems.

Although the concept of ecosystem services is relatively new, it has been a part of the lexicon in landscape architecture for some time (McHarg, 1969). Maintaining healthy and functional ecosystems in an urban setting, however, can be a challenging task because of the continuity of disturbance from human activities. In addition to the three types of ecosystem services, a more recent concept has identified a fourth type, supporting services (Reid et al., 2005). It describes the services that help the ecosystem to sustain itself, such as primary production, habitat formation, or nutrient cycling (Swinton et al., 2007). Although humans do not benefit directly from these services, without adequate supporting services - provisioning, regulatory, and cultural services can not exist (Reid et al., 2005).

The relationship between humans and ecosystems is not a one-way flow but reciprocal. This has led to the concept of human services to ecosystems (Comberti et al., 2015). It describes the continuous investment needed to sustain adequate ecosystem services. For example, management efforts such as habitat protection, weeding, planting, or fertilizing can be critical in maintaining a thriving ecosystem and provide better services to the residents. One critical consideration is how these efforts are applied during the process of landscape design.

The nature of ecosystem services makes them key elements in research on how human activities are affected by environmental changes. Previous studies on ecosystem services explored its concept and implication(Yu et al., 2020), evaluation approaches (K. Fang et al., 2018; Long et al., 2020), and the impacts of ecosystem services value change (Wang et al., 2019;

Xie et al., 2018; Xu et al., 2019). There has been increasing attention on assessing the response of ecosystem services to the process of urbanization, which is crucial for developing successful sustainability-driven landscape design and management strategies and improving the quality of life. For example, (Barthel et al., 2010) focus on social memory in relation to management practice that sustains ecosystem services, and analyze such memory of allotment gardens in the Stockholm urban area, Sweden. (Reyers & Selig, 2020) use a social-ecological systems framework to develop four recommendations for targets that capture the interdependencies between biodiversity, ecosystem services and sustainable development. Overall, understanding the dynamics of ecosystem services supply under a changing environment (driven by both natural and anthropogenic forces) is essential to evaluate how daily lives and welfare of the residents can be affected by the changes.

2.3.1 The Delivery of Ecosystem Services

In addition to the availability and quality of ecosystem services, accessibility is also an essential factor determining how much of the services can be received by the residents (Ala-Hulkko et al., 2016). The accessibility to ecosystem services can be represented as simple as a Euclidean or linear distance (Larson & Perrings, 2013). For example, greeneries within a radius of 500 meters from a dwelling were considered to be easily reachable by the residents (Czembrowski & Kronenberg, 2016). Nevertheless, quantifying accessibility with linear distance is considered to be an oversimplification in urban areas with complex transportation networks and different types of obstacles. As a result, the results of the estimation can be less accurate (Higgs et al., 2012). Network distance, in comparison, is much more powerful in simulating actual travel routes within a city (Du & Zhang, 2020). It should be notified that the transportation network in a city is not homogenous, the travel speed varies in different parts of the network.

Therefore, some studies have argued that the most accurate method to estimate accessibility is the travel-cost approach (Salonen & Toivonen, 2013). This model will simulate both the travel route in the network and the travel speed in different segments of the route, calculating the actual travel time.

Some of the recent studies have attempted to evaluate the quality and accessibility of ecosystem services at the same time, and developed indexes to present both factors together. (Paracchini et al., 2014) rated the cultural services in Europe with an index named Recreation Opportunity that was determined by both the cultural service provision by the green spaces, determined by the naturalness, the protection status, and water presence; as well as their distances from residential areas. Another example by (Stessens et al., 2017) represented the quality of urban green spaces by their sizes and sub-qualities such as biodiversity, quietness, or facilities. The accessibility was estimated by the travel time through the path network. Similarly, both accessibility and quality were combined into one indicator. These studies have provided important insights into the spatial disparity of ecosystem services delivery, helping the planners to understand where green spaces are most needed.

2.4 IMPACT ASSESSMENT AND ECOLOGICAL DESIGN

Landscape and cities are constituted of complex systems, both social and ecological. A meticulous understanding of these system will provide important insights that can turn to design principles. Comprehensive social-ecological assessment considers multiple economic, land use, transportation, climate, and other environmental factors to evaluate to what extent the design and management practices will exert ecological impacts. Such models aim to understand the interactions of human activity and the environment in urban systems. Investigating complex urban social-ecological systems where complex relationship exists between urban growth and ecological impacts are of great theoretical and practical significance in promoting sustainable urban resource management (Yang et al., 2020). For example, Pan et al. (2020) coupled human and ecological processes in a social-ecological modeling framework to understand greenhouse gases emissions associated with urbanization and human-driven land-use changes, while the modeling framework is less generalizable than this study. Xu et al. (2019) comprehensively reflect how ecosystem services response to rural-urban transitions in terms of the change of proportions of urban land, changes of available land, and land use types. These prior researches have pointed out the close relationship between ecological impacts and intensified human activities within large, complex urban systems and how a system thinking approach helps us to understand the cleaner production of the urbanization processes and urban systems.

A spatial-explicit social-ecological model can help to explain not only the magnitude and intensity of growth-induced ecosystem service loss, but also where such ecological degradation is more significant, thus inform targeted sustainable urban management and help to craft localized planning policies (Nesshöver et al., 2017). For example, Pan et al. (2019) found geospatially varied changes in ecosystem services by integrating disparate types of geospatial data under multiple policy scenarios. Furthermore, increased spatial granularity and incorporation of feedback dynamics in the model can better simulate uncertainties in real life.

2.4.1 Comparative urban studies

A comparative approach focuses on uncovering rich details in the context and features of two or more instances of specific phenomena through discovering contrasts, similarities, or patterns across the cases (Yin, 2009). This is of particular significance in urban studies because understanding the varied structural elements and behavioral preferences across multiple cities can improve and extend the urban management approaches and further adapt planning policies.

A theory of comparative urbanism was developed by (Abu-Lughod, 2000) and later evolved by (Lange, 2012) who proposed a framework of methodologies of comparative analysis. Lange marked that there could be intersections between direct comparative methods with other approaches that had been widely adopted by planners, such as case study and social-science analytical methods.

A common challenge for comparative urban study is to develop a reproducible and systematic methodology (Davis, 2005). McFarlane & Robinson (2012) present a set of methods for comparisons between different cities, including: tracing a connection between different locations, exploring the replication of similar phenomena across different contexts, and comparing similar or different outcomes across more than one city. Fossheim & Andersen (2017) present a systematic review method that compares the contents, topics and focuses of sustainable policies in different countries based on a set of predefined criteria. Recent research has focused more on quantitative, multi-scale and spatial-explicit methods (Lopez-Carreiro & Monzon, 2018; Xu et al., 2019; Yigitcanlar et al., 2015). As geographic data become more widely available and computing software become more user-friendly, key technical barriers towards more detailed and systematic comparative urban analysis have been elevated.

2.5 CARBON MANAGEMENT IN LANDSCAPE DESIGN

Carbon dioxide (CO_2) constitutes only a small fraction of the atmosphere. Despite its low concentration, the gas plays a very important role in shaping the Earth's environment, not only because it is a vital ingredient of photosynthesis, but also because of its contribution to the greenhouse effect. In the pre-industrial era, the atmospheric concentration of CO_2 had been oscillating between 140 and 300 ppm. The trend of CO_2 concentration was found to be strongly associated with global temperature (Lüthi et al., 2008). In natural condition, CO_2 can be released into the atmosphere by combustion, respiration, or decomposition. The atmospheric CO_2 can also be removed by photosynthesis or dissolving in the ocean (Battin et al., 2009). While the CO_2 concentration remained stable until the early 1800s, the equilibrium had been broken since the industrial revolution. Human's dependence on fossil fuels has ended up in extreme amounts of emission CO_2, far more than the capacity of natural sequestration processes (Blunden & Arndt, 2019). The global CO_2 concentration reached 416 ppm in March 2021 (NOAA Global Monitoring Laboratory, 2021). To mitigate the warming effect of greenhouse gas emissions, many countries, states, and cities have proposed their climate plans, aiming to re-establish the balance of natural carbon cycle. The Illinois Climate Action Plan (iCAP, https://sustainability.illinois.edu/wp-content/uploads/2020/10/iCAP-2020-FINAL-WEB.pdf) has set up a goal to reach zero-emission, or carbon neutrality by no later than 2050. This goal will be achieved by two means: to reduce emission by phasing out the usage of fossil fuels, and to increase the sequestration capacity.

Carbon sequestration refers to the removal of carbon dioxide from the atmosphere and preventing the stored carbon from being released back to the atmosphere in long term. While carbon sequestration can happen through geological processes, biological sequestration, which converting atmospheric CO_2 into biomass through photosynthesis, is considered to be much more cost-effective and controllable (Farrelly et al., 2013). Since vegetation is the vital element of both carbon sequestration and landscape design, landscape architects have the potential to contribute to climate change mitigation by promoting better sequestration in their designed spaces. Nevertheless, the current knowledge and resources of carbon sequestration have not been optimized for landscape architects, and certain interdisciplinary communications and translations are required to fill the gap (Lynn, 2021).

2.5.1 Estimating Forest Carbon Sequestration

Forest is one of the most effective terrestrial ecosystems in terms of carbon sequestration (Shuguang Liu et al., 2014). When proposing the climate action plan, it is very helpful to know the carbon sequestration potential of the forests, both the current value and the future trend. The accumulation of biomass happens when a tree is growing. When the size of a tree is known, a simple approach to estimate its sequestration potential is to apply empirical formulas between the diameter and biomass (Kiran & Kinnary, 2011). A more comprehensive study by (Strohbach et al., 2012) incorporated a life cycle approach to predict the sequestration potential by landscape trees in long term. A tree's efficiency in accumulating biomass varies among its life stages. While young trees are usually highly efficacious in carbon sequestration, their growth can come to a stall when reaching the mature stage. In old trees, decomposition can suppress growth, resulting in a net emission instead of sequestration. By applying the growth curve, the authors managed to project the trend of sequestration potential for the next 50 years. In addition to the sequestration by trees, other factors affecting carbon flux such as thinning, mortality, as well as energy consumption during maintenance. The site's carbon footprint was compared under several different scenarios and management plans. One limitation of these growth model-based estimations is that a detailed tree inventory is required. A more recent study by (Zhao & Sander, 2015) overcame this shortage by estimating the numbers and sizes of urban trees using LiDAR data. Overall, when assessing carbon sequestration potential is still a challenging task, advanced data and computational science is providing increasingly powerful tools to make more reliable estimations and predictions.

While the carbon sequestration potential from a single tree can be estimated and predicted using empirical formulas, a naturally-grown forest is constituted by trees in different sizes and species. As a result, assessment of carbon sequestration from forests can be more complicated, requires a thorough understanding of forest stand structure, such as species diversity and age composition (Mensah et al., 2016). Such studies can start from collecting information (such as species and diameter at breast height) of trees within sampling plots regularly, tracing changes in tree sizes and species. The above-ground biomass (AGB) and thus the estimated carbon sequestration can then be calculated (Nizami et al., 2017). Understanding the relationship between forest stand structure and the carbon sequestration potential helps to provide useful information to forest managers, such as identifying species that are contributing to most of the sequestration values.

CHAPTER 3: OPTIMIZING THE BENEFITS OF URBAN GREEN SPACE UNDER THE SMART LANDSCAPE FRAMEWORK

3.1 INTRODUCTION

Urban green spaces and natural areas are common objects of landscape design. These areas provide vital services to urban residents including climate and water regulation, disaster mitigation, and recreational opportunities (see Section 2.3 for more examples). While ecosystem service acts as a useful concept that has helped to define the value that ecosystems provide to human populations, its application to typical landscape design processes has been limited. To promote landscape designs that more readily consider and apply established ecological principles and useful constructs such as ecosystem services, scattered ecological datasets must be integrated and translated into a design centric philosophy in a language that landscape architects can understand and easily access (for more detailed explanations about the disconnection between ecosystem services and landscape design, and how the development of ICTs may help to fill the gap, see Chapter 1).

In this study, I propose a smart framework to help designers usefully access and utilize fragmented data (geospatial, environmental, ecological) from different sources (**Figure 3**). The framework utilized a planning support system, the Landuse Evolution and Impact Assessment Model Planning Support System (LEAM PSS), to connect design decisions to complex urban ecological models in an easily accessible way. The goal of the project was to turn discrete, unorganized data into suggestions that landscape designers can use to promote the wise management of ecosystem services in their projects. The framework has been tested in two different approaches: a stress analysis to identify natural areas that are susceptible to disturbances caused by urban development, and a scenario analysis to identify potential urban development plans that can provide adequate housing while preserving the most important natural areas (those with high ecosystem service values).

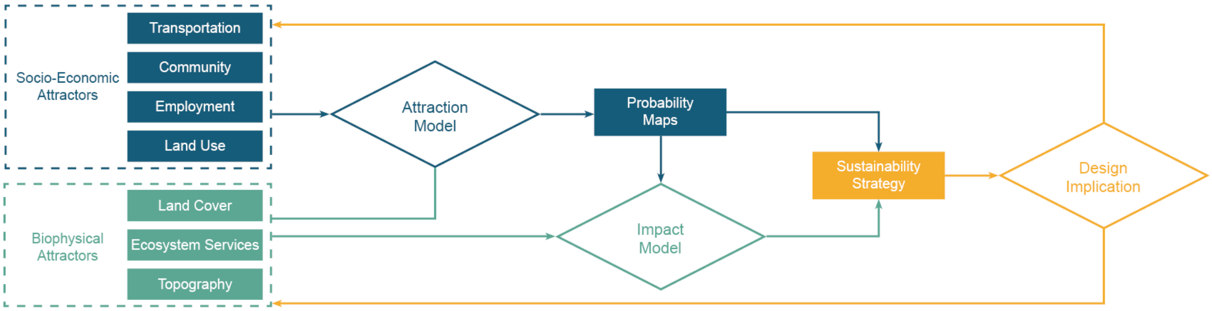


Figure 3 A conceptual framework of smart landscape that couples the LEAM PSS and ecosystem service impact models.

3.2 RESEARCH OBJECTIVE

This study aims to develop a quantitative analytical framework based on the smart landscape concept, which combines planning support systems, landscape stress analysis, and ecological modeling in support of decision making in land design and planning. The analytical framework is tested on a potential community development project to illustrate how designers can benefit from easy access to such information via planning support system models.

3.3 THE LANDUSE EVOLUTION AND IMPACT ASSESSMENT MODEL PLANNING SUPPORT SYSTEM (LEAM PSS)

LEAM is a dynamic spatial model based on network analysis. It is capable to estimate the probability of future developments as a result of biophysical (such as water and forest) and socio-economic factors (such as population, employment, and transportation). The philosophy and advantages of the LEAM PSS has been delineated previously. In this section I focus on some of the underlying mechanisms of and dynamics of the quantitative model in order to explain it potential connection to design decisions. Basically, factors influencing the location-choice of future developments are defined as “attractors”. The assumption of the model is that the attractors influence development probabilities in a gravity-type function in which the attraction power decays with increased travel time (using multiple modes). This gravity function can be determined through a shortest distance and time algorithm with data from various sources. The likelihood of development is determined by a set of accessibility scores to different types of attractors. Typically, the development probability of each land use cell from undeveloped land to developed land is defined as:

$$P_{k,t} = f\left(L_{k,t}\right)\left(\sum g(c_{i,k,t}) + N(\theta_{k,t-1})\right)$$

(Formula Credits: Haozhi Pan, Cong Cong)

Where $P_{k,t}$ is the developmental probability for land-use cell k at time step *t*; $f\left(L_{k,t}\right)$ is the function of land use zoning effect and biophysical factors that restrict growth on certain types of lands for land-use cell k at time step *t*, $c_{i,k,t}$ is the connectivity of cell k to urban attractor type i and $g()$ is the function of mapping the connectivity value to probability of land use change; $N(\theta_{k,t-1})$ is the function that converts number of cells neighboring cell k at time step $t-1$ to a probability number. Adding probability to cells neighboring existing development can result in a more organic growth pattern (X. Li et al., 2017).

After calculating the developmental probability $P_{k,t}$ of all undeveloped land use cells k at time step t, the model allocates commercial and residential growth by:

$$D_{k,t} = \begin{cases} 1, & \sum_{m\ for\ all\ P_{m,t} > \ P_{k,t}} d_m < \ E'_t \\ 0, & \sum_{m\ for\ all\ P_{m,t} > \ P_{k,t}} d_m \geq \ E'_t \end{cases}$$

(Formula Credits: Haozhi Pan, Cong Cong)

Where $D_{k,t}$ represents whether cell k at time step t develops (1 for develop and 0 for not develop); $d_{m,t}$ is the density of employment of each land use cell; E'_t is the total growth demand at time step t specified by the official planning document and information for each city.

3.4 IMPACT ASSESSMENT BASED ON THE LEAM PSS

The LEAM model helps to conduct ecosystem service impact analysis in two ways. First, natural areas and urban green spaces providing ecosystem services can be considered as attractors and their effects on future urban development can be projected. Second, the final LEAM change map allows an assessment of potential environmental impacts caused by future development so preventive measures can be taken before the actual damages might occur.

Ecosystem service values (ESVs) were used to represent the importance of a range of natural area functions in an urban setting Several approaches are typically applied to estimate the values of each type of ecosystem service (for a detailed description of ecosystem services types, see Section 2.3). The estimation of provisioning values is usually straightforward, using the market value of the products (Costanza et al., 2014). The value of regulatory services can be estimated by replacement-cost method (Costanza et al., 2008). For example, the value of water purification service of a wetland is equal to the construction and maintenance cost of a purification plant with the same capacity. It is difficult to make explicit estimation of the cultural services since they are usually non-market values. But it can still be estimated by people's willingness to pay (Spash, 2008).

In this study, the National Land Cover Database (NLCD) 2011 land cover map for greater Chicago region is used to determine land cover type for ecosystem service analysis purposes. While a newer set of land cover data (2016) was available, this study was a part of a more comprehensive project that aims to develop the LEAM PSS itself, including a model calibration process. The changes in land uses were simulated using previous data. The model prediction result was then compared with actual urban development in the real world for calibration and validation. The LEAM PSS for Stockholm uses the 2011 land cover data set.

The ESV of each landcover type was based on literature review using a searchable database (Van der Ploeg et al., 2010). Because of the complexity of ESVs, one type of land cover usually provides many different kinds of ecosystem services. For example, forests have provisioning values by providing fruits and firewood, regulatory values by providing wind protection and cooling, and cultural values by providing recreational spaces (Krieger, 2001). Research projects of ecosystem service values, on the other hand, usually focus on one or a few types of series. In this study, I reviewed publications to find estimations of different types of ecosystem services values, which were added up to get a total value of the certain land cover. Newer publications with a setting in Midwestern United States were preferred. If a location-specific estimation was unavailable, a more general estimation under a similar climate zone would be accepted for calculation.

The impact analysis overlaps the LEAM probability maps with the ecosystem services value map. LEAM probability maps provide information of when and where development might occur. This helps identify areas - in this case areas of high ecological importance, that may be stressed by the potential changes. It also allows an analysis of the potential loss of various eco-system services. Once impacts are established, mitigation or compensation design interven-tions can be developed. I adapt these LEAM probability surfaces to quantify the stresses that potential development has on ecological resource areas. LEAM estimates the probability of development in each succeeding year for every 30m x 30m cell in the area of study. This raster map represents the spatial distribution of probability within a specific geographic area. I use this distribution to statistically analyze how it impacts the resource of study - in this case, ecosystem services. I use the 80th, 90th, or 95th percentile (i.e. the top 5%, 10%, and 20% of the study areas with the highest probabilities of change). These are classified as ‘stressed’. **Figure 4** overlays developmental probability surface with high performing ecosystem service areas to exhibit how these ecologically important areas might be affected. Design interventions, such as setting up no-growth zones or buffers that help to preserve important ecological resources can be tested and verified simply using this modeling framework.

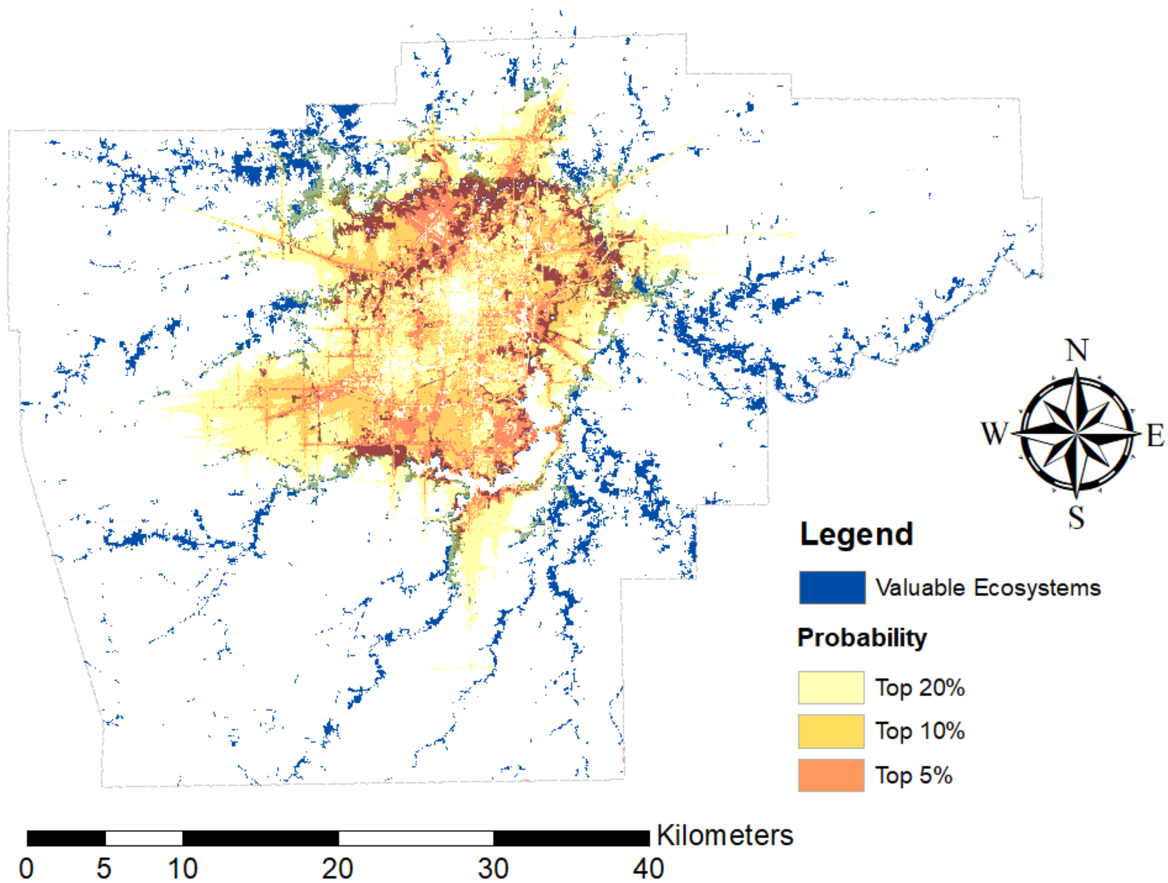


Figure 4 A stress test of Sangamon County illustrating how ecologically important areas (forests, wetlands, and riparian areas) may be affected by potential development. This is an example of LEAM-based impact analysis.

The framework makes ecosystem service calculations available for design decisions using a 'what-if' design intervention-based scenario approach. Deign scenarios can be tested using the decision framework (Pan, Zhang, et al., 2019) through a prototype API driven interface. **Figure 5** shows the interface describing how future developments may affect carbon sinks in the Stockholm, Sweden region (http://www.leam.illinois.edu/stockholm2017/impact_assessment/impact_assess.html).

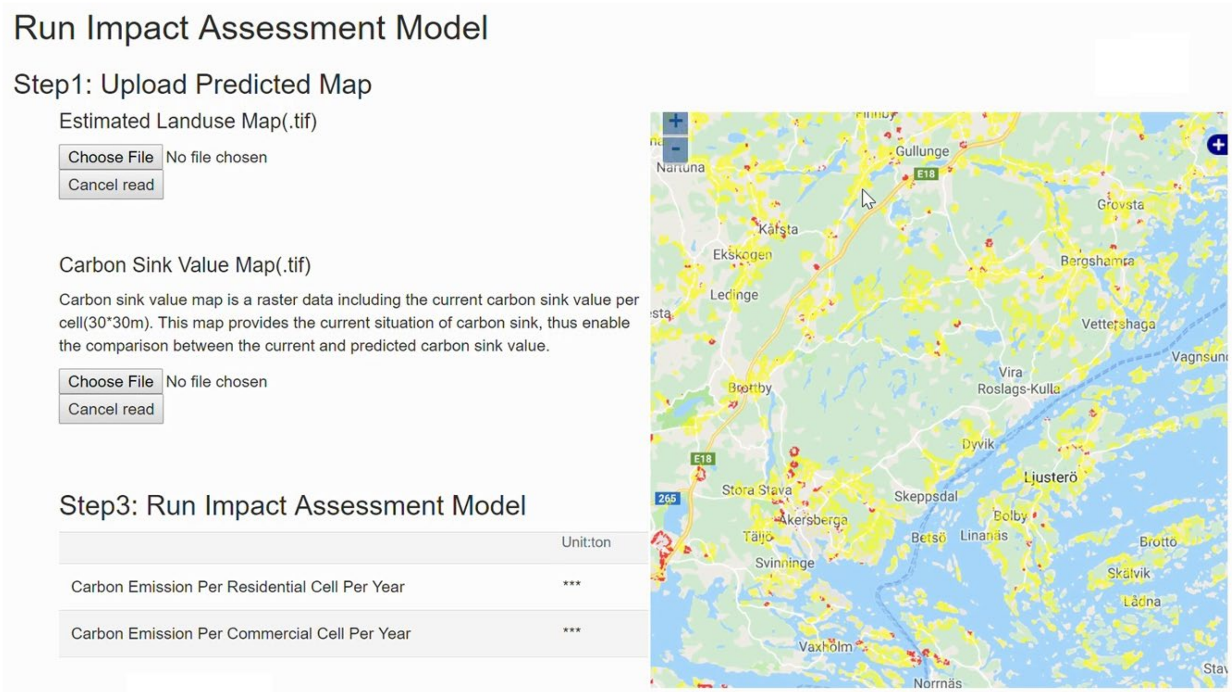


Figure 5 Stockholm impact assessment model API to estimate and visualize the loss of carbon sinks.

3.5 APPLICATION OF THE FRAMEWORK IN SANGAMON COUNTY

The smart framework is tested in Sangamon County, Illinois in a scenario planning analysis. The study area features a complicated mosaic of land use types over 877 square miles in central Illinois. The senescence and growth of urban, residential, commercial, industrial, and agriculture land uses are occurring simultaneously in varying parts of the county. The area also hosts valuable natural areas including forests, wetlands, and riparian zones. Many of the natural areas and urban open spaces are interlaced with rural developments. While providing important ecosystem services, the natural areas in the county are vulnerable to a range of disturbances. A LEAM PSS forecast (to 2050) is used to identify the timing and location of potential changes in land uses. It is also used to establish the potential developmental stresses likely to be placed on regional ecological assets and the possible impacts on ecosystem services (i.e. wildlife habitat, flooding protection, and carbon sequestration).

To demonstrate how this framework can help landscape designers, I tested our impact analysis model on a potential residential development project in Sangamon County. Regional economic models suggest that employment in the County will grow for the next several decades and additional housing is needed to accommodate new workers and their families. A potential site for development of additional housing units was identified using LEAM probabilities. LEAM growth maps indicated high probability scores of community facilities (such as school, medical facility, and police station), industrial areas, and urban commercial areas at a site of about 1,630 ac (6.59 km2) located to the east of the downtown area of Springfield (**Figure 6**). Local planners confirmed the site's relative attractiveness for future development with easy accessing to current developments as well as future community services, employments, and commercial areas. Nevertheless, the site was also close to Lake Springfield, with a significant portion of the area covered by ecologically important wetlands and riparian forests. Moreover, more than a half (915 ac) of the area was in medium to high flood risk. The complexity of the land had made the residential development a challenging task for the designers. Multiple factors need to be taken into consideration, and tradeoffs need to be made.

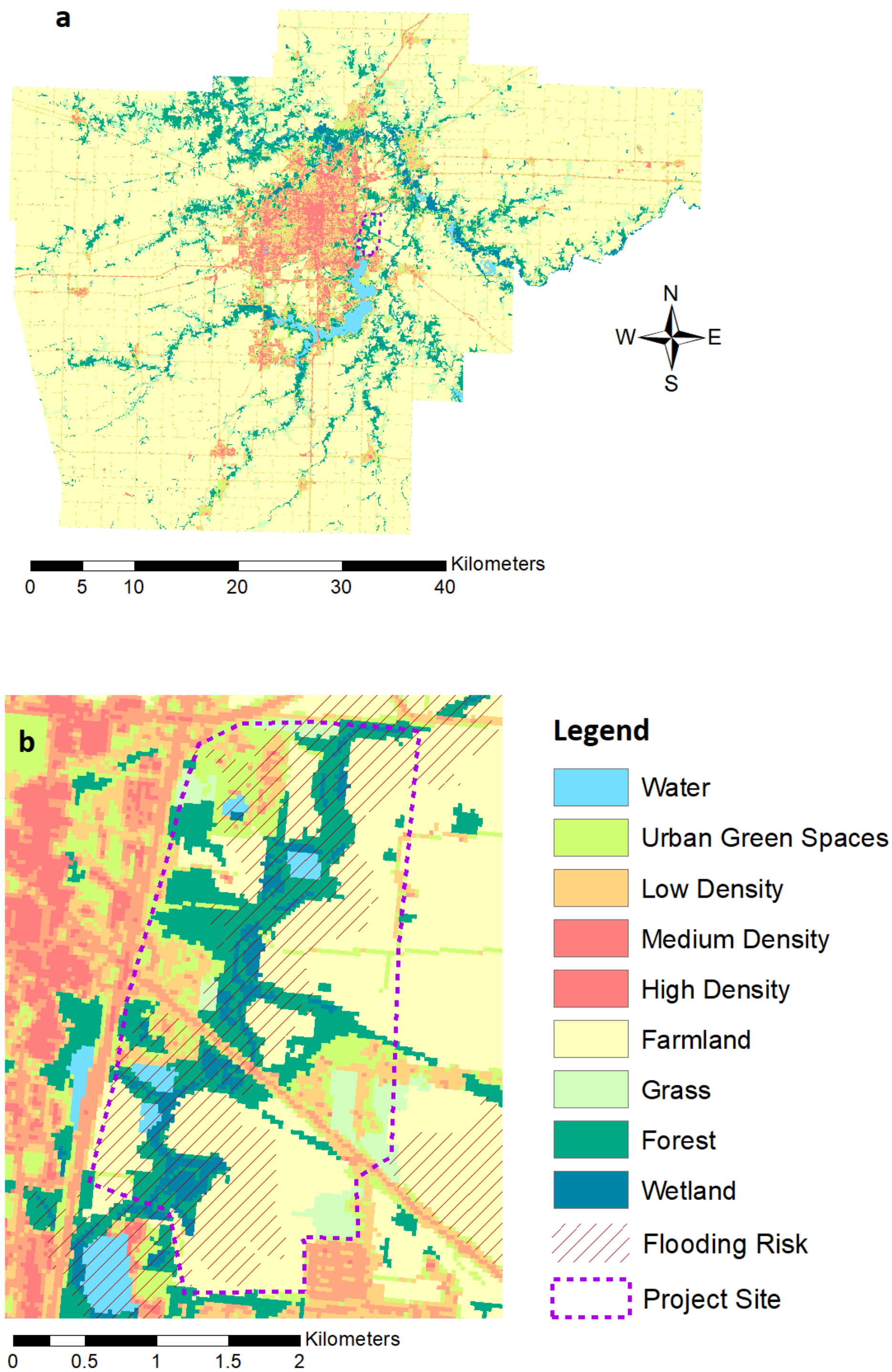


Figure 6 a) Current land use of Sangamon County, b) current land use of the project site – slated for housing developments. The hatches indicate areas with high flood risk.

A successful design of a residential community under such an environmentally sensitive setting requires a balance between housing, conservation of ecosystems, and flood safety. With this in mind, I set four goals for the study: 1) provide at least 3,000 housing units, 2) total ecosystem services value shall be at least 954,048 USD/yr (95% of current value), 3) total carbon sequestration shall be at least 390,776 kg/yr (95% of current value), 4) no developments in areas of high flood risk. I analyzed the current condition and 3 different development scenarios with the impact analysis model. The total ecosystem services value and carbon sequestration were calculated using the land cover map. For each type of the land cover, its ecosystem services value(Van der Ploeg et al., 2010) and carbon sequestration (Zhu & Reed, 2014) were assessed. The flooding hazard map was acquired from the National Flood Hazard Layer (NFHL), provided by the Federal Emergency Management Agency (FEMA, https://www.fema.gov/flood-maps/national-flood-hazard-layer). Flooding risk was determined by percentage of the residential development within areas under medium to high risk of flooding. The results were shown in (**Figure 7**).

Current Condition. The project site was largely undeveloped, with very few scattered residents (**Figure 6**). The estimated population was 212 (Falcone, 2016), or 98 housing units. The area was ecologically valuable, hosting 346 ac of forests and 112 ac of wetlands. The total ecosystem services value was 1,004,262 USD/yr, with a carbon sequestration value at 411,343 kg/yr. The rest of the site were mainly farmlands.

Scenario 1. Low Density Development: Under this scenario, 60% of the site area would be converted into single family houses with a density of 2 housing units per acre, which were common in suburban areas in the US. Because of the abundant green spaces within the low density development, the total ecosystem services value and carbon sequestration increased after the development. Nevertheless, the low density also mean only about two-third of the desired housing units could be provided, while further expanding the development would inevitably damage the forests and wetlands. Another disadvantage of this scenario was that a significant portion of the developments would face flood hazards. Besides, low density developments can augment the sprawling, and the increased emission from the transportation can naturalize the additional carbon sequestration brought by the development.

Scenario 2. Mid-High Density Development: Under this scenario, there would be two types of developments: medium density single- and two-family houses with an average density of 4 housing units per acre, and high density multi-family houses or townhouses with an average density of 6 housing units per acre, each of them could take up to 20% of the total area. This scenario would provide enough housing while the flooding risk would be relatively low. Nevertheless, the environmental impact would be significant, to the extent that it could not be compensated by green roofs.

Scenario 3. Very High Density Development with Green Roofs: Under this scenario, the development would be multi story buildings with an average density of 9 housing units per acre, covering 22% of the site area. In this way, adequate quantities of housing might be provided while minimizing the risk of flooding. In addition, density could be restricted to less ecologically sensitive areas to minimize environmental impacts. In this case, 27.6 acre of green roof (7.7% of the impervious area) would be enough to compensate the loss of carbon sequestration. With enough green roofs, this will be the desired scenario that fulfills all of the goals.

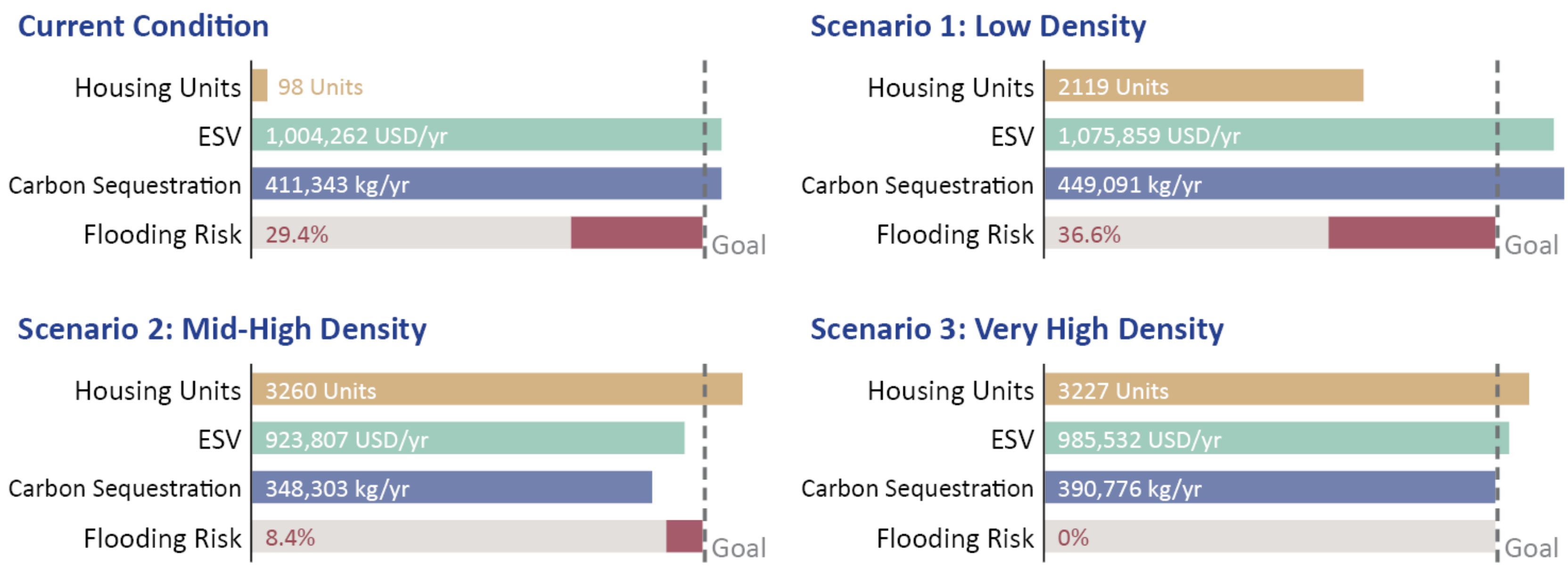


Figure 7 Comparison of the housing units provided, ecosystem services value (ESV), carbon sequestration, and flooding risk among current condition and 3 different scenarios.

3.6 CONCLUSION

In this study I have demonstrate a framework for applying a smart landscapes approach using an available and accessible PSS. This framework is based on a similar set of data (such as land cover and digital elevation maps) that are commonly utilized by landscape architects in their site analysis processes. Instead of jumping to design conclusions intuitively however, the PSS allows the interpretation of the landscape design using quantitative reasoning. This is an advantage that can readily be achieved by smart approaches. Hidden information not revealed explicitly by the data can be revealed by extensive (yet automated) manipulation models. In this case, I was able to determine how much of the ecosystem services and carbon pools might be affected and how many of green roofs could compensate the loss. This not only helps the designers to better understand their project sites, but also introduces better reasoning process, making the design decisions easier to defend when facing clients and stakeholders.

This study generally follows my early conceptual model of the smart landscape framework (**Figure 2** top). Basically, the decision-making process in landscape design is supported by data analysis. At its current stage, the framework was not very different from existing concepts such as evidence-based design (Ulrich et al., 2010) or GeoDesign (Steinitz, 2012). Nevertheless, this study has provided examples that tools, platforms, and methodologies that are developed to support urban planning (such as PSSs) can also be applied to landscape design with proper modification and adaptation. It also leads to two features of the smart landscape framework, “sentient” and “reasonable” (**Figure 8**). A more detailed introduction of these concepts can be find in the next chapter. From a wider perspective, constructing a data framework that is similar to what has been applied in a smart city is possible in landscape design.

In the following chapters, I am going to demonstrate how the smart landscape framework has been further developed from the current stage.

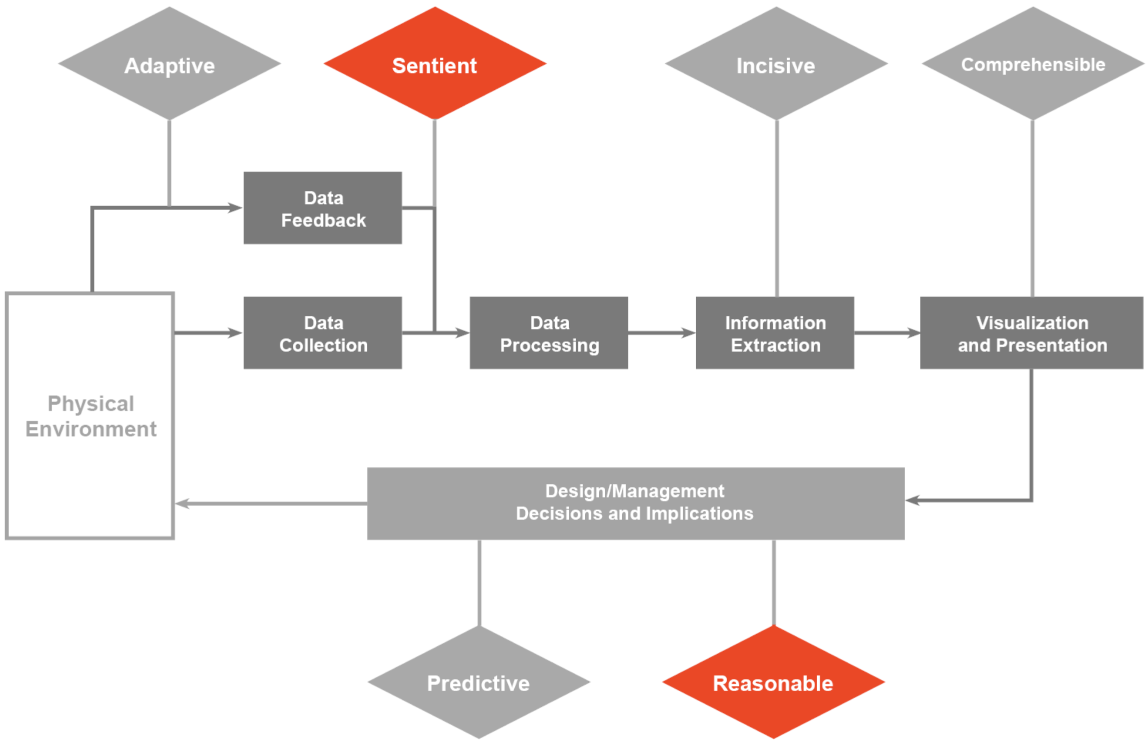


Figure 8 Contribution to the smart landscape framework, Chapter 3 & 4

CHAPTER 4: SOCIO-ECOLOGICALLY INFORMED COMPARATIVE MODELING TO PROMOTE SUSTAINABLE URBAN TRANSITIONS: A COMPARATIVE CASE STUDY IN CHICAGO AND STOCKHOLM

4.1 INTRODUCTION

Sustainability science has been largely unable to inform policy and stimulate actions that will help cities transition toward a more sustainable future (Elmqvist et al., 2019; Gary R. Watmough et al., 2019). One obstacle is the place-based silos from which laboratories have traditionally generated and applied new knowledge. While there is interaction and knowledge exchanged across context in terms of science, there is much less (if any) successful exchange in terms of policy and other implementation responses. For example, approaches to sustainable urban management have typically included the application of best-management practices without a critical understanding of the complex relationships that underlie the systems to which they are being applied—often resulting in failed contextual outcomes (Long et al., 2020; Yang et al., 2020). Understanding the differences in institutional structures, behavioral preferences, and development patterns between regions will help cities achieve effectiveness in their policy implementations.

Comparative study has been an increasingly important tool that helps urban planners and policymakers to evaluate sustainable policy and promote best practice learning. Firstly, comparative studies can reveal contextual differences between places through generalizable methods (Amelang, 2007; J. Lin & Ban, 2017; McFarlane, 2010). Using comparative approaches can facilitate a deeper understanding of underlying issues than if the analyses are conducted within a single context (Davis, 2005). Furthermore, relationships between socioeconomic determinants and urban growth processes are not distinct in different realities or places but show generalizable patterns (Rotmans et al., 2001). This suggests that cities can learn from one another in their practices of developing sustainability strategies. Comparing the efforts of sustainability-minded cities in different national contexts highlight these embedded relationships and offer valuable insights into key factors and conditions that help global urban centers to achieve their stated sustainability goals.

Nevertheless, comparative analyses could fall short of methodological insights if they provide a mere descriptive comparison of similarities and differences between cities in parallel (Pan, Page, et al., 2019). Context specific experience provides a detailed account of the specific research situation but offers limited reference to sustainability strategies in many different contexts. It is more important to understand the relationships underlying the social ecological systems to which sustainability strategies are being applied. In this regard, socio-ecologically informed data collection and analysis has emerged as a reproducible methodological framework to understand crucial human and natural interactions (Pan et al., 2021; Gary R Watmough et al., 2019; You et al., 2020). Social ecological models use space explicit statistical methods to identify urbanization explanatory variables and their relative effect on urban growth, thus address the shortcomings of descriptive comparative study and improves the transferability of the analytical approach.

This study has investigated the city of Chicago, US and Stockholm, Sweden. It is generally perceived that US cities are more sprawled when European cities are more compacted (J. Huang et al., 2007; Larondelle & Haase, 2013). Sprawling refers to the rapid expansion of developments, especially low density suburbs far away from downtown and commercial areas.

Extensive sprawling can lead to negative effects including prolonged time of commuting, increased energy consumption and carbon emission, poor infrastructure, as well as fragmentation wildlife habitats (Frumkin, 2016; Johnson, 2001). The “compact city”, in comparison, is characterized by developments restricted in a relatively small range from the core of the city, featuring high densities and short travel distances (Dieleman & Wegener, 2004; Schwarz, 2010). Nevertheless, standards of telling “sprawling” from “compact” are usually empirical. To better identify the similarities and differences between cities in a comparative study, the pattern of development should be described in a more accurate manner.

4.2 RESEARCH OBJECTIVE

The goal of this study is to apply a socio-ecologically informed comparative model in 2 cities, Chicago and Stockholm, in the analysis of development patterns and environmental impacts. It aims to answer the following research questions in each setting: 1) How the location and probability of development responds to different biophysical and socio-economic factors? 2) How is the natural environment affected by the developments in different patterns? 3) How to mitigate the environmental impacts?

4.3 STUDY AREAS

Chicago, IL, USA and Stockholm, Sweden are taken as comparative case study sites. Stockholm is both the political capital and commercial center of Sweden. Chicago is the largest city in the Midwest United States and one of the most important commercial centers in the country. Both cities have continuous metropolitan areas that extend beyond the municipal borders. In 2016, the estimated population of Stockholm was 930,000 within the city border, while the population of the county was around 2,269,000. The estimated population of Chicago in 2017 was 2.7 million at the city core and almost 10 million for the entire metropolitan area (U.S. Census Bureau, 2017).

4.4 DATA ACQUISITION

I extracted the existing population centers, roadway network (highway, major roads, access points on ramps and major intersections) in Chicago from the U.S. Census data. Existing employment centers in Chicago are from D&B Hoover Industry Directory (Dun & Bradstreet Inc., 2017) and land cover raster data in 30x30 meter resolution are from National Land Cover Database (NLCD). The land cover data will be used to identify land use types including residential, commercial, forest and water. Stockholm data was provided by the Tillväxtoch regionplaneförvaltningen (TRF), which is the Regional Development and Planning department at the Stockholm County Council. This data includes data about existing features of the region (a digital terrain model, the existing population and employment centers, land-use and existing roads and public transport networks), and "no-growth" zones (such as protected natural areas where no development will be allowed). Both land cover data used the latest availability version by the time the study was conducted (2016 for Chicago and 2018 for Stockholm).

Landsat 8 images used to calculate NDVI values were acquired from the United States Geological Survey (USGS) Earth Explorer. The Stockholm images were taken on June 17, 2019, while the Chicago images were taken on July 5, 2020. Ideally, images of both cities should be acquired in the same year to minimize errors such as the influence of the fluctuation of global temperature. Nevertheless, a significant portion of the satellite images was not qualified to be used in the analysis due to high cloud coverage. The images had a spatial resolution of 30x30m, which was the same as the land use data.

4.5 COMPARATIVE MODELING PROCESS

The comparative model is constituted by two steps: 1) using the LEAM PSS to project land use changes in both Chicago and Stockholm by the year 2040 (the philosophy and algorithm of the LEAM PSS has been delineated in Section 2.1.1 and Section 4.3), and 2) forecast and compare impacts on ecosystem service value (ESV) and NDVI. Key socioeconomic (population, employment, and transportation) and biophysical factors (forest and water) that are expected to affect urban development are selected as input variables for the LEAM PSS to predict the land use changes in both cities over a 30-year period until 2040. ESV (presenting ecosystem services provided by the green spaces) and NDVI (presenting the density of vegetation) are two indicators used to assess the environmental impacts. The urban growth maps are overlaid with the ESV and NDVI maps to evaluate the impact of urban development on green spaces and natural areas. This allows us to give suggestions on policies that may be effective for mitigating negative environmental impacts in each city.

The modeling results will be presented in the forms of 1) projected urban growth maps, 2) environmental impacts, as ESV and NDVI maps, and 3) suggested sustainable policy practices.

4.6 EVALUATION OF ENVIRONMENTAL IMPACTS

In this study, the ESV of each type of land cover is determined using a database by (Van der Ploeg et al., 2010) as a reference. The database was a summary of previous estimations of ESVs in different ecosystems. ESVs results were associated with the corresponding land cover type to get the ESV map. The map was then clipped using 2040 growth map as a mask to show the spatial distribution of ESVs within areas that were expected to be occupied by potential developments.

The NDVI value is calculated from Band 5 (near infrared or NIR, 851-879nm) and Band 4 (red, 636-673nm) of the Landsat 8 images by comparing the intensity of infrared radiation with red light (Pettorelli et al., 2005). The formula for calculation is:

$$NDVI = \frac{NIR - Red}{NIR + Red}$$

Similar to the ESV maps, the result NDVI maps were also masked by the 2040 growth map from LEAM to show the spatial distribution of NDVI values across the developments.

4.7 RESULTS

Figure 9 shows the land use forecasts on the year 2040 for Stockholm and Chicago, a result of the LEAM PSS simulation. Most of the new developments in Stockholm are expected to be close to existing urban centers, with a few developments scattering in existing sub-centers with emerging job opportunities and transportation hubs. The land use forecast for Chicago shows that newest developments occur on the urban fringe outside the city center.

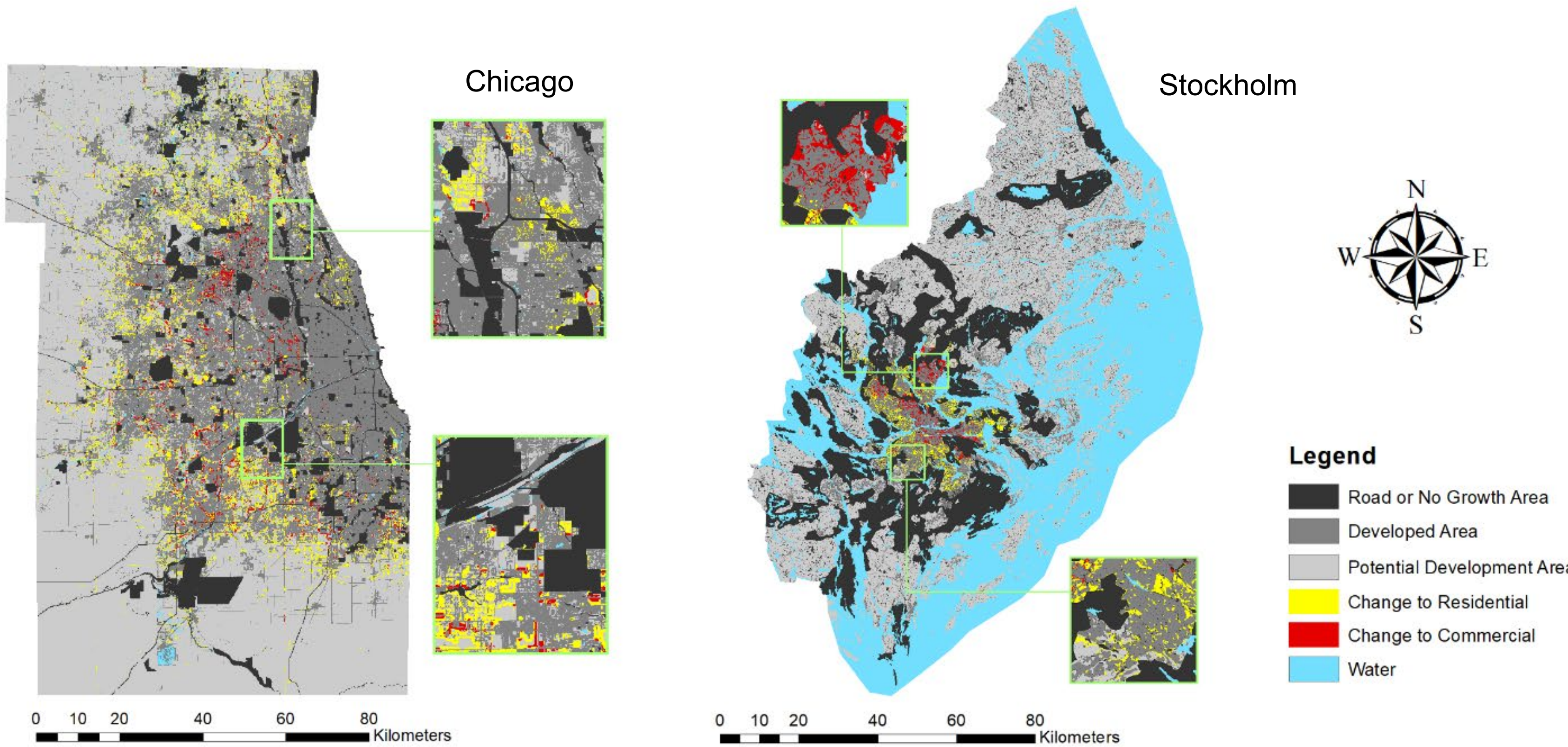


Figure 9 Land use change forecasts by the Landuse Evolution and impact Assessment Model (LEAM) for (left) Stockholm and (right) Chicago for the period 2010–2014.

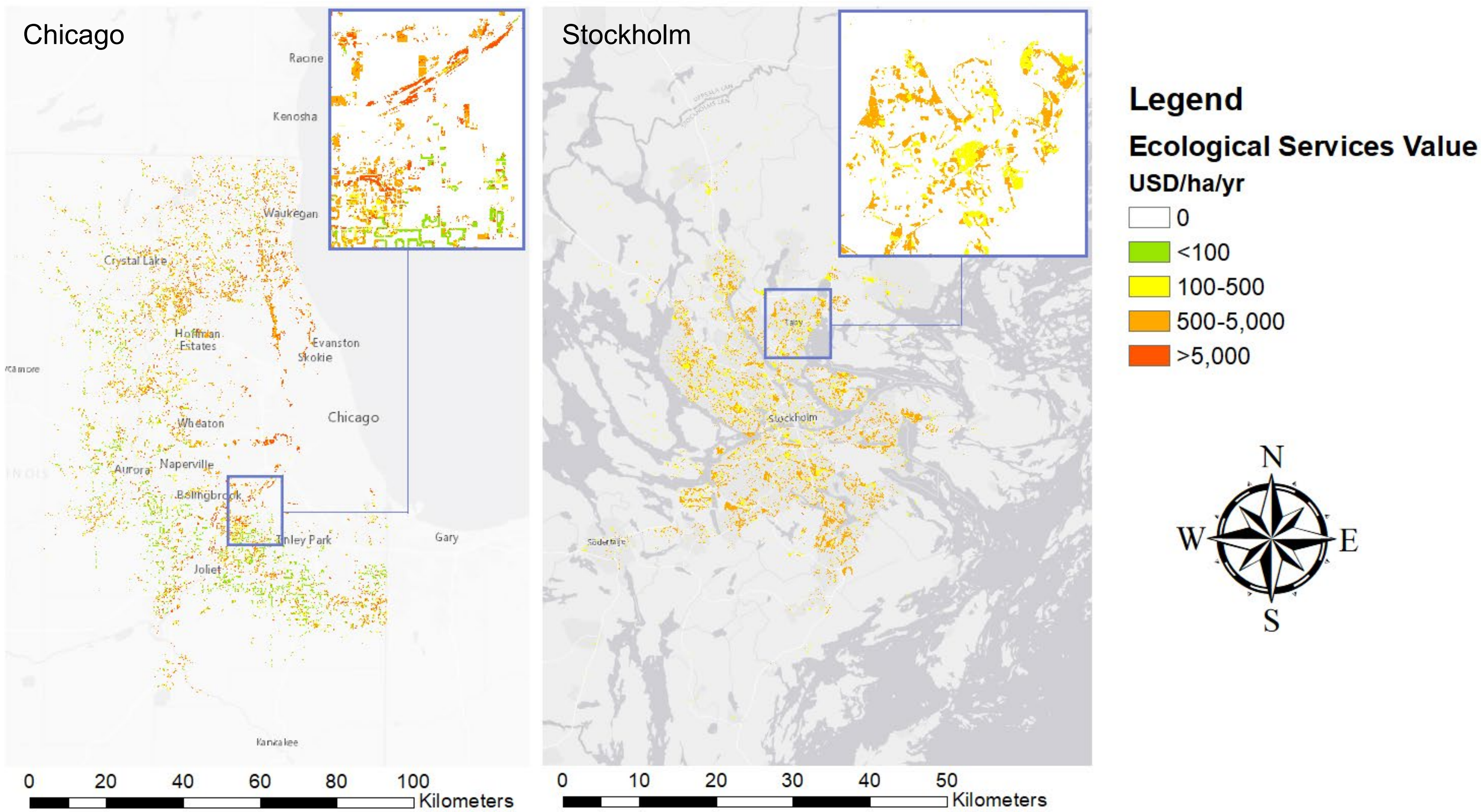


Figure 10 The spatial distribution of ecosystem services values within the potential developments. The scales of the maps are different because the size of the urban area in Stockholm is much smaller than in Chicago. The zoomed-in images show the striped pattern of green spaces in Chicago and scattered pattern in Stockholm.

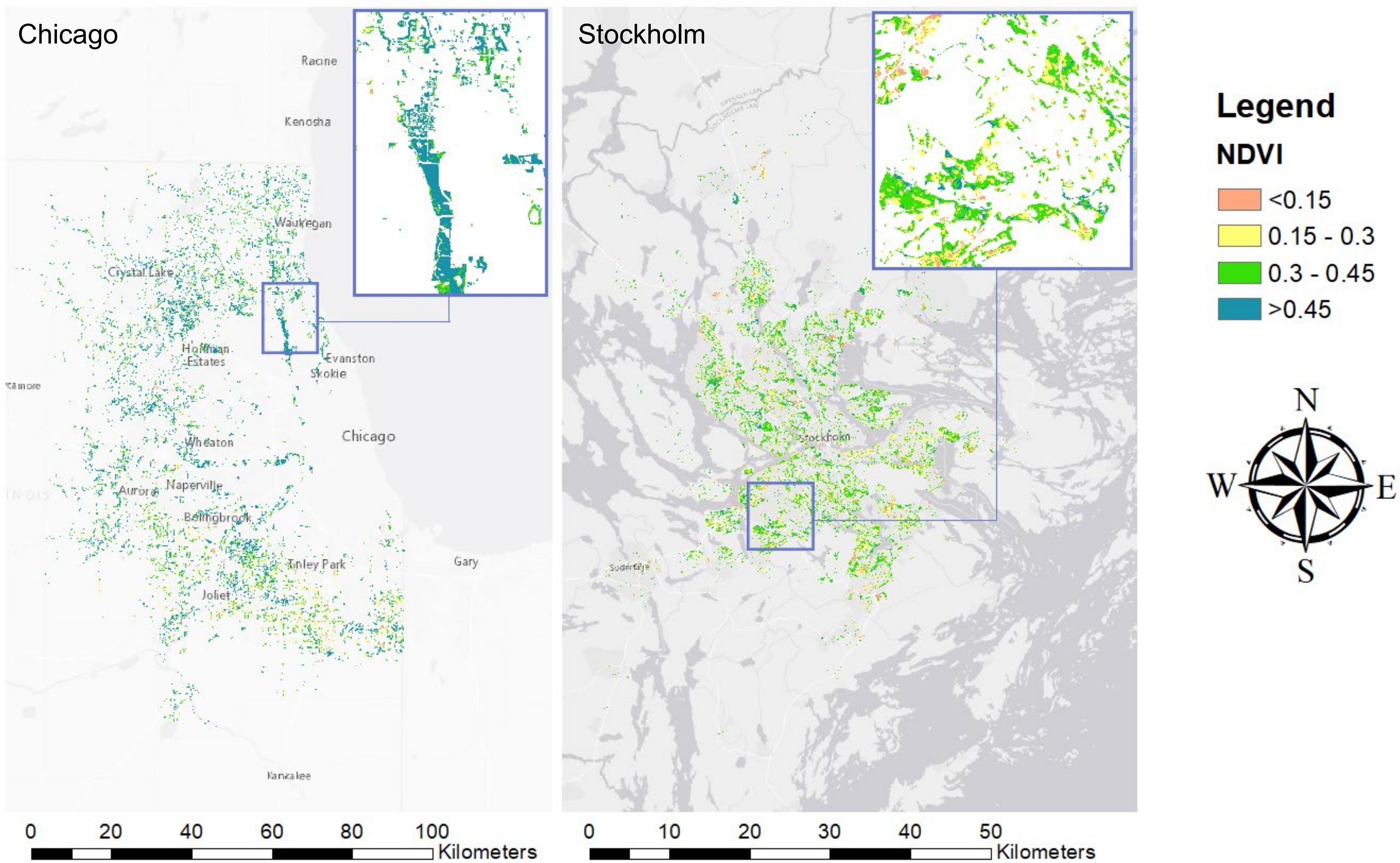


Figure 11 NDVI values within the potential developments. The scales of the maps are different because the size of the urban area in Stockholm is much smaller than in Chicago. The zoomed-in images show the striped pattern of green spaces in Chicago and scattered pattern in Stockholm.

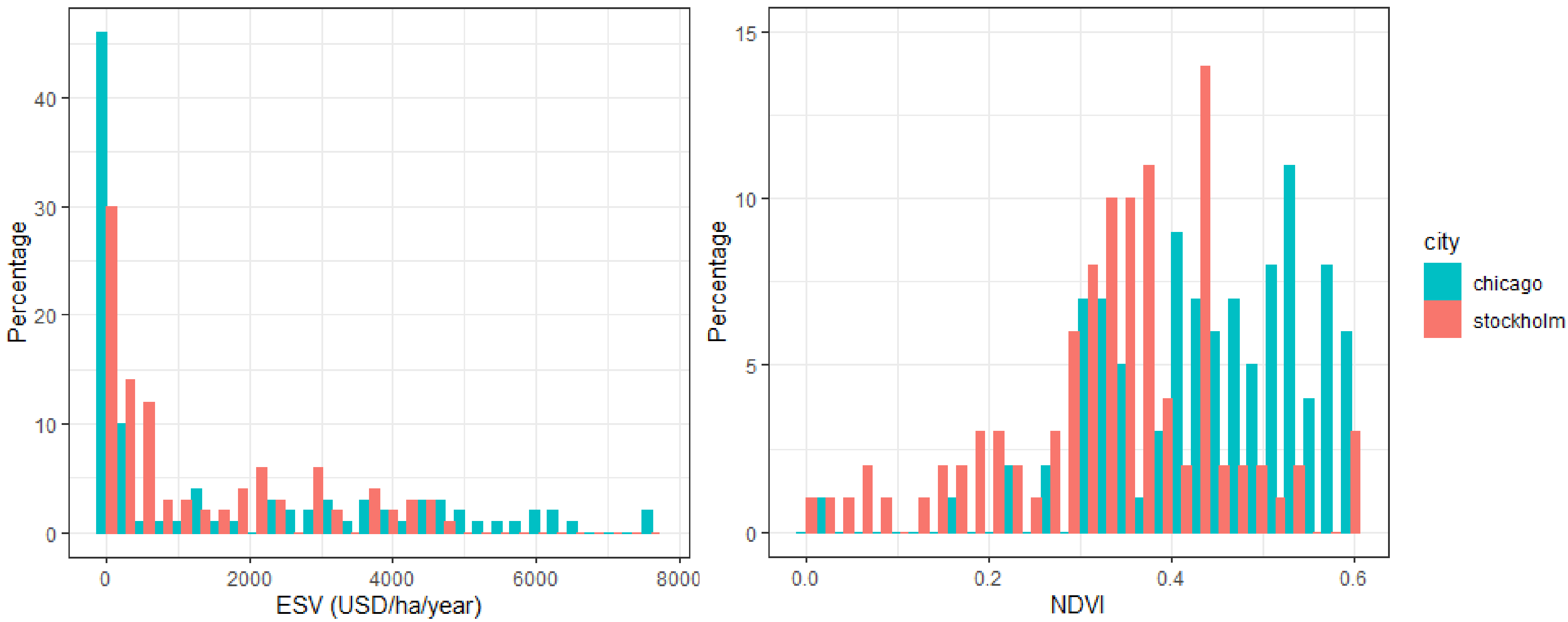


Figure 12 The distribution of numeric values, shown in percentage of values in different ranges: (a) ecosystem services value and (b) NDVI.

Image Credits: Cong Cong

The environmental impacts caused by the potential developments are evaluated by the spatial distribution of ESV (**Figure 10**) and NDVI (**Figure 11**) in the 2040 growth map. High ESV within the developments means the potential loss of important ecosystem services. High NDVI within the developments means the potential loss of dense vegetation.

Model outcomes suggest development patterns in Chicago are concentrated in two areas: (1) urban green space, forming several stripes extending towards the city center along major rivers, and (2) the fringe of the urban area. Developments on the fringe which is constituted by mainly farmland and pasture are expected to cause relatively little harms on the environment. However, forests, wetlands, and riparian areas near the rivers host valuable ecosystems and dense vegetation, exhibiting very high ESV and NDVI on the map. Developments in these areas may result in major environmental impacts. Comparing with Chicago, Stockholm exhibits a very different pattern of development. New growths happen in a scattered pattern and spread evenly over the city instead of concentrating on a specific area. Extremely high and low ESV values are rare. In other words, environmental impacts caused by future developments can be less location-specific in Stockholm.

The distribution of numeric values of ESV and NDVI is summarized in **Figure 12** and **Table 1**. Chicago shows a more dispersed distribution of ESV and NDVI with more extreme values. Areas with an ESV of higher than 5,000 USD/ha/year make up 11% of the future developments in Chicago, while the fraction is negligible in Stockholm. Similarly, areas with a NDVI value of over 0.5 constitute for 28% of the future developments in Chicago and 7% in Stockholm. This means Chicago could suffer higher loss of ecosystem benefits due to urban growth and requires carefully crafted sustainable urban management strategies.

Table 1. Summary of environmental impacts

	Chicago	Stockholm
Total Area of Development (ha)	69,886	18,792
Total Loss of ESV (USD/yr)	143,302,252	28,133,898
Average Loss of ESV per Hectare (USD/ha/yr)	2,050	1,497
Average NDVI	0.44	0.33

4.8 DISCUSSION

My evaluation of the environmental impacts has revealed that, depending on the nature of the cities, development can affect the environment in very different ways. Areas close to the city center of Chicago have been filled by existing developments, leaving little room for future growth. As a result, developments in Chicago happen in either urban green spaces that offer vital ecosystem services to the city, or in agricultural lands surrounding the city which are relatively unimportant from the aspect of ecology. Preserving the highly valuable ecosystems helps to mitigate the environmental impacts of new development in Chicago. In a previous (Pan, Zhang, et al., 2019) my colleagues and I proposed an ecological preservation district (EPD) scenario in which the growth in areas with high ESV was prohibited. The model simulation revealed the EPD scenario helped to reduce the total loss of ESV by around 40%. Nevertheless, this scenario also resulted in more development on the fringe of the city, which may contribute to sprawling. Besides, compensations need to be made for privately owned farms occupied by the development. In short, the key to a successful sustainability strategy in Chicago may be to find a balance point between the preservation of ecologically important areas and maintaining a compact city form.

In Stockholm, there are plenty of open spaces which are interlaced with existing development. This could allow sufficient space for future development. As a result, new growths are distributed evenly across the city. These areas also have lower ESV than urban green spaces in Chicago. Therefore, a certain level of development should be allowed within the city center of Stockholm to prevent sprawling. It should be noted that the current “loose” build pattern offers multiple benefits (Haase et al., 2012; Larondelle & Haase, 2013). More open spaces in the center of the city allow better ventilation. Large pervious surfaces facilitate stormwater management. Unobstructed evapotranspiration of the plants also enables better climate regulation. Overdevelopment in the area can end up in erasing these advantages. It is important to find a proper upper limit of growths within the current range of the city. The building pattern should also be carefully designed to preserve the structural integrity and connectivity of ecosystems.

Both ESV and NDVI have been selected to present the quality of ecosystems in this study. Each of the indicators has its own advantages and disadvantages. ESV in this study is derived from the land cover map and published estimations about ecosystem services provisioning from different types of ecosystems. This means the final ESV map represents the total value of different types of services (e.g. wood production, erosion prevention, and recreational opportunities from forests). Nevertheless, since it was based on the classified land cover map, it has to be assumed that each land cover class possesses a uniform ESV. This may reduce the spatial accuracy of the estimation, and can overestimate the value of green spaces in poor condition.

NDVI on the other hand, is derived from Landsat images. With a spatial resolution of 30 by 30 meters, the NDVI map can reflect different conditions of vegetation within a same type of land cover. The limitation of NDVI,however, is that the indicator is not directly related to ecosystem service provision. NDVI is determined by photosynthesis rate, and a higher NDVI value can indicate a denser canopy (Gamon et al., 1995). This may lead to an improper interpretation that areas with high NDVI values are thriving forests that should be preserved in the development plan. Cultivated crops can have even higher NDVI than trees during the growing season (Simonneaux & François, 2003). Despite the high NDVI value, crop fields usually provide very limited ecosystem services other than food production. This phenomenon was obvious in Chicago, where the spatial distribution of NDVI in Chicago does not correspond to the distribution of ESV. While the forests and parks near downtown area present relatively high NDVI values, the highest NDVI values are found in the suburban areas which are farmlands. In other words, the NDVI values failed to reflect the ecological importance correctly in Chicago. In comparison, NDVI can be a much better indicator in Stockholm, where the spatial distribution of NDVI more or less resembles the distribution of ESV. In absence of large agricultural lands, areas with high NDVI are associated with dense forests.

In sum, ecological indicators used in accessing the biophysical condition of a city can be situational and location-specific. Even if the study of a city using certain ecological indicators has been successful, the same set of ecological indicators can be ineffective, or even misleading in another city. This result agrees with the conclusion of the study by (Schwarz, 2010), that suitable ecological indicators vary among different urban forms and that sustainability strategies useful in one context may not be simply copied to another one. The difference in representative power of NDVI between Chicago and Stockholm suggests that the methodology of land assessment needs a more detailed resolution or site specificity. Nevertheless, this does not preclude finding insights and paths forward via comparisons. Comparisons, especially those using the specificity in data and approach as required dynamic urban models (such as LEAM) can be useful for determining commonalities and successful strategic interventions across contexts.

4.9 CONCLUSIONS

This study demonstrates an example implication of a PSS in comparative studies. With the help of the socio-ecological model, I analyzed the complex relationship between multiple developments and attractors. I then identified how certain environmental factors, such as water and forest, work differently in the two cities. Without a powerful quantitative tool, these differences can easily be neglected. The model also helped us to evaluate the long-term effect of the developments and finding optimized solutions to mitigate the environmental impacts. Overall, the socio-ecological model has exhibited the ability to identify similarities and differences between cities and provided information which is valuable in proposing a better sustainability strategy.

Similar to Chapter 3, this work also starts with the LEAM PSS. Compared with the relatively simple approach in the previous chapter, however, this project has brought the PSS-assisted design decision making process to a broader context, testing the performance of the data processing methodology under different conditions. The comparison of different cities and different ecological indicators has brought up one of the six features of smart landscapes, “sentient” (**Figure 8**). This means a smart data-processing framework needs to be aware of its context of application as well as the expectations of its target users. Landscapes are complex systems with location-specific characteristics. Therefore, models and methodologies developed in one project are usually not ready to be transplanted to another project in a different location and different context of applications, without modification. For example, in this study, NDVI was considered a proper indicator to represent ecosystem quality in Stockholm. Nevertheless, it was not an effective indicator in Chicago because of the presence of crop fields. The term "sentient computing" refers to a data-processing framework that retains the flexibility to be optimized under different conditions, responsive to users, and responsive to the physical world (Deal et al., 2015; Hopper, 1999). Chapter 3 and 4 demonstrate "reasonableness", that data collected from the real world can provide evidence for ecological landscape design.

One potential next step of this study is to include more ecological indicators such as habitat connectivity and conservation status. Another opportunity is to incorporate a demand and supply model to better reflect the dynamics of ecosystem services. Whether or not the supply of ecosystem services meets the needs of the residents is more important than the numeric value of ecosystem services themselves (Burkhard et al., 2012). These approaches will be demonstrated in the next chapter.

CHAPTER 5: DELIVERY OF ECOSYSTEM SERVICES FROM URBAN PARKS AND GREEN SPACES: A SUPPLY AND DEMAND ANALYSIS IN STOCKHOLM, SWEDEN

5.1 INTRODUCTION

Urban residents can benefit from ecosystem services in many different ways, including food supplies, building materials, improved water and air quality, natural climate regulation, rainwater mitigation, protection from natural disasters, recreational opportunities, aesthetic and spiritual inspirations. These ecosystem services help to improve urban life quality in a sustainable manner. Evaluating the delivery of ecosystem services, however, can be a challenging task because of the complexity of both ecosystems and the structure of urban areas. Many types of ecosystem services, especially those in the cultural service category, can only be enjoyed when the residents are visiting the park or natural area (Paracchini et al., 2014). Therefore, when evaluating how well ecosystem services are received by urban residents, accessibility to urban parks and natural areas should be an important consideration. In Stockholm, these natural areas are also defined as “green-blue areas”. Green areas refer to open lands covered by natural vegetation such as forests. Blue areas refer to areas covered by water such as wetlands (Goldenberg et al., 2018). While not designated as recreational destinations like urban parks, green-blue areas host naturally formed ecosystems and thus can provide solutions to environmental problems in cities (Gómez-Baggethun et al., 2013; Keesstra et al., 2018).

While plenty of studies have been conducted on the delivery on ecosystem services (see Lit Review Section 2.3.1), existing studies have certain limitations. First, is a challenging task to estimate the accessibility to ecosystems, especially in a complex urban fabric. Second, delivery of ecosystem services is affected by both the quality and accessibility, is the system worth accessing for example. Many existing studies explore accessibility but without consideration of ecosystem quality. Those studies that consider both factors were usually qualitative, representing the results in tiers or ranks instead of quantitative values, rendering them less useful for design implementations. Finally, one important goal to evaluate the delivery of ecosystem services is to identify residential areas that are in short supply of the services. Quantitative ecosystem service supply and demand analysis however, has usually been conducted at a low spatial resolution, only being able to identify the imbalance between supply and demand in city districts, but not in specific residential areas.

Typically, Euclidean or linear distances are used to represent urban accessibility. This approach does not accurately reflect accessibility, because it does not factor in hard to cross impediments, like rivers or railroads or major roads. These impediments can add significant distance and/or the time it takes to reach a destination. To overcome this limitation, more sophisticated approaches have been developed to estimate accessibility (G. Lin et al., 2002). These approaches can generally be classified into two primary categories: raster based and network based analysis (Mulrooney et al., 2017). A network analysis is used to disseminate the values of the various user selected attractors (where the travel starts) across the study area based on the travel times from each individual attractor to all points along the available road, biking, walking, bus, rail, and ferry networks. These networks are modeled as abstract objects consisting only of nodes and connections between nodes (edges) (Hagberg et al., 2008). In comparison, a raster based analysis of accessibility disseminates the values of the various user selected attractors across the study area on a cell-by-cell basis. For a cell with a known attraction value, new values are calculated for each of the eight neighboring cells according to the cost of traversing each cell laterally for cells that share a side or diagonally for cells at the corners. The cost depends on the difficulty of traversing the land cover at that location. For example, it is easier for a cyclist to travel on a cell representing residential areas comparing with a cell representing farmlands. The traveler is assumed to take a route towards the attractor that has the least travel cost.

A straight network analysis is generally accurate when evaluating travel within a transportation network, raster based analysis offers more flexibly. A major restriction of network analysis is that travels outside the network (e.g. walking from a parking lot to the entrance of a building through a lawn) are very difficult to model. When the analysis needs to move beyond the available networks, raster based analysis is the preferred option. A hybrid raster/network approach uses the raster based cost method until a network is encountered, where travel speeds are then used to complete the 'least cost 'destination analysis. This hybrid raster/network approach is the approach taken for this study.

This study aims to present a new methodology that evaluates the delivery of ecosystem services via different modes of travel, including walking, biking, driving, and public transit. Since walking and biking are not usually restricted to the road network, a raster based travel cost model is applied to simulate travels both within and outside the road networks. The supply of ecosystem services is assessed by attraction maps, in which providers of ecosystem services including urban parks and green-blue areas, are considered as attractors. The attraction starts from the attractors and decays when spreading through the raster. The rate of the decay is determined by the impedance (travel cost). For each of the cells in the attraction map, its attraction value is determined by both the travel cost to nearby attractors and the weight of the attractors (ecosystem services quality). The advantage of this approach is that both factors affecting the delivery of ecosystem services (quality and accessibility) can be evaluated on a same map using one single attraction value. An example output of the raster based accessibility model is shown in **Figure 13**.

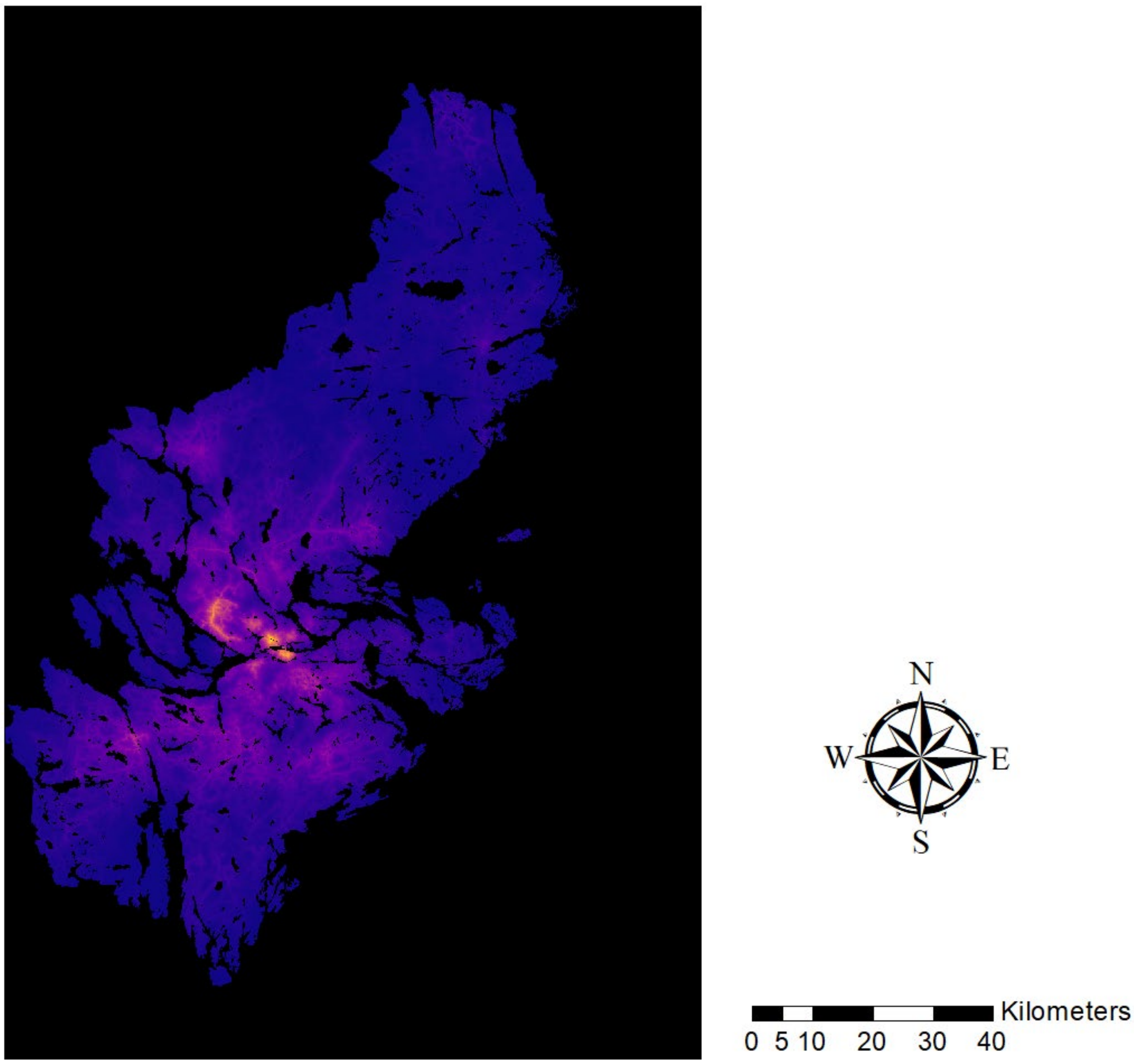


Figure 13 An example attraction map from the raster based travel cost model

5.2 RESEARCH OBJECTIVES

The goal of this study is to demonstrate a methodology of analyzing the delivery of ecosystem services across Stockholm County, Sweden. The study proceeds in the following steps: 1) represent the quality of ecosystems in urban parks and green/blue areas based on different ecological indicators; 2) estimate the delivery of ecosystem services within the urban fabric; 3) compare the supply and demand of ecosystem services, 4) give suggestions on the future development of green spaces based on the supply and demand situation, and 5) demonstrate the potential of accessibility models to be applied in the evaluation of urban ecosystems.

5.3 STUDY AREA AND DATA SOURCES

This study focuses on Stockholm County (Stockholms län), Sweden. The county hosts the city of Stockholm, the capital of Sweden, as well as many other smaller settlements. It covers 6,500 km^2 of land areas, with a total population of around 2,269,000 in 2016 (Statistics Sweden, http://www.scb.se). The county has a long, fragmented coastline on its eastern side, bordering the Baltic Sea, with numerous small islands detached from the continent (Goldenberg et al., 2018). A significant portion of the land area is constituted by green-blue areas, which is considered to be above average compared with other European cities (Fuller & Gaston, 2009; Kabisch et al., 2016). Abundant green infrastructures can be found within or close to the densely populated urban areas (**Figure 14**), although significant fragmentation caused by new developments has been confirmed (Colding, 2013).

The land cover map, road network, and shapefiles of urban parks were acquired from the Tillväxtoch regionplaneförvaltningen (TRF), the Regional Development and Planning department at the Stockholm County Council. The protected areas and green-blue areas were acquired from the Naturvårdsverket (Environmental Protection Agency, SE). The Landsat 8 image used to calculate NDVI values were retrieved from the United States Geological Survey (USGS) Earth Explorer. The quietness suitability index (QSI) map and the forest coverage map were acquired from (European Environment Agency, 2009, 2020). The population density map was acquired from (WorldPop, 2018).

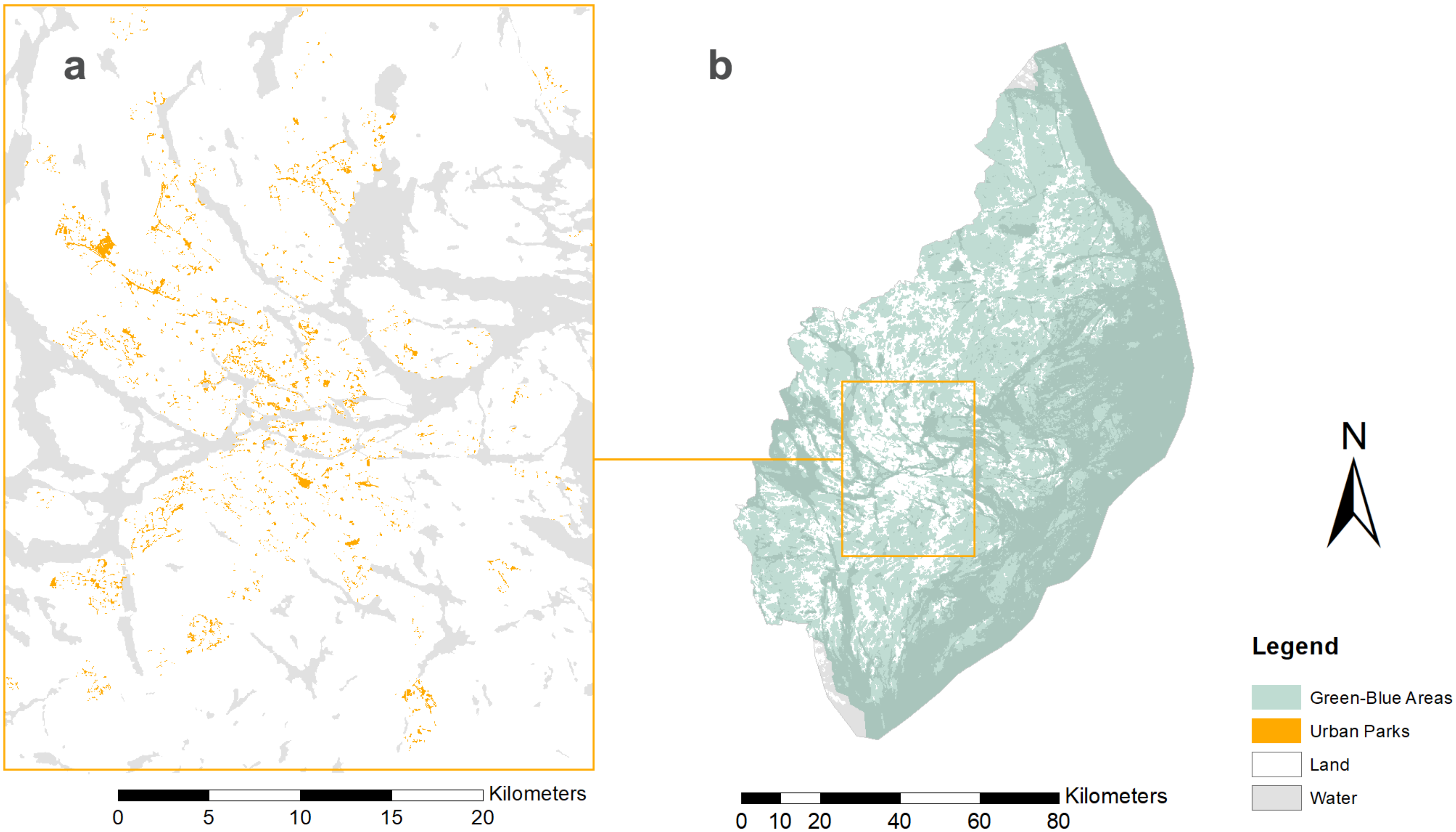


Figure 14 (a) urban parks and (b) green-blue areas in the Stockholm County

Table 2. Key variables and data sources

Variables	**Data**	**Data Sources**
Land ESV	Land cover map	TRF
Presence of Water	Land cover map	TRF
Habitat Integrity	Land cover map	TRF
NDVI	Landsat-8 Imagery	USGS
Forest Coverage	Forest Density Map	EEA
Quietness	Quietness Map	EEA
Protected Areas	Protected Areas Shapefile	Naturvårdsverket
Attractors: Urban Parks	Urban Park Shapefile	TRF
Attractors: Green-Blue Areas	Green-Blue Areas Shapefile	Naturvårdsverket
Transportation	Road Network	TRF
Population	Population Density Map	WorldPop

Abbreviations. TRF: Tillväxtoch regionplaneförvaltningen; USGS: U.S. Geological Survey; NLCD: National Land Cover Database; EEA: European Environment Agency.

5.4 INDICATORS OF ECOSYSTEM SERVICES

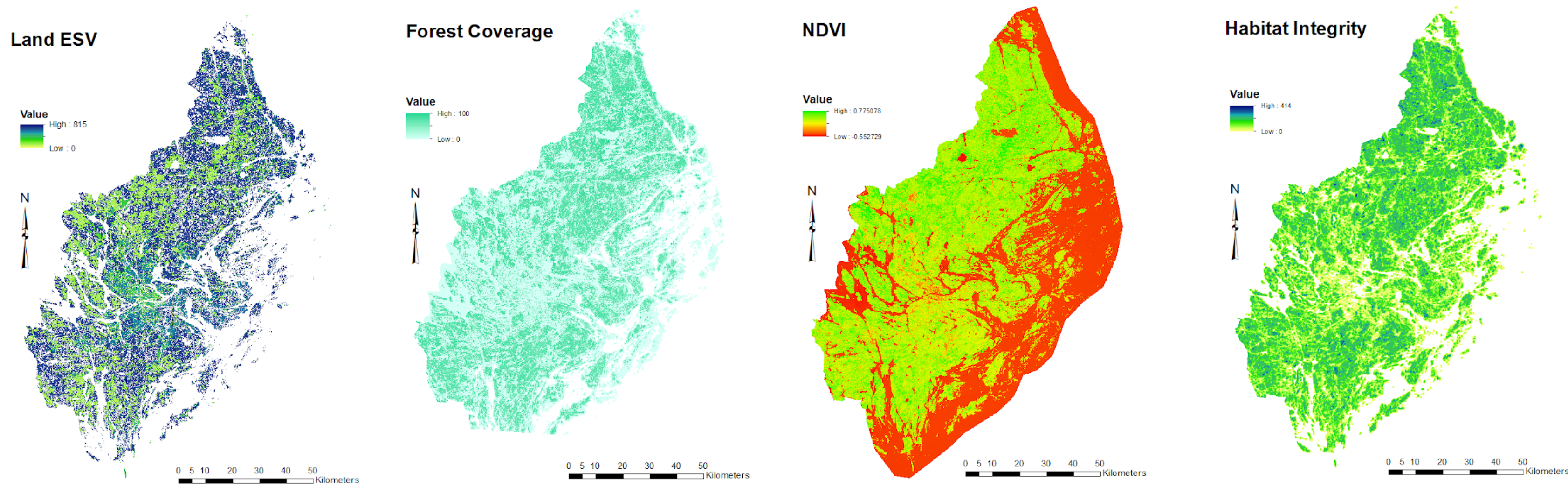
The first step of this study is to quantify the ecosystem services provided by urban parks and green-blue areas. A total of 7 ecological indicators were used to map ecosystem services over the Stockholm County: ecosystem services values (ESV), normalized difference vegetation index (NDVI), forest coverage, habitat integrity, protected areas, water, and quietness (**Figure**

15). These indicators are either directly associated with ecosystem services, or can provide insights to ecosystem services quality.

Ecosystem Services Value (ESV). The ESV is the monetary value of ecosystem services. It reflects the economic importance of the ecosystem and (see Section 4.4 for a more detailed introduction). In this study, the ESV of each type of land cover is retrieved from a searchable database by (Van der Ploeg et al., 2010). The values are then combined with the land cover map to create the ESV map.

Normalized Difference Vegetation Index (NDVI). The NDVI is an indicator derived by comparing the intensity of infrared radiation with red light (Pettorelli et al., 2005). Since infrared lights are emitted during the photosynthesis process, a higher NDVI indicates more active photosynthesis, and thus denser and healthier vegetation. High NDVI values are not always associated with ecologically important natural areas, for growing crops are highly efficient in photosynthesis and create intense infrared emission. The Stockholm County, however, hosts very few agricultural lands thus NDVI makes a suitable indicator of lush vegetation in parks and green areas (See Section 4.7). In this study, the NDVI value is calculated from Band 5 (near infrared or NIR, 851-879nm) and Band 4 (Red, 636-673nm) of the Landsat-8 images using the following formula:

$$NDVI = \frac{NIR - Red}{NIR + Red}$$



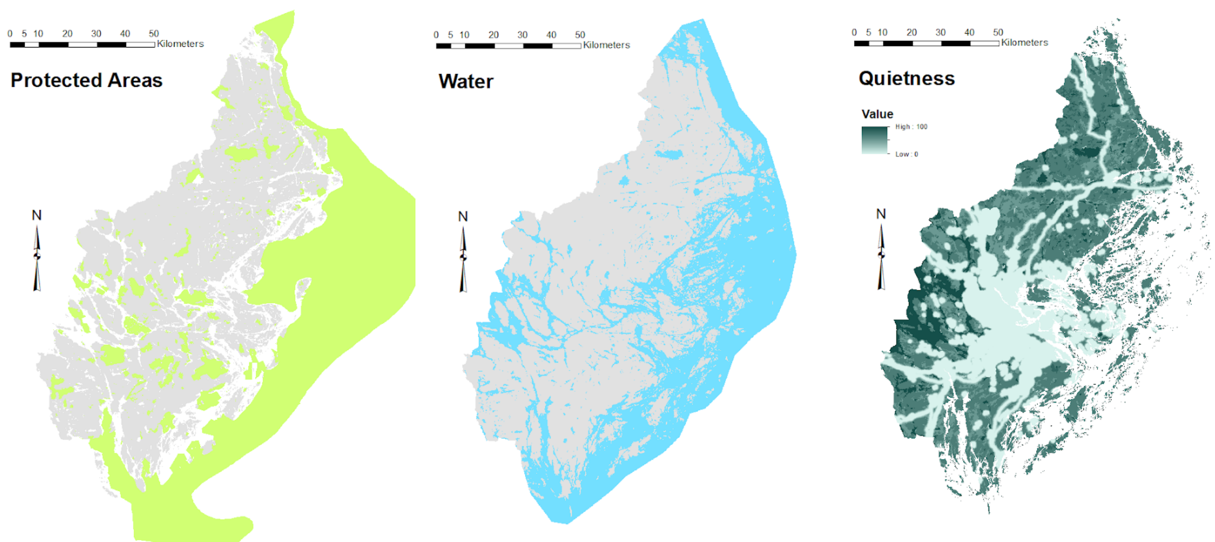


Figure 15 Ecological indicators used to evaluate the quality of ecosystem services provided by urban parks and green-blue spaces.

Forest Coverage. Forest is one of the most important provider of ecosystem services among terrestrial ecosystem types. Benefits humans can gain from forests include food and timber production, air purification, climate regulation, runoff control, soil formation and stabilization, as well as recreation and aesthetic opportunities (Krieger, 2001). In this study, the presence of forests is evaluated by the EEA Forest Map of Europe (European Environment Agency, 2009). Forest density is presented by the percentage of land surface that is covered by tree canopies.

Habitat Integrity. Anthropogenic habitat fragmentation is one of the most prominent threat to natural ecosystems. Fragmentation has negative effects on habitats by reducing the size of patches, increasing the isolation between patches, increasing the proportion of edge areas, and reducing the area of core habitat (Rogan & Lacher, 2018). In this study, habitat integrity is estimated using FRAGSTATS (McGarigal, 1995). The data source is the NLCD land cover map, with all forests assumed to be patches and all other land cover types assumed to be the matrix. The radius of gyration (gyrate) calculated by a 1km moving window is used to represent habitat integrity. The radius of gyration is an index to measure area and edge metrics that can be calculated using the following formula (McGarigal, 1995):

$$gyrate = \sum_{r=1}^{z} \frac{h_{ijr}}{z}$$

In which h_{ijr} is the distance from each cell to the centroid of the patch and z is the number of cells. A higher gyrate indicates better integrity, that the patches are larger, more compact, and less fragmented.

Protected Areas: A total of 14.3% of Sweden's total land and inland water areas are under protection, among of which 97% are permanently protected. The protected areas include national parks, nature reserves, nature conservation areas, habitat protection areas, the National City Park, and Natura 2000 (Statistics Sweden & Swedish Environmental Protection Agency, 2020). While the protection status is not directly associated with the ecosystem services supply, areas under protection are less susceptible to disturbances and damages, and thus are more likely to remain sustainable sources of ecosystem services in the future. In this study, protected areas in Stockholm County are presented by a protected areas map provided by the Environmental Protection Agency in Sweden (Naturvårdsverket).

Water. Water contributes to terrestrial ecosystem services by maintaining moist environments where wetlands can survive and burgeon. Wetlands in turn generate ecosystem services such as water purification and erosion protection (Narayan et al., 2017). Moreover, the presence of surface water itself provides recreational opportunities (Edwards et al., 2013). In this study, the coverage of surface water is derived from the TRF land cover map.

Quietness. Noises cause negative impacts on the health of both humans (Jariwala et al., 2017) and wildlife such as birds (Parris & Schneider, 2009), terrestrial mammals (Kight & Swaddle, 2011), or marine mammals (Shannon et al., 2016). Reducing noise pollution has been considered as an urgent priority to protect wildlife (Francis & Barber, 2013), which act as essential elements of ecosystems. On the other hand, vegetation in ecosystems, especially forests, contribute to significant reduction of noise pollution (C.-F. Fang & Ling, 2003). Therefore, quietness (absence of noises) can be interpreted as both a feature that promotes ecosystem health, and a service provided by the ecosystems. In this study, quietness across the Stockholm County is represented by the quietness suitability index (QSI) map from the EEA. The QSI is calculated on a raster map. For each of the cells, its QSI value is determined by its distance from noise sources (major roads, railways, airports, and industrial areas) and the noise reduction caused by the land cover between the cell and the noise sources (European Environment Agency, 2014). A higher QSI value indicates better quietness, or lower noise level.

5.5 DETERMINING ATTRACTOR POINTS

In the attraction model, ecosystem services provided by urban parks and green-blue areas are represented by a series of attractor points. In this study, each attractor point has two features: a spatial location indicating the place where the ecosystem services are supplied from, and an attraction weight indicating the quality of ecosystem services delivered. The location of attractor points is determined by overlaying the transportation network with the urban park and green-blue area maps. An attractor point is generated where a road intersects with the border of an urban park or green-blue area polygon, creating an “entry point” towards the source of ecosystem services. A total of 4,004 attractor points are generated from the urban parks, and a total of 21,740 attractor points are generated from the green-blue areas. The attraction weight of an attractor point is determined by the quality of nearby ecosystem services, represented by 7 indicators described above. There are two sets of attractor points, one is the entry to urban parks, and the other is the entry to green/blue areas. For each entry point, its weight is determined by calculating the zonal statistics (sum or average) based on the value of the indicators within the surrounding area. For urban parks, the zonal statistics were calculated within the area of the park. For green/blue areas, the zonal statistics were calculated within a 1 km radius of the entry point. How attraction weights are derived from each of the ecological indicators are summarized in **Table 3**. The weights are then normalized to a 0-100 scale. Attractor points ready for the attraction model are shown in **Figure 16**.

Table 3: Zonal statistics to determine the attraction weights

Ecological Indicator	**Urban Parks**	**Green-Blue Areas**
Land ESV	Total value within the park polygon	Total value within the 1km circle
NDVI	Average value within the park polygon	Average value within the 1km circle
Forest Coverage	Average value within the park polygon	Average value within the 1km circle
Habitat Integrity	Average value within the 1km circle	Average value within the 1km circle
Protected Areas	Total area within the park polygon	Total area within the 1km circle
Presence of Water	Total area within the park polygon and 1km radius from the park	Total area within the 1km circle
Quietness	Average value within the park polygon	Average value within the 1km circle

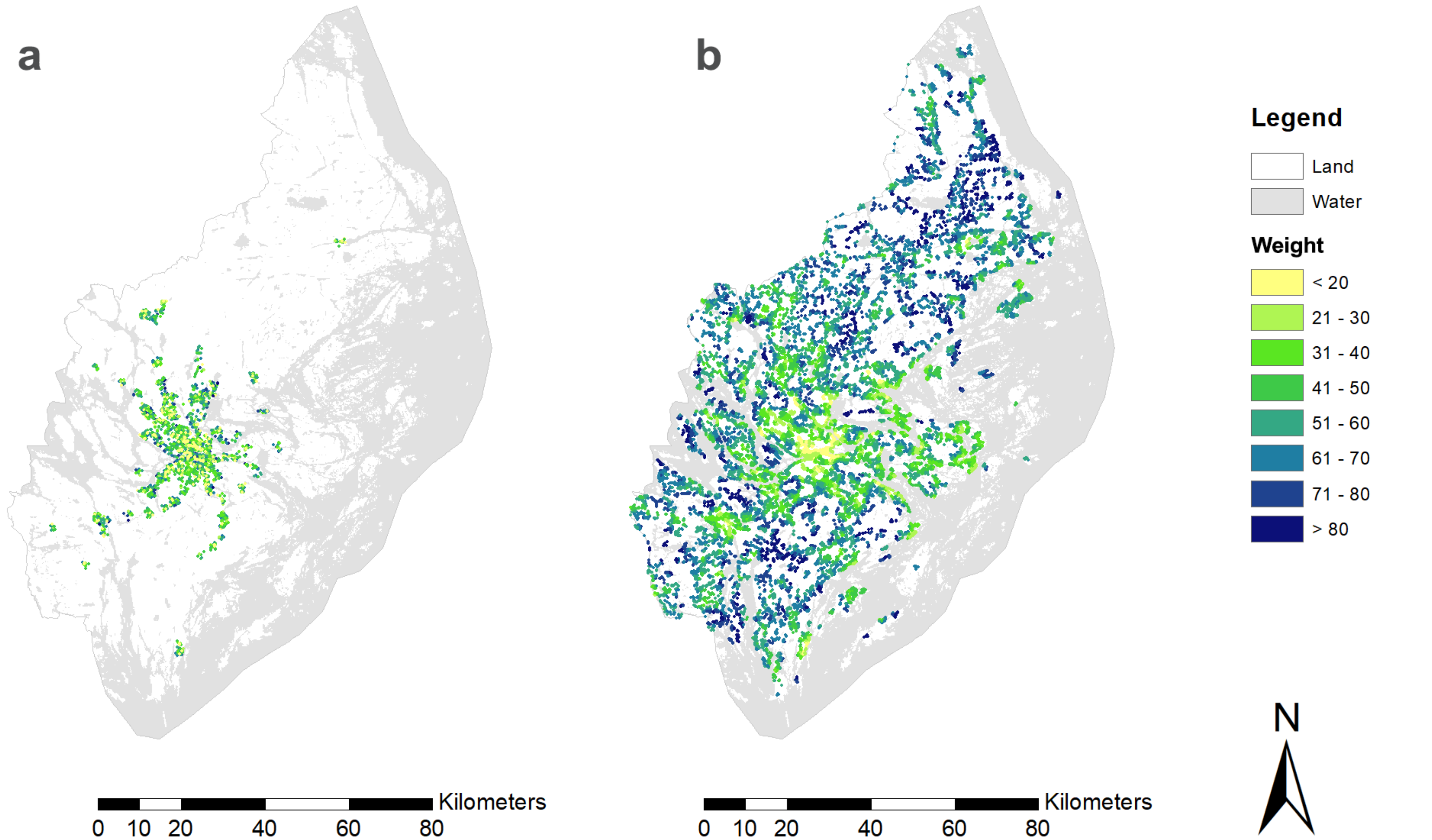


Figure 16 Attractor points representing the ecosystem services delivery from (a) urban parks and (b) green-blue areas. The weight (quality of ecosystem services) is displayed by different colors, with the bluish points having higher weights.

5.6 ATTRACTION MODELS

Attraction models offer a way to measure the impact across the study area of various user-defined locational attractors according to the accessibility of these attractors from any location in the study area. Attractors can be either point locations or areal features (polygon or raster) and usually have some sort of weighting. In this study, the attractors are entrances to urban parks and green-blue areas weighted by the quality of ecosystem services.

The attraction model starts from 4,004 attractor points representing urban parks and 21,740 attractor points representing green-blue areas using their weights as initial attraction values and moves out cell-by-cell in all directions. The attraction value decays when moving over a cell, based on a rate that is determined by the travel speed on the cell. It uses network speeds (depending on mode) once it hits a network. Travel modes include walking, biking, driving, and public transit (various forms of rail travel, buses, and ferries combined). The travel speed within the transportation network by different modes are shown in **Table 4**. This model also allows the user to customize the travel speed to better reflect the setting of the simulation. For example, For example, driving speed can be reduced during the rush hours. This model assumes that pedestrians and cyclists can travel outside the transportation network, the speed of which is determined by the land cover type (**Table 5**). The driving and transit travel modes also include a walking component to allow for travel from the park to a road, bus stop, train station, and other destinations outside the transportation network. This gives us separate attraction value distributions based on accessibility and on ecosystem services qualities, one for each travel mode. Attraction maps are generated separately for urban parks and green-blue areas, for the weights of the attractor points are calculated by different approaches (within the park polygon for parks, within the 1km circle for green-blue areas) and thus not comparable. The result attraction maps are shown in **Figure 18**.

Table 4: Travel speed within the transportation network

Travel Mode		Travel Speed (kilometers per hour)
Walk		5
Bike		23.5
Drive		40
Public Transit	Bus	10
	Ferry	8
	Railway	35
	Train	35
	Subway	22.5
	Light Rail	17.5
	Tram	17.5

Table 5: Travel speed on different types of land cover outside the road network

Land Cover Type	Travel Speed (meters per minute)	
	Walking/Driving/Transit	Biking
Open Water	0	0
Developed, Open Space	60	180
Developed, Low Intensity	60	180
Developed, Medium Intensity	70	180
Developed, High Intensity	70	180
Barren	20	35
Forest/Shrubland	10	20
Herbaceous	10	30
Cultivated	10	30
Wetlands	5	5

5.7 COMPARING THE SUPPLY AND DEMAND OF ECOSYSTEM SERVICES

The attraction model creates attraction maps that represent the supply of ecosystem services over the county. How well the ecosystem services are received by the residents, however, is determined by whether a high supply of ecosystem services spatially fits with a high demand (Ala-Hulkko et al., 2016). This supply-and-demand relationship is investigated by overlaying the attraction maps with a population density map (**Figure 17**). It is assumed that a higher population density indicates a higher demand of ecosystem services. The attraction maps are resampled to a spatial resolution of 600 by 600 meters. This is due to the limitation that population density map is in a lower resolution, and the resolution of the attraction maps need to be reduced so their cell grids can be aligned with the population density map. The resampling is conducted by combining 400 cells (20 by 20 square) into a larger one, with the value of the new cell is the mean of the 400 cells combined. The supply-demand relationship is evaluated by an

Ordinary Least Squares (OLS) regression, in which the population density is set as the dependent variable and all of the 8 attraction values (4 from urban parks, 4 from green-blue areas, by different travel modes) are set as explanatory variables. The purpose of the OLS is to identify the mismatches between the demand (population) and the supply (attraction values). The output of the regression is a map displaying the residuals (differences between observed values and predicted values). A residual larger than zero indicates the population versus ecosystem services supply ratio is higher than average, suggesting an undersupply in the location. A residual smaller than zero indicates the population versus ecosystem services supply ratio is lower than average, suggesting an oversupply in the location. Outliers where significant mismatches between the supply and the demand are highlighted in the map (**Figure 19**).

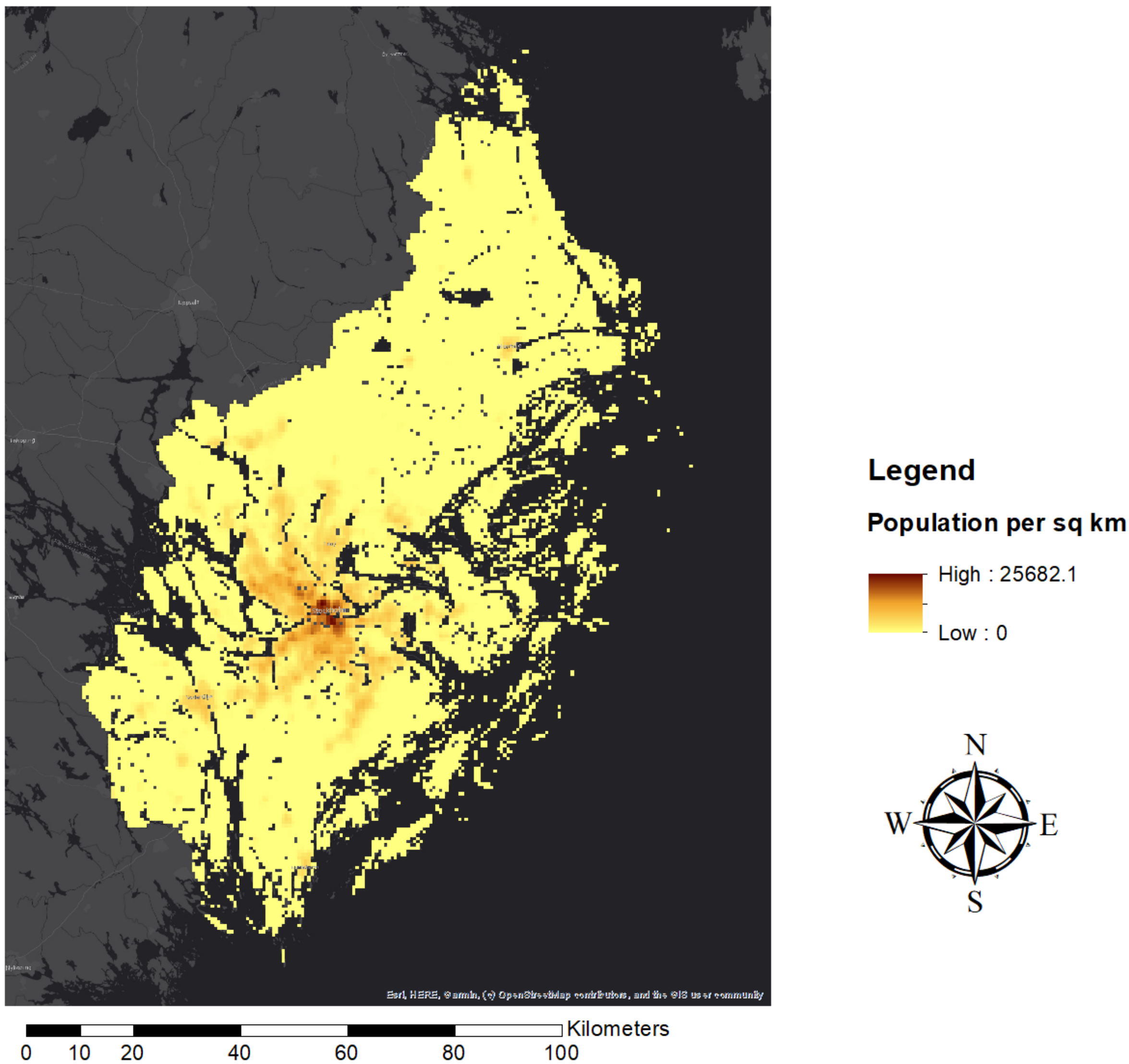


Figure 17 Population density map of the Stockholm County. Unit: people per square kilometer

5.8 RESULTS AND DISCUSSION

Result maps from the attraction model representing the supply of ecosystem services are shown in **Figure 16**. The vast majority of urban parks in the study area locate within the city of Stockholm, while a few of the parks can be find in smaller localities such as Norrtälje (northeast) and Nynäshamn (south). Attraction spreads from the locations of parks. The range of influence is

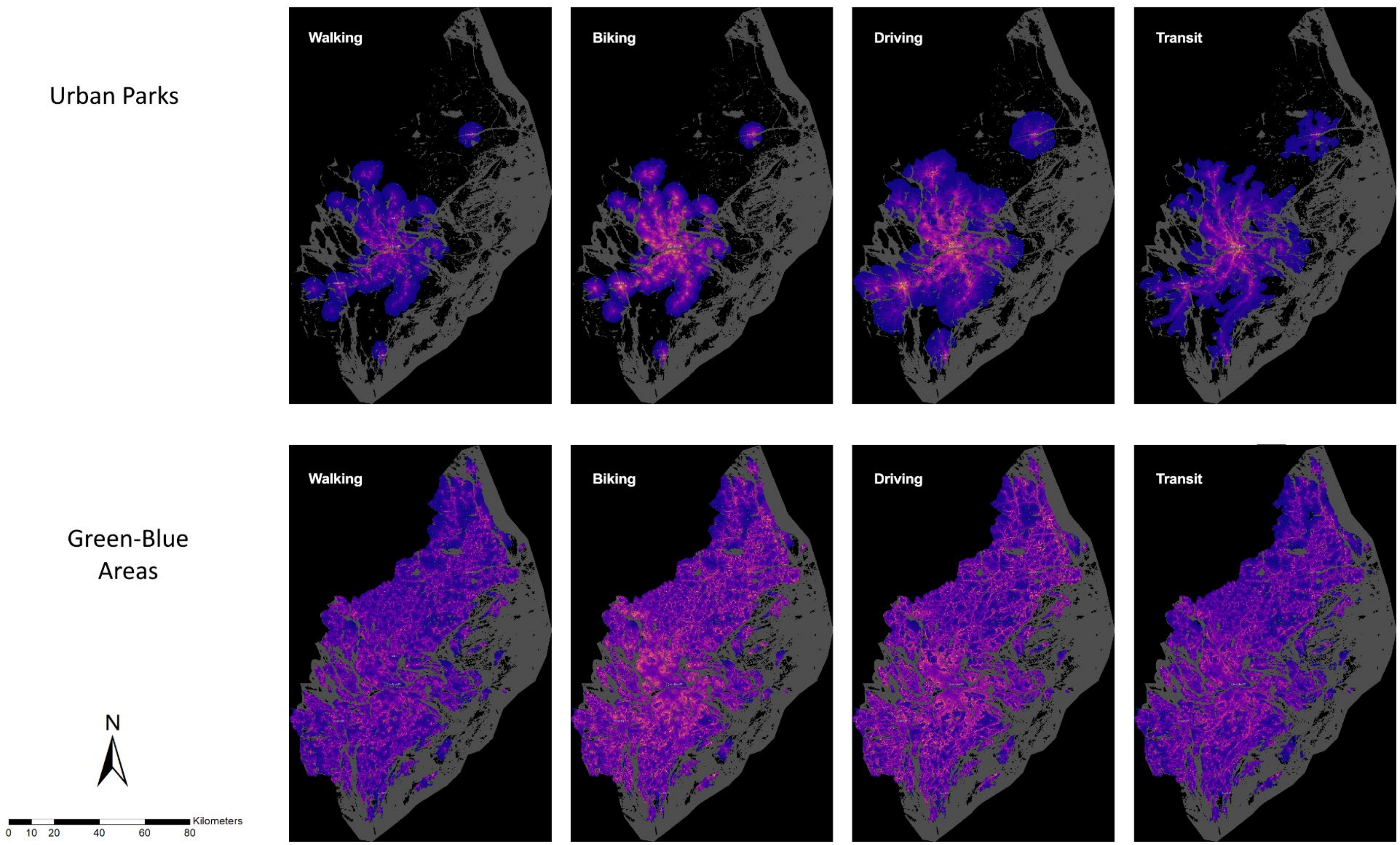


Figure 18 Attraction map output from the attraction model. The attraction values indicate the supply of ecosystem services, determined by the travel cost to attractors and the ecosystem services quality at the attractors. Blue colors indicate low attraction values, magenta colors indicate mediocre attraction values, and orange colors indicate high attraction values.

determined by the speed of travel and the structure of road network. Driving and transit attractions can extend further along major roads. Widespread green-blue areas in the county have promoted the delivery of ecosystem services, which can be accessed all across the study area. Highest attraction values from green-blue areas can be find among the outer rim of the city center, where developments are adjacent with natural areas. This agrees with our assumption that the delivery of ecosystem services is determined by both the quality of ecosystem and the cost of travel. While suburban areas host plenty of high-quality ecosystems, the sparse road network in these areas has prevented the services from being delivered efficiently. The city center, on the other hand, has a dense road network but lacks high-quality ecosystems to generate the services.

The ideal condition to deliver ecosystem services is to possess adequate amount of both high-quality ecosystems and roads at the same time, which can be find in the outer rim of city center. Nevertheless, the close distance between developments and ecosystems in these areas means the ecosystems are susceptible to disturbances and damages from human activities. In addition, roads themselves can be a major driving force of habitat fragmentation and isolation (Shiliang Liu et al., 2014). Therefore, it is recommended to take measures that help to reduce the impact of human activities on the natural areas. For example, noise barriers along the roads will mitigate the negative health effects caused by noises on both human and wildlife. Corridors and overpasses help to separate the traffic of humans and wildlife, reducing the chances of disturbances and promoting the connectivity between habitats.

Among the travel modes, walking, biking, and driving generate similar pattern of attraction, with the range of influence increasing with the travel speed. The delivery of ecosystem services by public transit is restricted by transit routes, and thus has a smaller coverage comparing with driving. Nevertheless, the services can be delivered to faraway places along the transit routes. Biking yields overall higher attraction values comparing with other modes of travel, especially in areas that are not far from settlements. This is probably caused by the ability to move outside the road network and the higher speed comparing with walking.

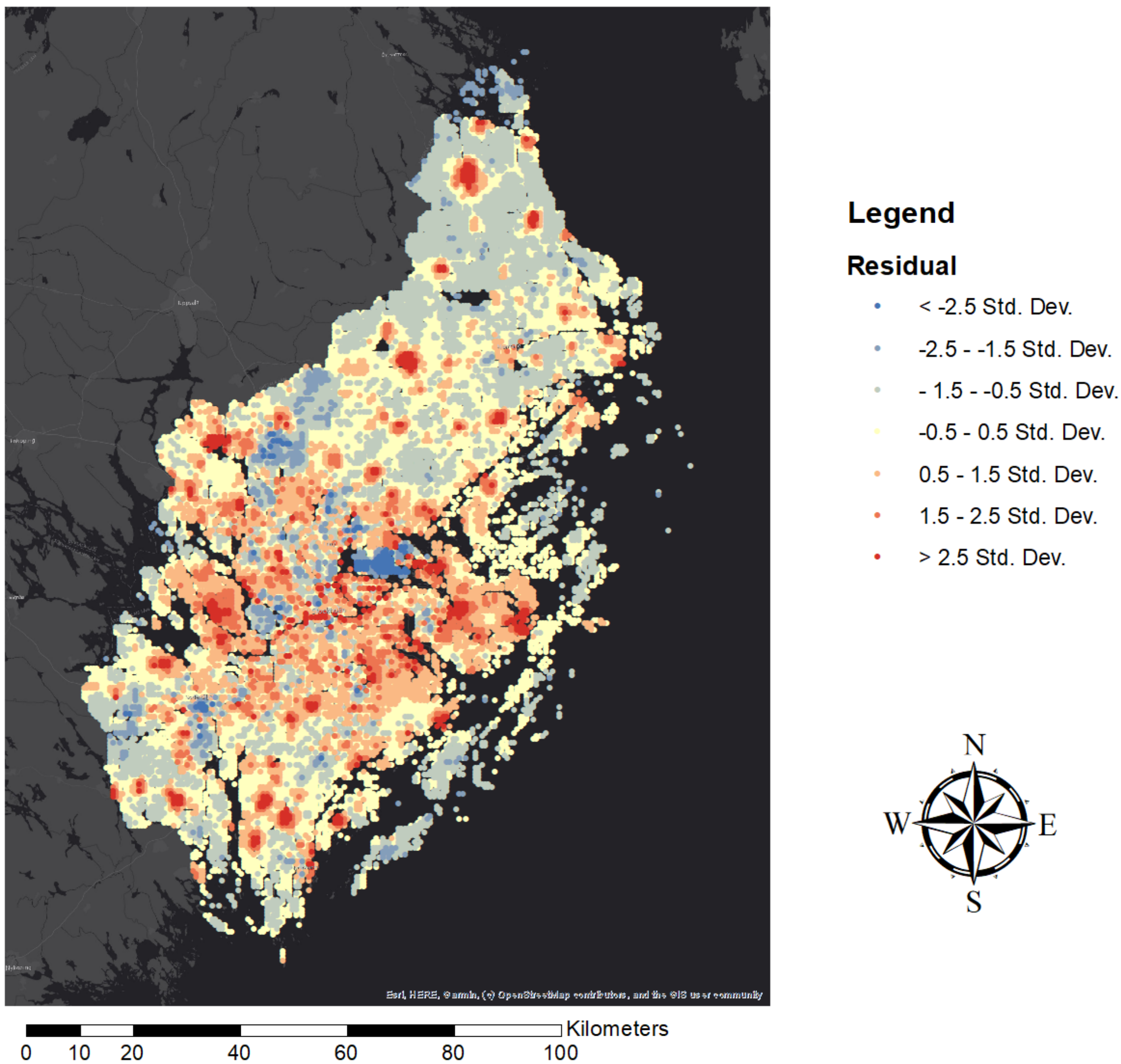


Figure 19 Residuals from the OLS regression (Unit: standard deviation). A positive residual (red) indicates an undersupply of ecosystem services at the location, when a negative residual (blue) indicates an oversupply.

Residuals from the OLS regression are shown in **Figure 19**, representing the mismatches between the supply and demand of ecosystem services. The pattern of residuals is complicated.

Drastic changes can occur among nearby neighborhoods. Not all of the densely populated areas face a shortage of ecosystem services. Some of the areas in the city center are well-supplied by nearby urban parks. The fragmented coastline has restricted the construction of road network in Stockholm, and prevents ecosystem services from being distributed evenly. An example is the blue area close to the center of the map, which represents the Bogesundslandets naturreservat. Locates on a peninsula, this natural preserve hosts high-quality ecosystems but is connected to the city with only one road, ending up in services that can not be easily accessed from the city. To facilitate the delivery of ecosystem services and promote environmental equity in the city, it is recommended to improve the connection towards major sources of ecosystem services and increase the quality of ecosystems near major roads.

While the main objective of this project is to demonstrate the attraction model and its power in assessing ecosystem services delivery, the supply and demand analysis also has the potential to provide information about environmental justice. The concept of "inequity" can be defined as the uneven distribution of resources within a population (Nyelele & Kroll, 2020). Therefore, I have assumed that each resident will need a similar amount of ecosystem services provisioning. This has led to a linear regression when evaluating the relationship between the population density and ecosystem services delivery. More recent social science studies, however, have separated the definition of equity from equality. The term 'equality' refers to sameness in benefits received, while the term of "equity" refers to the fairness in the provision regarding individual circumstances (Espinoza, 2007). As a result, the regression analysis can be considered an assessment of equality rather than equity. This is one limitation of this study, as fair distribution of resources (equity) is usually more effective in promoting social justice than even distribution (equality) (Paul, 2019). A study of equity should investigate the needs and requirements of the residents, as well as whether they are treated fairly based on their needs. A common situation of environmental inequity is the insufficient distribution of ecosystem services among disadvantaged groups (de la Barrera et al., 2019; Mullin et al., 2018; Paul, 2019). One example is that (racial or ethnic) minorities and low-income neighborhoods usually have lower vegetation coverage, despite residents in these neighborhoods may have a heavy reliance on the shading and cooling effects of tree canopies since they are unable to afford air conditioning (Nyelele & Kroll, 2020). The next step of this study will investigate more types of demographic data such as ethnic groups and income maps, locate neighborhoods with disadvantages, and evaluate the supply of ecosystem services in these areas. This will provide more valuable information about environmental equity.

5.9 CONCLUSION

This study demonstrates an attraction model and how it can help to delineate the delivery of ecosystem services in a complex urban fabric. While there have been similar studies to evaluate ecosystem services distribution, these studies have certain limitations. For example, the supply or demand of ecosystem services was represented qualitatively as tiers of ranks (Ala-Hulkko et al., 2016; F. Li et al., 2020). The spatial analysis was carried out in a rough resolution, using districts or census tracks as units (Stessens et al., 2017). The advantage of my approach is that it allows the quantification of ecosystem services and a high spatial resolution at the same time. From the perspective of the smart landscape concept, this study has brought up two additional smart system features: incisive and comprehensible (**Figure 20**). The term "incisive" indicates that a smart landscape framework should include in-depth data processing and analysis, being able to reveal hidden information from the data. In this study, fragmented data from different sources (a total of seven ecological indicators, road network, and land cover) was processed and translated into a single attraction map that represented the delivery of ecosystem services, providing information that can only be unveiled after extensive analysis. Moreover, compared with the unprocessed ecological and transportation data, the result attraction map is much easier to interpret for designers. This leads to the concept of "comprehensibleness", which means the results of the data processing should be presented and visualized in a way that is easily understandable to audiences who do not possess professional knowledge in related fields (e.g. ecology and transportation in this study).

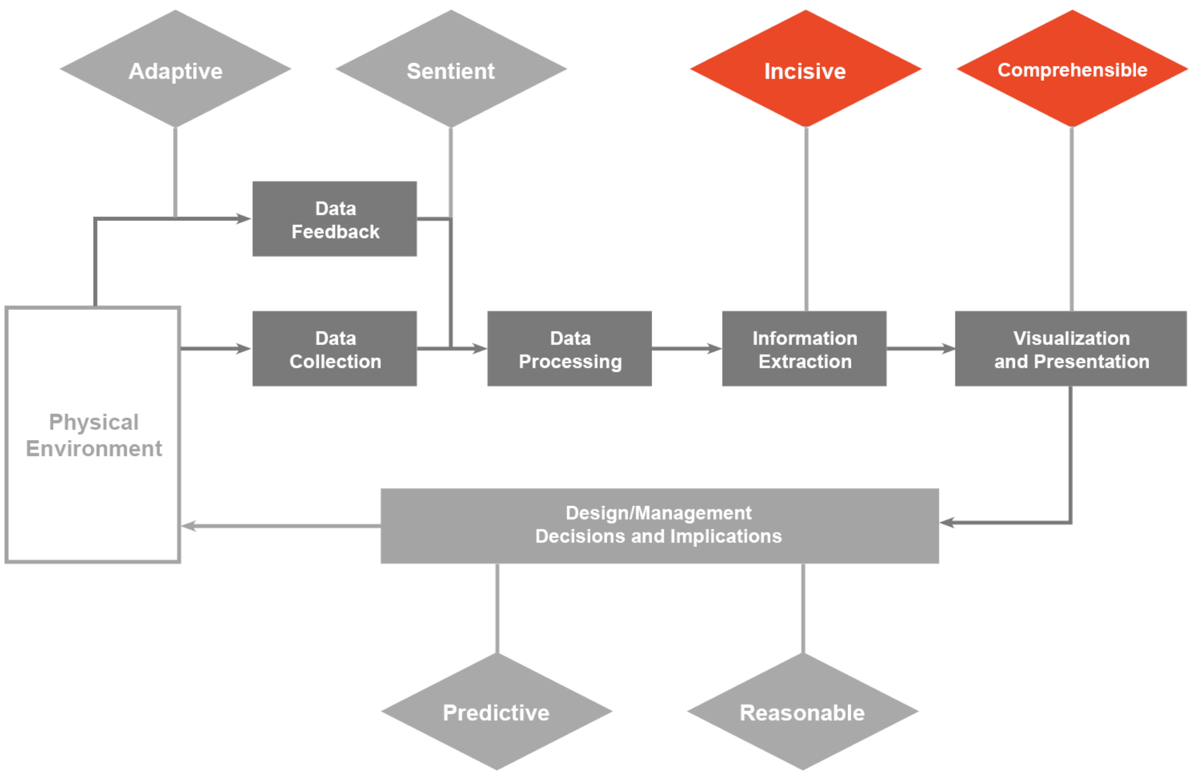


Figure 20 Contribution to the smart landscape framework, Chapter 5

CHAPTER 6: ESTIMATING FOREST CARBON SEQUESTRATION IN ILLINOIS: A FOREST STAND STRUCTURE APPROACH

6.1 INTRODUCTION

To counter the effects of climate change caused by anthropogenic greenhouse gas (GHG) emissions, more and more countries, states, and cities are proposing their own plans of emission reduction and restoration of carbon sinks. The success of these plans depends on accurate estimation of both emission and sequestration, both current values and the trends in near future (Sil et al., 2017). While carbon neutrality requires the re-establishment of the natural equilibrium of the carbon cycle, knowing exactly how far we are from the balancing point allows policymakers opportunities to establish mitigation and planning goals, i.e. how much of the emissions need to be conserved and what carbon sinks need to be preserved or restored.

Forests are by far the most effective vegetation type to sequester carbon dioxide. In published estimations, the sequestration potential of forests is at least several times higher than herbaceous land covers (Sha et al., 2022). Because of their efficiency, in many cases planting and/or restoring forested areas is a likely solution toward achieving carbon neutrality. A practical reforestation plan should provide the following information: what are the preferred species, how many trees need to be planted to meet the expected sequestration potency, when and where to plant them, and how to maintain the forest in the future. In order to answer these questions, we need accessible methods to estimate forest carbon sequestration. In this study I suggest that these methods should meet the following criteria. First, the estimation must be accurate and location specific, using as little general data as possible. Second, it should provide information of not only the carbon sequestration potential, but also the relationship between the forest structure (such as species composition and age distribution) and the sequestration potential, allowing the forests with high sequestration potential to be recreated artificially. Finally, the estimation should include a future projection to accommodate long term planning.

A common approach to estimate the sequestration potential from vegetation is to derive the primary production from remote sensing images. Roughly 50% of the dry mass from a tree is constituted by carbon (Lamlom & Savidge, 2006). Therefore, carbon sequestration potential can be in turn estimated from the primary productivity. This established method for evaluating carbon sequestration in large area has made a vital contribution in identifying the forests with the highest sequestration potential that should be preserved in future development plans. Nevertheless, these approaches have their limitations. For example, in these models sequestration potential is not directly related to the structure or condition of the forest. In other words, when we can identify effective carbon sinks, there is no guarantee that we will be able to create similar ones through forest restoration. Moreover, they provide little information to predict future trends. Therefore, despite being a widely accepted method, estimating from remote sensing images is not a preferred approach when supporting a design or development plan.

A more advanced approach to estimate forest carbon sequestration makes use of tree growth models (see Lit Review Section 2.5.1). The carbon element in a tree's biomass originates from the carbon dioxide fixed by photosynthesis. Therefore, the accumulation of biomass through growth can be easily translated into the amount of sequestered carbon. This also means that by knowing the growth rate, the biomass gained each year, of an individual tree, we are able to derive its sequestration potential during its different life stages. Usually, a tree starts from growing relatively slowly as a young sapling. When the tree becomes larger, the growth accelerates, with more biomass attained every year. The growth will gradually slow down and halt eventually when the tree reaches its maturity (**Figure 21**). This growth curve also indicates the tree's sequestration efficiency during its lifetime. Empirical relationships between a tree's size and age have been established, which varies among species. Therefore, when a tree's species and size are known, its growth rate, as well as sequestration efficiency, can be calculated. This enables the estimation of carbon sequestration potential from a forest with a known tree inventory.

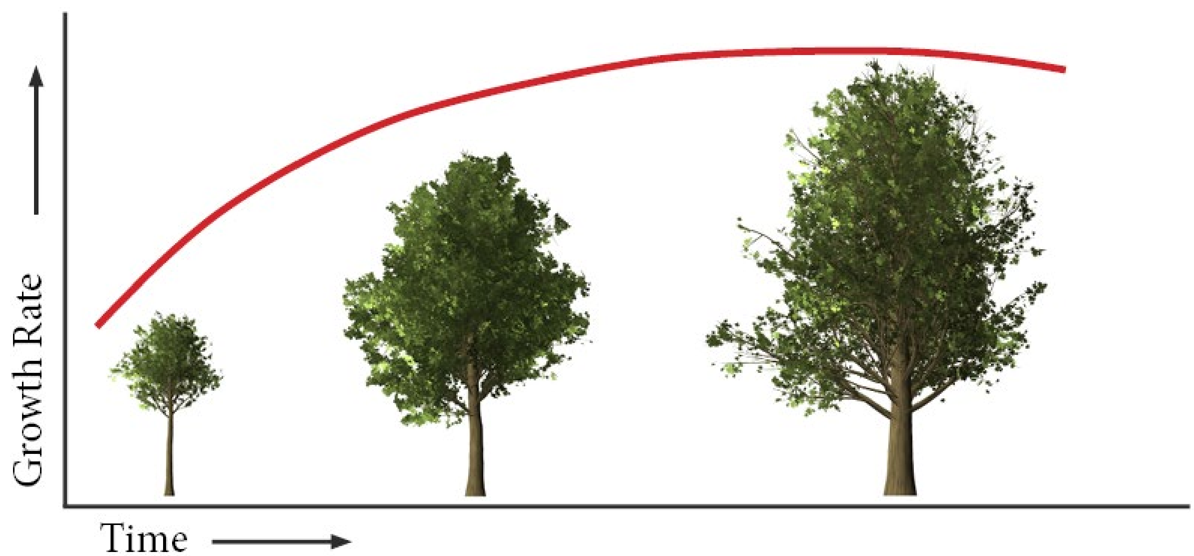


Figure 21 An example of tree growth curve, simulating Wing, K. (2019), https://forestry.ces.ncsu.edu/2018/12/cutting-at-financial-maturity-maximizing-the-economic-return-of-your-woodland/

6.2 RESEARCH OBJECTIVES

In this study I expect to fulfill the following research objectives: 1) estimate the carbon sequestration potential of a set of example forests in Illinois; 2) predict the trends of sequestration potential in the near future; and 3) propose a forest conservation strategy that is optimized to carbon sequestration. These elicit the following research questions: how the forest stand structure has changed during the study period 1997 – 2016, how it is going to change in the near future, how the changes in the forest stand structure can affect the carbon sequestration potential, and which forest conservation and restoration strategies should be applied to promote carbon sequestration.

6.3 PRELIMINARY STUDIES

I have conducted a preliminary study to estimate the carbon sequestration potential from forests in Illinois based on the basal area maps. The term "basal area" refers to the area of the trunk section measured at breast height, or 4.5 feet from the ground. I acquired the basal area maps of different tree species presented in Illinois in the year 2002 from the USFA tree species metrics dataset. These maps present the sum of basal area from trees in a specific species within 30 by 30 meters cells. While the total basal area within a cell is related to tree density, it does not translate to the number of trees directly. A higher basal area can indicate either a lot of smaller trees, or fewer but bigger trees. In order to calculate the carbon sequestration potential, however, I need to know the exact numbers of trees in different sizes. In this study, these numbers were estimated from the statistics of trees in Illinois from USFS DATIM. This dataset assorted trees of each species into different diameter groups based on their diameters at breast height (DBH) in an interval of 2 inches. Knowing the number of trees in different diameter groups, I calculate the age distribution, or the ratio of trees in different sizes. Since there was no further data about forest structure in different locations, I assume a uniform age distribution within the state.

With the age distribution, as well as the basal areas, I was able to estimate the number of trees. Assuming that the total number of trees of the species in a cell is N, and the percentage of trees in a given radius (r_i) is p_i, the total basal area of trees in that diameter group will be $p_i N \pi r_i^2$. Therefore, the total basal area of the species will be:

$$\pi N(p_1 r_1^2 + p_2 r_2^2 + p_3 r_3^2 + \ldots\ldots) = total\ basal\ area$$

The equation quantifies the total amount of trees in the cell (N) and the amount in any specific diameter group ($p_i N$). The carbon sequestration potential from an individual tree is determined by its species and size. This value was acquired from the database of i-Tree Planting (https://planting.itreetools.org/). The input was the species and DBH of the tree. The output was how much carbon can be sequestered every year by one tree of the species in the diameter group (CO_2 equivalent kg/year). The total sequestration value within a cell was calculated by multiplying the sequestration potential of one tree in a specific diameter with the number of trees in that diameter group, then adding the results from all diameter groups and species. The result of this approach is shown in **Figure 22**. The estimated statewide sequestration from forests was 16,903,518 tons CO_2/year.

This approach was considered to be more accurate (by Illinois IDNR personnel) comparing with land cover based models. Forest density in different locations is better reflected in the model. Nevertheless, it also possesses several limitations. First, the age distribution of trees was based on the state average values because of the lack of location specific data. Second, the morality and harvesting of trees were not considered, which could lead to an overestimation of the sequestration potential. Moreover, the basal area maps were created in 2002 and had not been updated since then. Finally, there had been no prediction of future trends. To provide a better insight to the situation of forests in Illinois, as well as predict the future changes of forest carbon sequestration, a forest stand analysis is conducted.

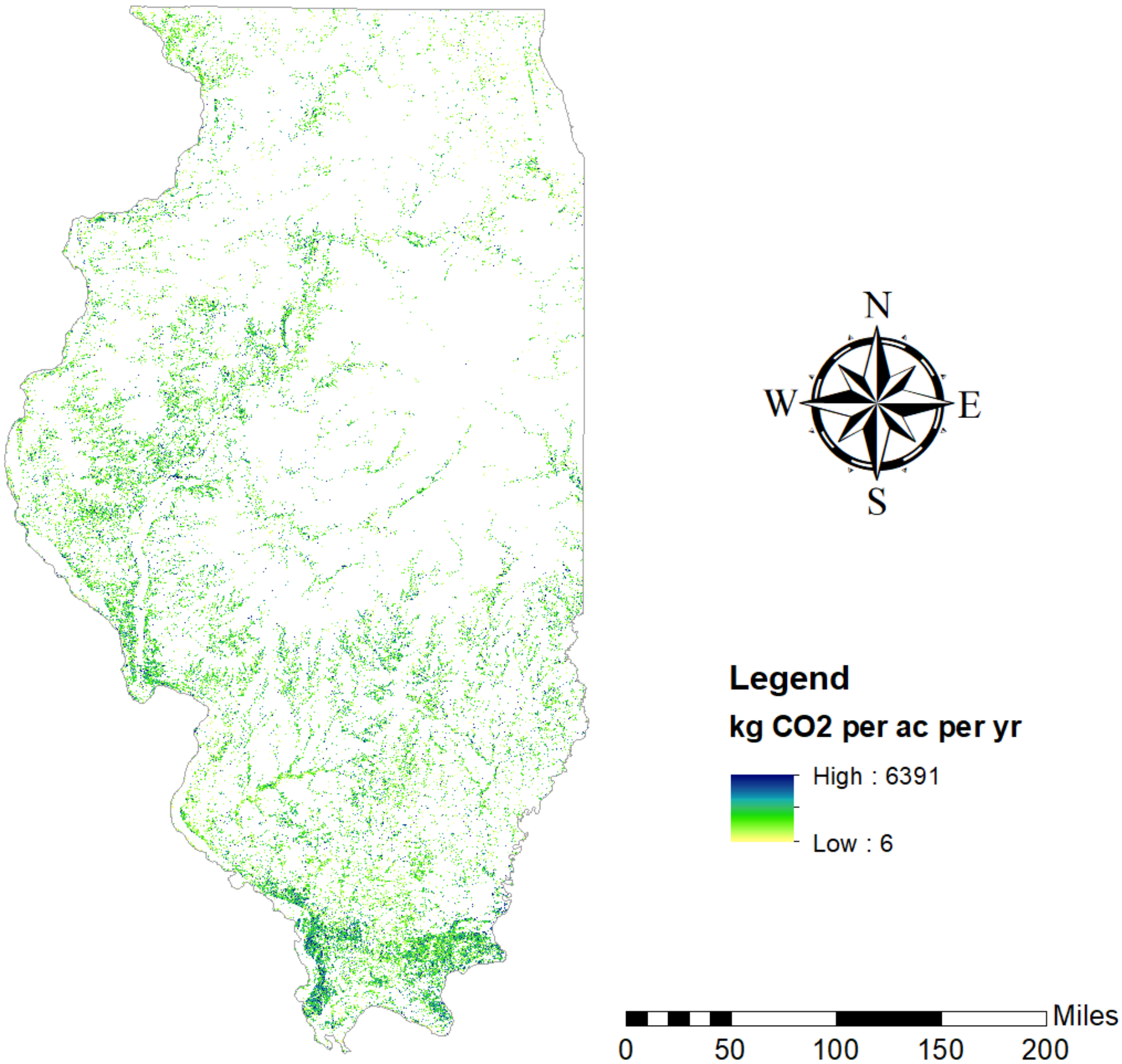


Figure 22 Forest carbon sequestration from forests in Illinois, estimated by basal areas.

6.4 DATA COLLECTION

The Critical Trends Assessment Program (CTAP) monitors the biological condition of forests, wetlands, and grasslands in Illinois. Since 1997, near 200 forest sites have been sampled by the CTAP. These sites are selected randomly within the forests in Illinois. For each of the sites, the tree plot areas are defined by three 50m by 10m rectangular transects that radiate out from the center point. For each of the trees within the plot areas, its size and species are recorded. The sampling sites are scheduled to be visited repeatedly at an interval of 5 years.

In this study, records of 190 forest sampling sites, visited within the period from 1997 to 2016, were retrieved from the CTAP archive. Since the sampling process is scheduled to repeat every 5 years, each of the sites is expected to be visited 4 times - once during each of the following time periods: 1997 through 2001, 2002 through 2006, 2007 through 2011, and 2012 through 2016. Sites that do not meet this requirement (having missing or duplicated records) were removed from the data set. Among the 190 sites, 125 were included in this study (**Figure 23**).

Similar to the preliminary study, carbon sequestration from trees is acquired from the database of i-Tree Planting (https://planting.itreetools.org/). The data is summarized in **Table 6**, each number presents the carbon sequestration potential (CO_2 sequestered in a year) from a single tree, determined by its species and size.

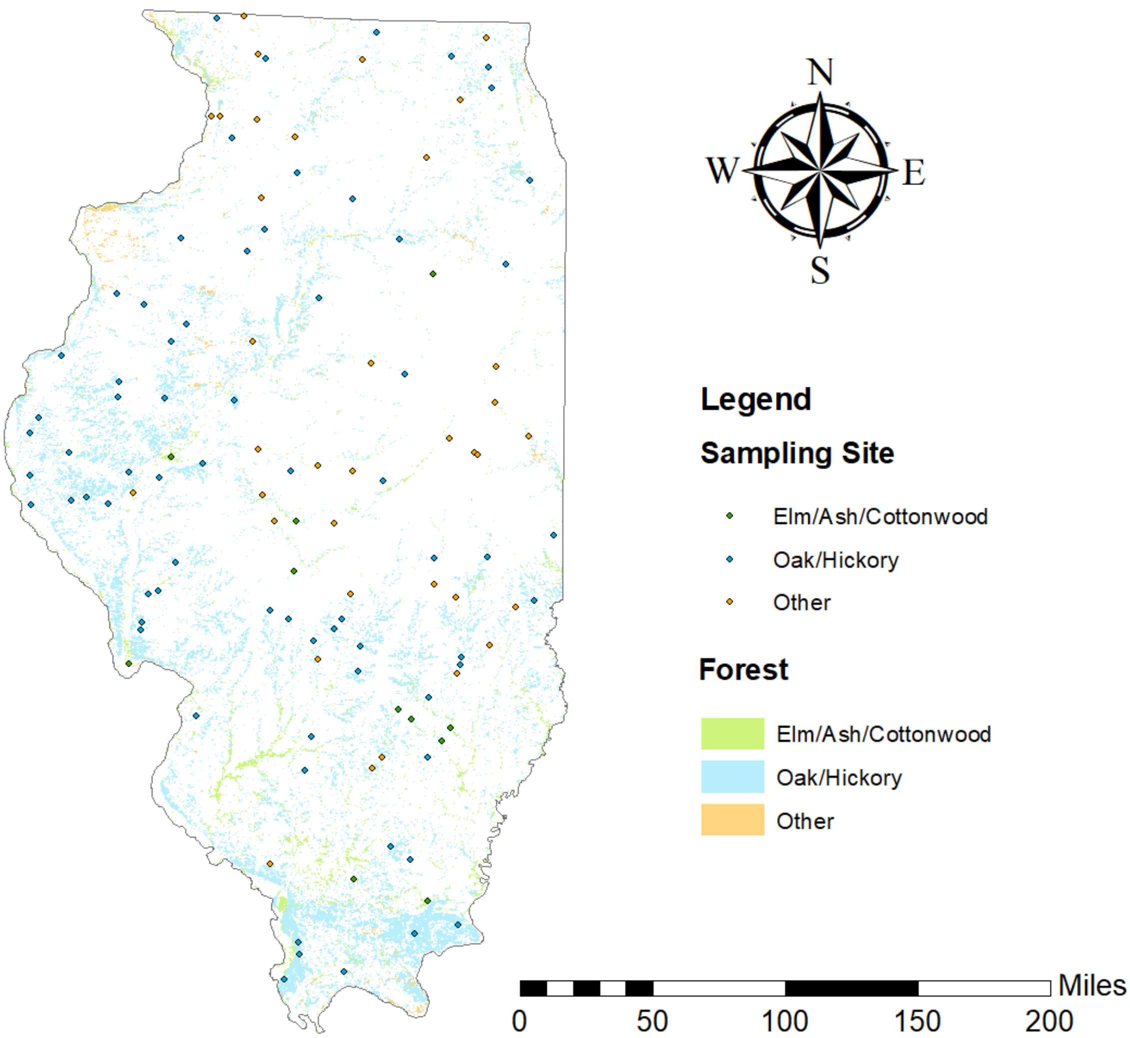


Figure 23 CTAP forest sampling sites, data collected from which has been used in this study, overlaid on top of the Illinois forest map. Two major types of forests, Oak/Hickory and Elm/Ash/Cottonwood, are highlighted with different colors. The base forest type map is acquired from (US Forest Service, 2003).

Table 6 Carbon sequestration value by species and size groups

Scientific Name	Common Name	Carbon Sequestration (kg CO_2/yr) by Size Group (DBH)								
		5-10cm	10-15cm	15-20cm	20-25cm	25-30cm	30-40cm	40-50cm	50-60cm	>60cm
Fraxinus pennsylvanica	Green Ash	12.9	21.8	31	40.5	50.3	65.2	85.6	106.4	127.7
Fraxinus americana	White Ash	11.5	22.3	34.9	48.9	64.1	88.8	124.9	164.1	206.1
Tilia american	American Basswood	6.8	13	20.3	28.2	36.9	50.9	71.3	93.3	116.9
Betula nigra	River Birch	13.7	27.4	43.8	62.6	83.4	118.2	170.4	228.4	291.4
Acer negundo	Boxelder	14.7	26.9	40.6	55.5	71.4	96.9	133.5	170.4	210.7
Aesculus glabra	Ohio Buckeye	30.3	59	93.1	132	175.2	247.4	356.1	477.4	610.2
Rhamnus sp.	Buckthorn	17.4	33.6	53	75	99.6	103.5	N/A	N/A	N/A
Juniperus virginian	Eastern Red Cedar	8.3	15.4	23.4	32.1	41	53.8	70.9	89.6	89.6
Prunus serotina	Black Cherry	15.5	30.9	49.2	70	92.8	130.5	186.7	248.6	315.7
Populus deltoides	Eastern Cottonwood	12.5	24.9	40	57.3	76.5	108.6	156.3	209.3	267.3
Cornus florida	Flowering Dogwood	12.2	22.8	34.8	48	62.2	62.6	N/A	N/A	N/A
Ulmus sp.	Other Elms	18	35.4	55.8	78.6	103.4	144.1	203.9	269.2	339.3
Ulmus americana	American Elm	11.1	21.6	33.9	47.6	62.5	87	122.9	162.1	204.2
Celtis occidentalis	Northern Hackberry	5.1	7.2	9	10.7	12.3	14.5	17.3	20	22.5
Carya sp.	Hickory	12.2	24.1	38.4	54.7	72.8	102.7	147.1	195.5	246.8
Gleditsia triacanthos	Honeylocust	9.5	17.9	27.9	39.4	52.3	73.9	106.8	143.8	184.7
Ostrya virginiana	Eastern Hophornbeam	5.8	11.6	18.6	26.5	35.1	49.4	70.4	93.6	104.7
Acer rubrum	Red Maple	16	30.6	47.8	67.2	88.6	123.8	175.8	232.9	294
Acer saccharinum	Silver Maple	19.9	33.4	47.3	61.6	76.1	98.4	128.7	159.6	191
Acer saccharum	Sugar Maple	9.2	17.7	27.5	38.5	50.4	69.7	97.4	126.8	157
Morus sp.	Other Mulberries	6.4	12.6	20	28.3	37.3	38.4	N/A	N/A	N/A
Morus alba	White Mulberry	6.7	13.3	21.1	29.9	39.5	55.2	78.5	104	116
Quercus rubra *Quercus velutina* *Quercus imbricaria* *Quercus palustris*	Red Oak Group	10.5	21	33.4	47.4	62.8	88.1	125.6	166.8	211.3
Quercus alba *Quercus macrocarpa* *Quercus stellata*	White Oak Group	7.1	14.4	23.1	33.2	44.4	63.4	92	124.1	159.2
Cercis canadensis	Eastern Redbud	6.9	13.1	20.6	29.2	38.8	40.3	N/A	N/A	N/A
Liquidambar styraciflua	Sweetgum	7.4	14.5	23.2	33.1	44.2	62.7	90.2	120.6	153.2
Platanus occidentalis	American Sycamore	7.2	14.2	22.7	32.4	43.2	61.3	88.3	117.9	149.9
Juglans nigra	Black Walnut	8.3	16.3	25.9	36.7	48.8	68.8	98.3	130.5	164.6

6.5 EVALUATING THE FOREST STAND STRUCTURE AND CARBON SEQUESTRATION POTENTIAL

Data of trees (species and size) from the 125 CTAP sampling sites is first divided into 4 groups based on the time of data collection (1997 - 2001, 2002 - 2006, 2007 - 2011, and 2012 - 2016). For each of the time periods, the structure of the forest stand is revealed by counting the number of trees within each size groups (5 - 10cm, 10 - 15cm, 15 - 20cm, 20 - 25cm, 25 - 30cm, 30 - 40cm, 40 - 50cm, 50 - 60cm, and larger than 60cm in DBH). Tree counts from each of the time period are then combined to present the changes in forest stand structure over time (**Figure 24**). This process is repeated for each species to reveal species-specific size composition.

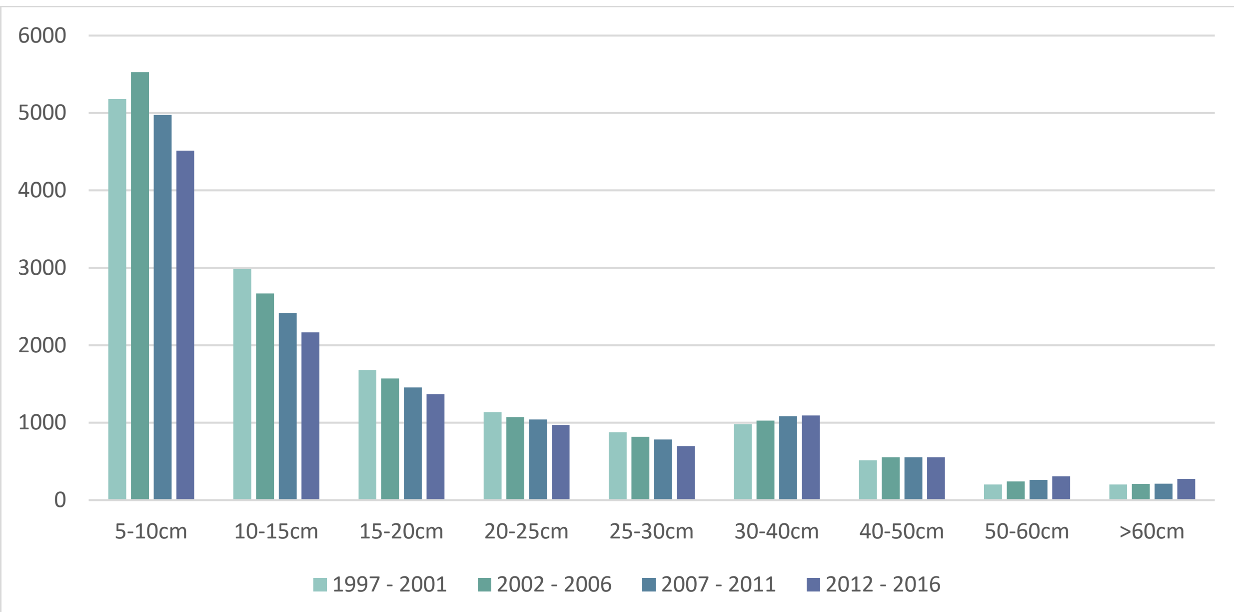


Figure 24 Forest stand structure dynamics for all species, from 1997 to 2016. Each cluster represents trees in a size group, while the columns in a cluster shows how the number of trees in the size group changes from one time period to another. Tree counts have decreased in all size groups smaller than 30cm, and increased in groups larger than 30cm.

Future dynamics of the forest stand structure are projected based on existing trends. The basic unit of the projection is trees from the same species and in the same size group. After the statistics from the previous steps, each unit has 4 data points (one from each time period) representing the amount of trees in a given species and size group. For each unit, the trend line predicting how the tree count can change in the future is derived by either exponential or linear regression. In a unit where the number of trees is declining, an exponential regression is applied, assuming a steady rate of mortality. In a unit where the number of trees is increasing, a linear regression is applied, which has better fit to the existing trends. For each of the units, 4 new data points are generated based on the trend line, representing the number of trees in time period 2017 – 2021, 2022 – 2026, 2027 – 2031, and 2032 – 2036. In sum, this study evaluates the forest stand structure dynamics over a 40 years time span, from 1997 to 2036, in a total of eight 5 year periods (4 by summarizing existing data, 4 by projection). The results of the forest stand structure analysis are summarized in **Figure 25** and **Figure 26**.

For a tree with known size and species, its efficiently in sequestering carbon can be find in **Table 6**. While all trees in a unit are from the same species and in the same size group, the sequestration potential of the unit can be calculated by multiplying the number of trees with the carbon sequestration value that is correlated with the species and size. The results of the carbon sequestration estimation are summarized in **Figure 27** and **Figure 28**.

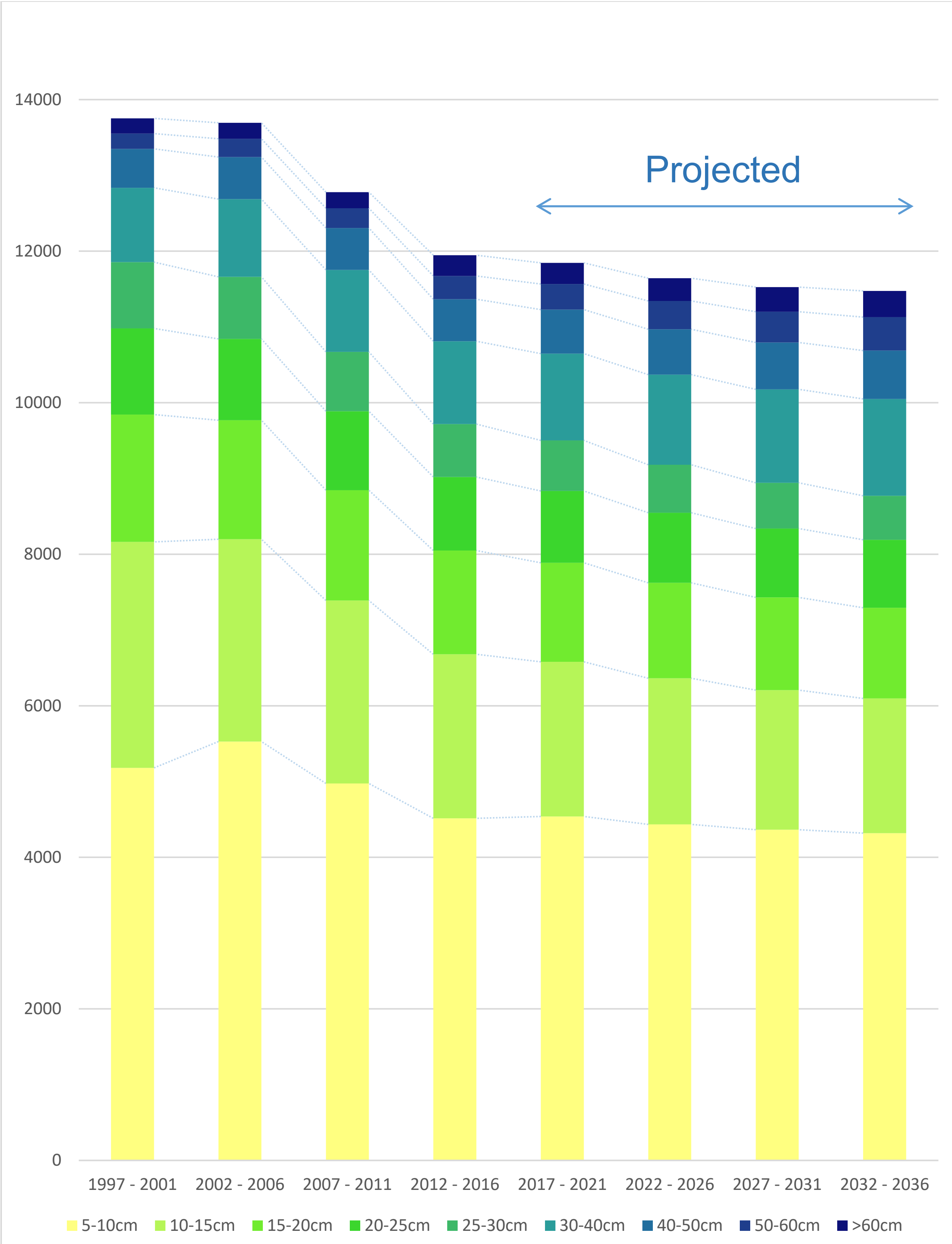


Figure 25 Tree counts by size groups

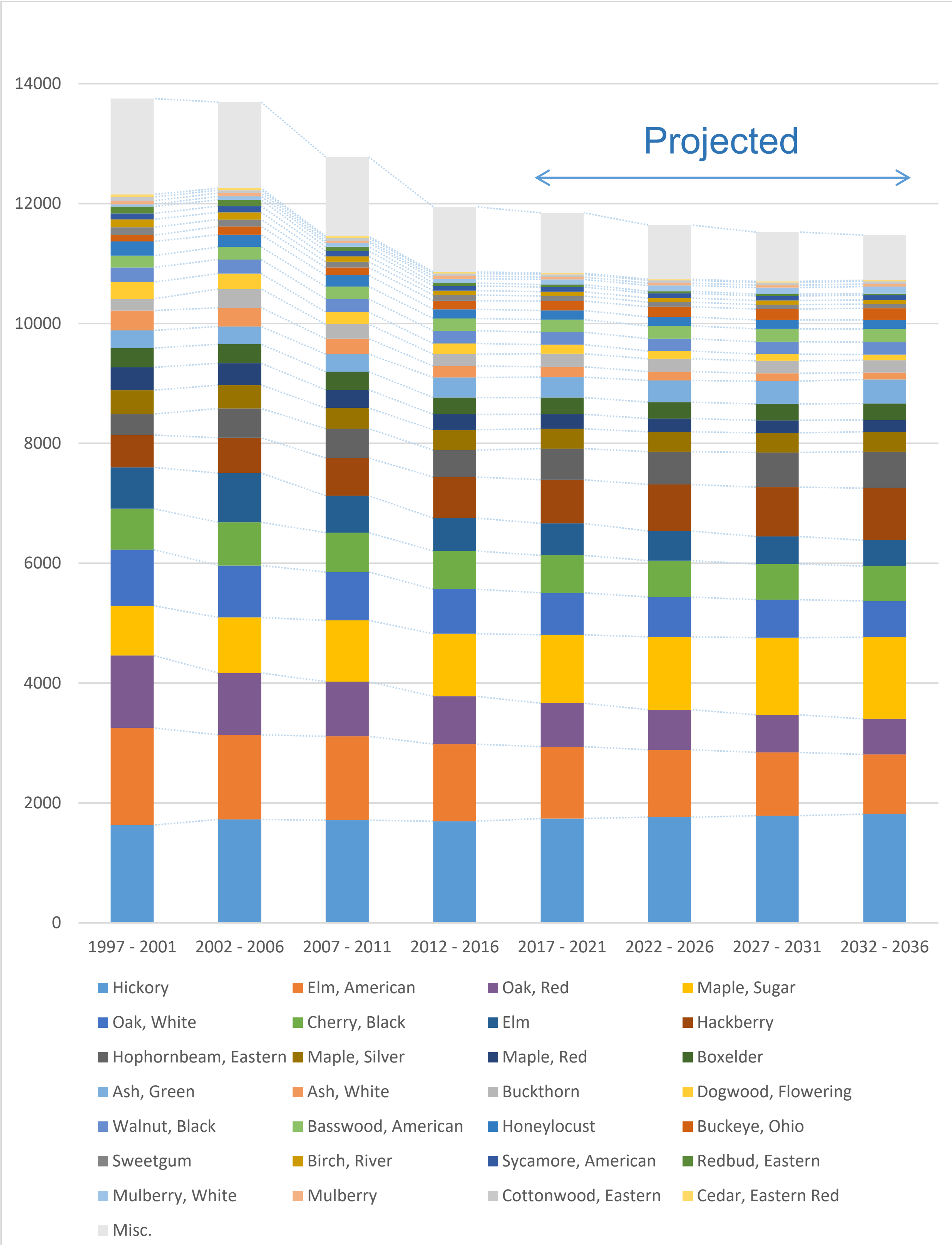


Figure 26 Tree counts by species

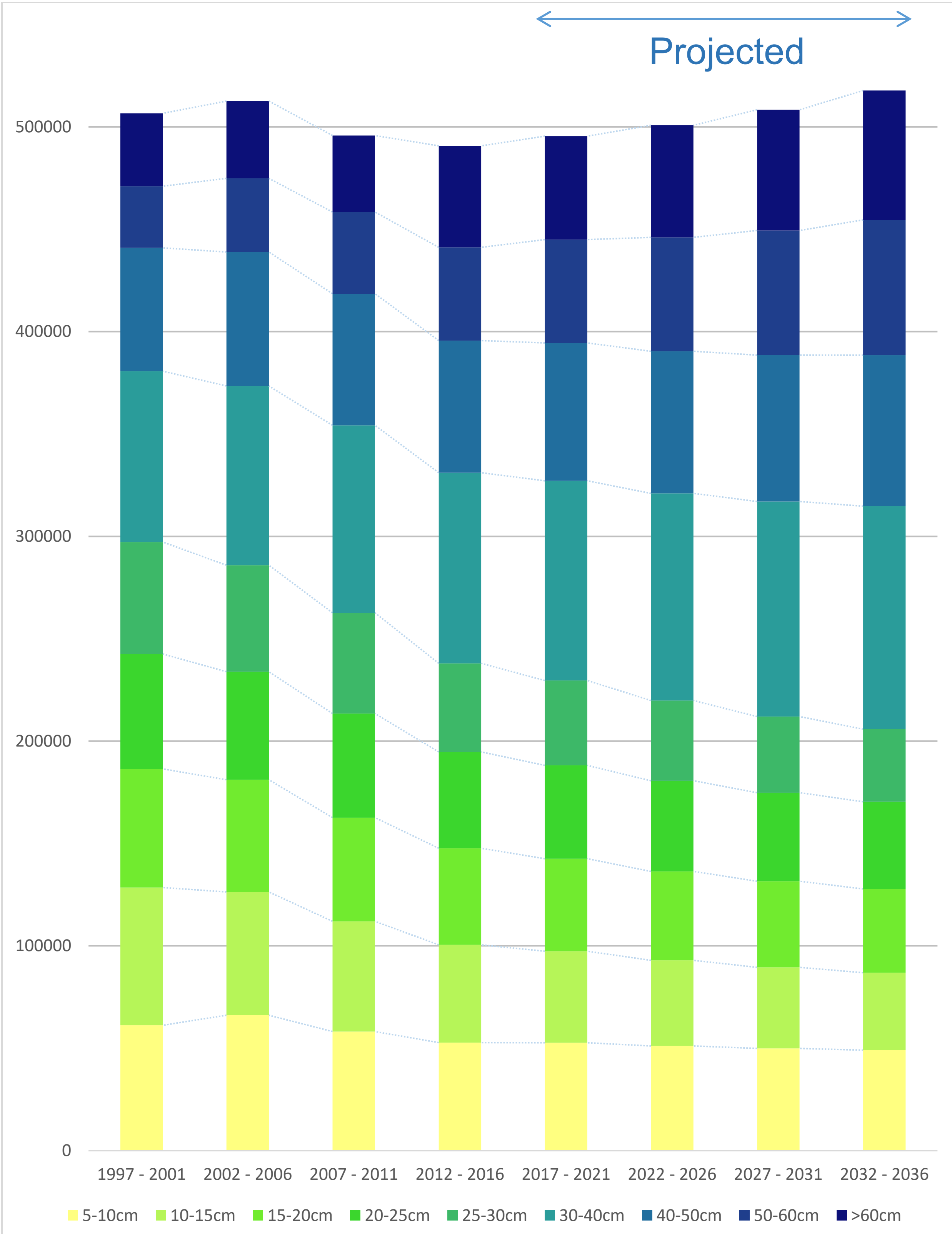


Figure 27 Contribution to carbon sequestration by size groups (kg CO_2 per year)

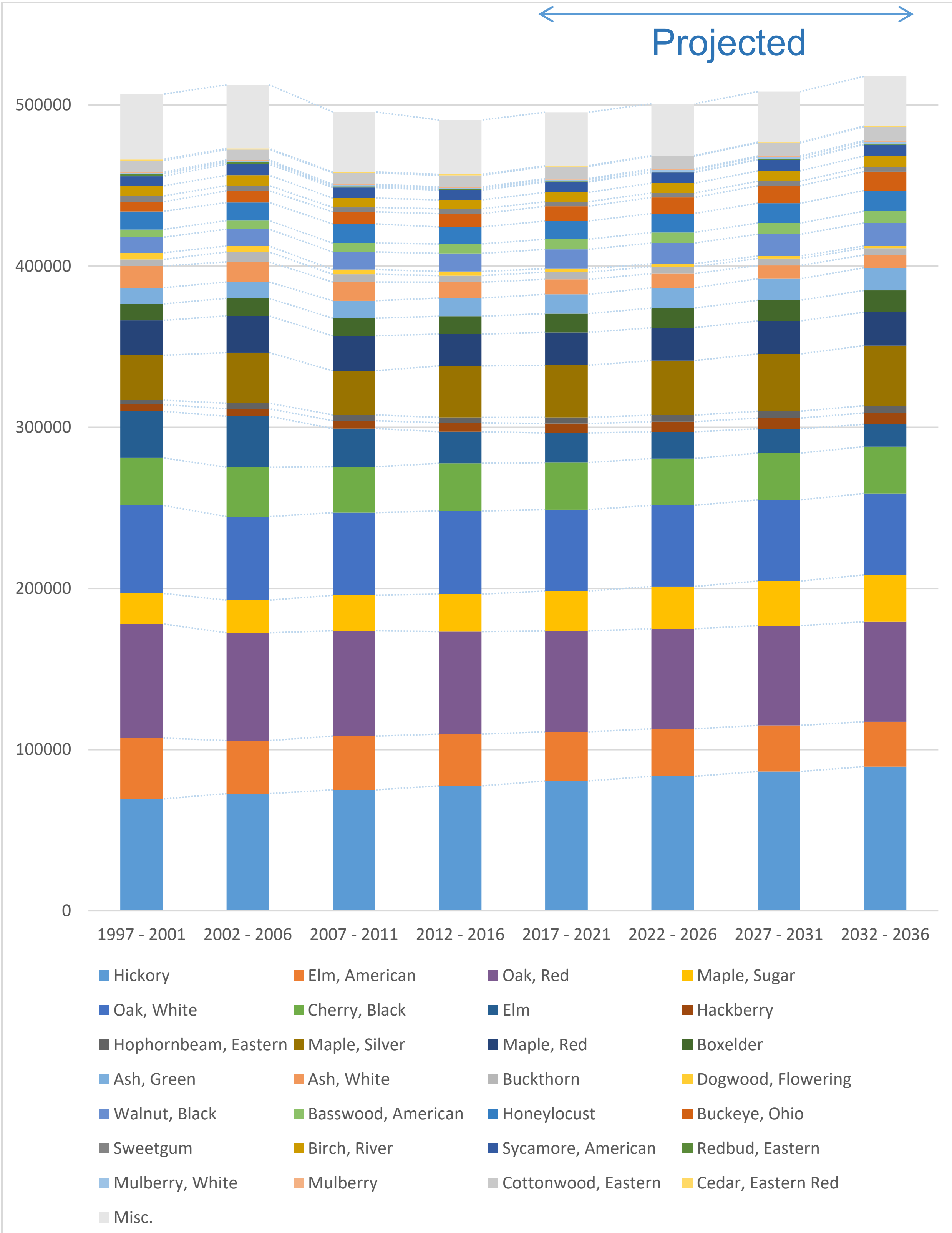


Figure 28 Contribution to carbon sequestration by species (kg CO_2 per year)

6.6 RESULTS AND DISCUSSION

The total number of the trees from 125 sampling sites has decreased from 13,752 in 1997 to 11,946 in 2016. The declining trend is expected to continue, and the number of trees is projected to be 11,475 in 2036. The most prominent change in forest stand is the thinning of smaller trees. All size groups below 30cm are expected to shrink. On the other hand, the number of trees in groups larger than 30cm will increase.

The total carbon sequestration value from the trees remains stable during the 40 years time span. Large trees are much more effective in carbon sequestration. As a result, the loss of sequestration caused by the thinning of smaller trees can be compensated by the increasing number of big trees, even when the total number of trees is declining. In 1997, 58.7% of the total sequestration value is contributed by trees smaller than 30cm. This ratio has decreased to 48.5% by 2016, and is projected to be 39.8% in 2036. With a total sequestration value of around 500,000 kg per year and a total area of 187,500 m^2, the CO_2 removal rate of the investigated forests is 2.67 kg per m^2 per year. This is among the higher end of published carbon sequestration values. According to (Bernal et al., 2018), the CO_2 removal rate ranges from 0.45 to 4.07 kg per m^2 per year for planted forests, or 0.91 to 1.88 kg per m^2 per year for natural forests. When retrieving the sequestration values from i-Tree, I assumed best health condition and full sun. This is an ideal condition which is unlikely to occur in real forests. Smaller trees may grow under canopies and thus have decreased growth and carbon removal rates. Assuming all trees are growing under partial shade, the carbon sequestration value will decrease by around one-third comparing with the full sun condition, so the CO2 removal rate will be 1.79 kg per m^2 per year.

According to the projection, the carbon sequestration efficiency from the forests will not decline within the next 15 years. Nevertheless, this stable trend will not last indefinitely. While transplanting of mature trees is not common, the only case when size groups larger than 30cm can get new members is the growth of smaller trees. In other words, smaller groups act as sources of larger groups. If the thinning of trees under 30cm continues, these sources will be depleted eventually, and the numbers of bigger trees will stop to increase. A collapse of carbon sequestration efficiency can then happen when older trees start to die. While changes in forest stand structure can lead to long-lasting effect on carbon sequestration, it is recommended that actions to be taken immediately to increase the survivability of smaller trees, so the efficiency of forest carbon sequestration can be preserved in long term.

The most common type of forest in Illinois is the oak/hickory forest. The keystone species of these forests, however, are under threat. Drastic changes in species composition can happen soon if no action is taken to preserve the keystone species. The phenomenon of thinning is obvious in oak species. The number of trees that are smaller than 30cm in red oak species has been reduced from 818 in 1997 to 387 in 2016. The numbers are 491 and 313 for white oak species. While the total number of hickories is projected to increase, the growth is contributed by young trees in the 5-10cm size group. The number of specimens with a DBH between 5-10cm has increased from 497 in 1997 to 571 in 2016. For trees with a DBH between 10-30cm, however, their number has decreased from 935 to 832 in the same 20 years. Therefore, despite the increasing abundance of saplings, most of them has failed to survive and grow into larger groups. Being the keystone species, oak and hickories are essential for the forest ecosystem, providing foods and shelters for many other plants and animals (Spyreas, 2009). Decline in numbers of these species can cause crippling effects on biodiversity. Moreover, oaks and hickories also possess high growth rates and are effective in carbon sequestration. Therefore, preserving of these species will be beneficial for both ecosystem health and GHG removal.

In Illinois fragmentation is a leading cause for a thinning of keystone tree species. Agricultural and urban land use changes are the main culprit. Common buckthorn (*Rhamnus cathartica*), an invasive shrub that is adapted to edge habitats (K. S. Knight et al., 2007), has been discovered in many of the sampling sites. This can be an indicator of ongoing fragmentation. While oaks and hickories produce nuts, their seeds are usually dispersed by small animals, notably rodents (Moore et al., 2007). Increased fragmentation and isolation of forest patches will impede the movements of these animals, and thus reduce the efficiency of seed dispersal. In comparison, species have their seeds dispersed by wind (such as maples and ashes) or birds (such as hackberry and black cherry) are less prone to isolation, and thus more likely to proliferate in fragmented habitats (Spyreas, 2009). In this study, sugar maple and hackberry are among a few species that are not affected by the thinning. Numbers of trees smaller than 30cm from these two species have increased during the time period 1997-2016. To support the survival of oaks and hickories, the integrity and connectivity of forest patches need to be improved. This can be achieved by restoring forests along fragmented edges. Corridors facilitating movements of wildlife can also mitigate the isolating effect of roads and accommodate seed dispersal.

Another reason contributing to the decline of oaks is the absence of fire. Low-intensity, periodic surface fires were common during the regime of Native Americans and early European settlements (Brose et al., 2001). The fire is thought to be a major driving force helps to shape the species composition of mixed oak forests. Oaks are tolerant to fire damages thanks to their thick barks and seeds protected in nuts. In comparison, understory species such as maples usually have thinner barks and are more likely to be removed during the fire. Therefore, low-intensity fire favors the survival of oaks and help to increase their abundance (Abrams, 1992). Nevertheless, modern fire safety regulations have eliminated the periodic fires in oak forests. The fire-tolerance feature of oaks is no longer an advantage when competing with understory species. To restore oak abundance, it is recommended to introduce controlled burning to oak/hickory forests.

In addition to fragmentation and fire suppression, other reasons, notably invasive pests, fungal diseases, drought, and soil compaction from human activities are contributing to the decline of oaks and hickories (Borowy, 2020; Juzwik et al., 2010). This has made the conservation of oaks and hickories a challenging and exhaustive task. While proper management practices can help to eliminate or mitigate some of the contributing factors, it may not be enough to bring the trees back to healthy because of the complexity of the syndrome. Knoot et al. (2010) pointed out that saving all the oak forests could be impractical when resources are limited. Instead, the conservation effort should target forests that are most (ecologically and socially) important to maximize the cost-benefit. Suitability analysis can help to identify these important forests. Hernández-Lambraño et al. (2019) investigated the spatial correspondence between many environmental factors (both biotic and abiotic) and oak decline. It was discovered that oak decline was significantly related with human activities (especially farming and ranching), dry soil, and south-facing slopes with low gradients. Based on these empirical relationships, a model was developed to map the risk level of oak decline. These examples demonstrate the potential of spatial analysis in forest conservation by identifying stressed areas and optimizing resource utilization.

One limitation of this study is that both the data collection (CTAP surveys) and data processing (my statistical works) parts have been conducted manually, which is labor-intensive. Moreover, this has limited the “smartness” of the data processing framework. A more sophisticated framework of carbon sequestration estimation should focus on automation, improving efficiency, and reducing human inputs. Contemporary sensing technologies, notably LiDAR, have enabled the identification of tree properties such as heights (H. Huang et al., 2011), canopy sizes (Omasa et al., 2006), or trunk sizes (Bargoti et al., 2015) from remote sensing data. Tree identification processes in earlier studies were not yet fully automatic, requiring human intervention to generate more accurate results. Recent developments in image processing algorithms, however, have validated the possibility of automatic classification and statistics of trees from LiDAR data (Guan et al., 2015; J. Li et al., 2013). This has enabled an automatic collection of forest stand data that is much more efficient than manually collecting species and size data. In the future, I plan to extend the current forest stand structure and carbon sequestration model into a full smart framework including an automatic collection of tree metrics, programmed data processing, and a portal to communicate the results of the analysis. This will not only improve the efficiency of data collection and analysis drastically but also enables the monitoring of forests in large areas (instead of a limited number of sampling sites). In addition, while sensing by LiDAR can be less labor-intensive than manual measurements, it is possible that data collection can be conducted more frequently, compared with the 5-years interval of CTAP surveys. This allows continuous monitoring of forest dynamics and timely feedback for forest management approaches.

6.7 CONCLUSION

This project started from field survey data, it summarizes and predicts forest stand dynamics to estimate the changes in forest structure and carbon sequestration potential. Compared with Chapters 3 through 5 in which the analysis has been conducted at a city level, this project is “larger” in both spatial and temporal dimensions. A state-level analysis can bring challenges to data collection and processing because of the sheer volume of data flow. Moreover, the growth of a tree can take several decades. This means damage to young saplings (the most vulnerable stage) can result in long-term effects. Therefore, a data processing system designed to support forest conservation should utilize model simulation to forecast the outcomes of different scenarios during a long time span. This leads to another feature of smart landscapes: predictive (**Figure 29**). Landscapes tend to change and evolve over time intrinsically (Cantrell & Holzman, 2015). As a result, proper predictions should be conducted during the design process to evaluate how a constructed landscape will behave in the future. A proper prediction allows early detection of hidden harms and dangers in their incubation stage, thus mitigation efforts can be applied in advance before the damage has been obvious.

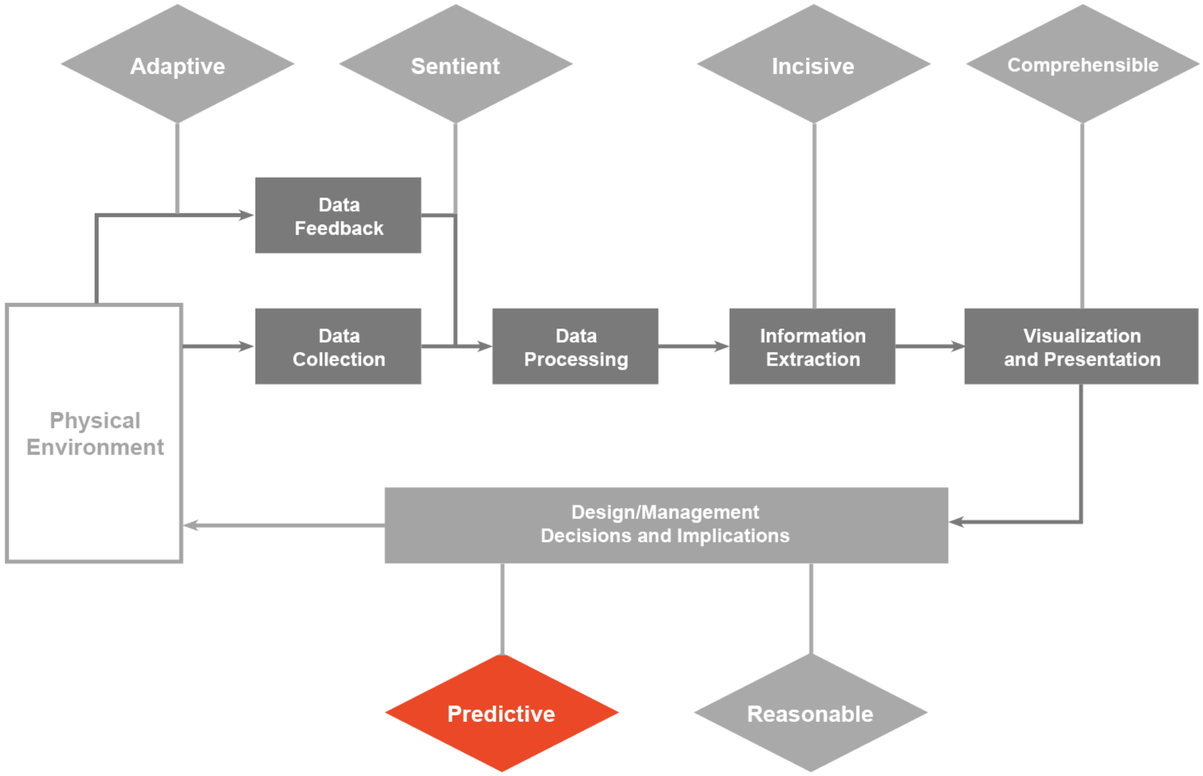


Figure 29 Contribution to the smart landscape framework, Chapter 6

CHAPTER 7: DISEASE VECTOR CONTROL WITH LANDSCAPE INTEGRATED PEST MANAGEMENT: THE PREDATORY LANDSCAPE

7.1 INTRODUCTION

A common job of landscape architects is to work with outdoor drainage facilities and green infrastructure such as retention or detention basins, ditches, ponds, wetlands, as well as other vegetated lands that become submerged after rain. Without proper design and management, all of these facilities can become breeding grounds for mosquitos, which can act as a vector for several potentially lethal diseases including malaria, yellow fever, and West Nile virus. The world is facing a resurgence of vector borne diseases primarily due to climate change and rapid urbanization (See Lit Review Section 2.2.1). For decades, chemical pesticides have been a primary approach for disease vector control. Biological control is theoretically applicable in eliminating disease vectors, but wide implication has not yet been feasible (Thomas, 2018).

In the past decade, we have witnessed the development of integrated pest management (IPM) approaches especially in agricultural production. Landscape architects have also started to adopt the idea as landscape IPM. Most of the applications so far have been focused on agricultural pests. IPM is an information driven approach. The success of the management relies on a thorough understanding of the interaction between the pest and their interaction with different factors within the ecosystem, both biotic and abiotic. Moreover, monitoring the current situation of the pest such as its population and migration is a key to make correct decisions and react in time. This asks for a data processing system that is able to model complex ecological interactions and make responses based on real time data. For these reasons, a smart landscape driven system makes sense for application to this problem.

In order to explore the potential of the smart system and IPM in controlling disease vectors that breed in built landscapes, I propose a concept of "predatory landscapes". This name was inspired in part by pollinator landscapes. A pollinator landscape or garden is designed to provide enough blooming plants in different seasons so bees and butterflies can feed on them. The abundance of food will facilitate the survival and reproduction of the pollinators. A predatory landscape, on the other hand, aims to create an unfavorable environment for the disease vectors to proliferate. The word "predatory" has two meanings here. First, a predatory landscape is designed to support the population of predators feed on the disease vectors, so the landscape is full of predators. Second, a predatory landscape aims to eliminate the disease vectors by creating an environment that is hostile to them (but friendly to humans), so it acts as if the landscape itself is "predating" the vectors.

In this study, I plan to develop a framework for a 'smart' data driven design and management process for the development of predatory landscapes. I hope to provide an effective, environment friendly, sustainable, and possibly cheap way of mosquito control. Moreover, a predatory landscape focuses on constructing suitable habitats instead of simply introducing a predator species. Thus it helps to construct a more resilient ecosystem in a larger context. The elimination of mosquitos in green infrastructures also makes them more attractive spaces for recreational activities. Last but not least, I expect the concept of predatory landscape to promote communication between landscape, ecology, and public health communities for wider cooperation in disease control (Thomas, 2018).

7.2 RESEARCH OBJECTIVES

The goal of this project is to develop the concept and framework of the predatory landscape. It can be perceived as an implication of landscape IPM that focuses on predators. While the project is mainly conceptual, there will also be a quantitative model simulation. The project proceeds in three steps: 1) define and streamline the predatory landscape concept; 2) demonstrate the potential of ecological models in predicting the predator-prey dynamics; and 3) develop a data collection and processing framework following the philosophy of smart landscapes. Research questions include: What is a predatory landscape?; and How might these types of landscapes be usefully employed in landscape design projects?

7.3 MODEL SIMULATION OF MOSQUITOFISH-MOSQUITO POPULATION DYNAMICS

To demonstrate how ecological models can help us to understand the interaction between disease vectors and predators, I have created a stock and flow model to explore the population dynamics of mosquitos and Western Mosquitofish (*Gambusia affinis*). Mosquitofish is a small freshwater fish that is commonly used as an agent of mosquito control. They are hardy in a wide range of temperature but are intolerant to harsh winters, thus are not commonly seen in central or northern Illinois (Illinois Department of Natural Resources, https://www.dnr.illinois.gov/education/CDIndex/WesternMosquitofish.pdf). Both mosquitoes and mosquitofish are ectotherms that are sensitive to winter temperature, and a warming climate may facilitate the proliferation of both of them. Nevertheless, the benefits of warmer winter may be more prominent to one species than to the other, ending up in changing population dynamics. It is expected that mosquito-borne diseases will spread towards the north in this century (Wudel

& Shadabi, 2016). Nevertheless, if the impact on mosquitofish is even more significant, then it can become a more effective biological control agent in the future.

The model is set to a hypothesized water retention pond with an established mosquito population at the beginning. The goal is to simulate the population dynamics of mosquitos after the introduction of mosquitofish, and how the effect can change under a warming climate. The population dynamics were simulated by a stock-flow-fund model (**Figure 30**). Both the mosquito and the mosquitofish were presented by three life stages. The growth and death rate of all stages were affected by the environmental temperature. The death rate increased when the temperature went too low or too high. The seasonal changes of temperature within a year were simulated by a sine wave. Mosquito larvae acted as a prominent food source of mosquitofish, the abundance of which contributed to the survival of both juvenile and adult fishes, reducing their death rates.

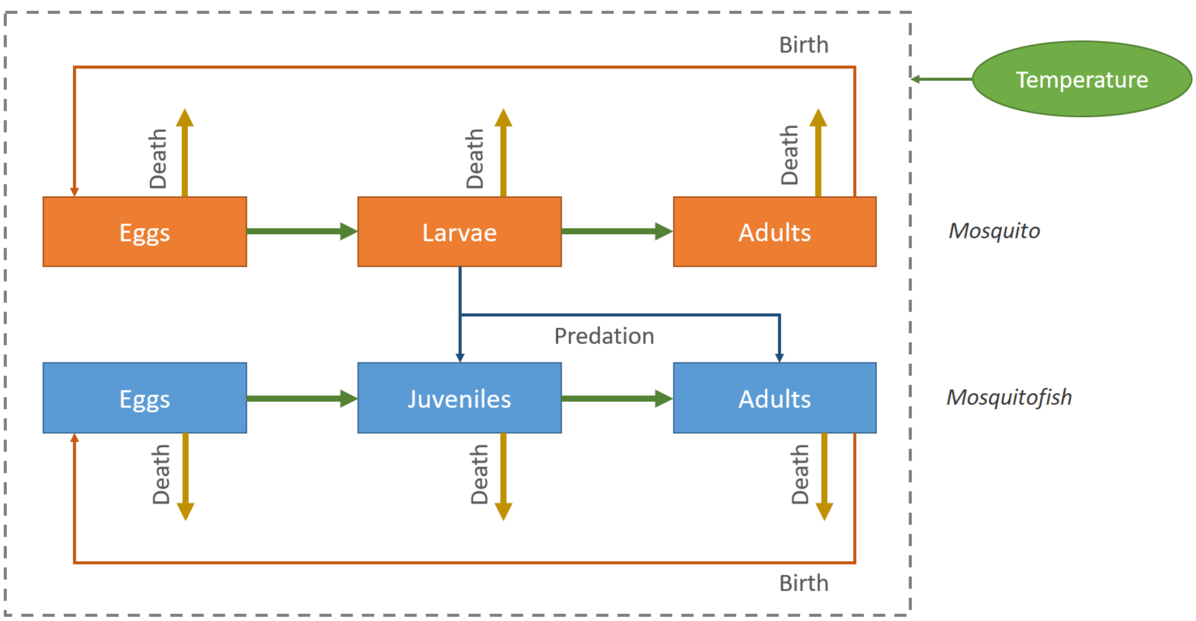


Fig. 30 Population dynamics model of mosquito and mosquitofish

The results of the simulation are shown in **Figure 31**. Under current climate, the pond can breed 93 millions of adult mosquitos during the two years period of simulation. The introduction of mosquitofish will reduced the mosquito population by 12.9%. When the environmental temperature increased by 3°C, the mosquito population would be reduced by 33.3%. While both mosquito and fish could benefit from higher temperature, the effect on fish was more significant. Therefore, mosquitofish can be potentially introduced to Central Illinois as an agent of biological control in the future because the efficiency of the biological control will be benefited from a warmer climate.

7.4 THE SMART FRAMEWORK FOR PREDATORY LANDSCAPES

The first goal of the predatory landscape is to provide favorable environments for predators. A predatory landscape is expected to have high ecological complexity. It should support the survival and proliferation of insects and other small animals that are not disease vectors, which in turn become the food of the predators. In other words, we need to repair the crippled ecosystem and improve the low biodiversity that is common in built drainage facilities and green infrastructures. This complexity poses a challenge to the designer. Multiple landscape elements such as grading, material, and vegetation will affect the quality of habitat. The predatory landscape needs to be able to support multiple species when also providing aesthetic and recreational services. All the different goals can compete with each other sometimes, and trade offs need to be made (R. L. Knight et al., 2003). This calls for a cost effectiveness analysis during the design process. Ecological models and decision support systems can play an important role. This section will introduce the entire workflow of data processing, including the required data inputs and how they can be collected; the core model to analyze the data and project mosquito population, as well as the decision-making based on the outputs of the model.

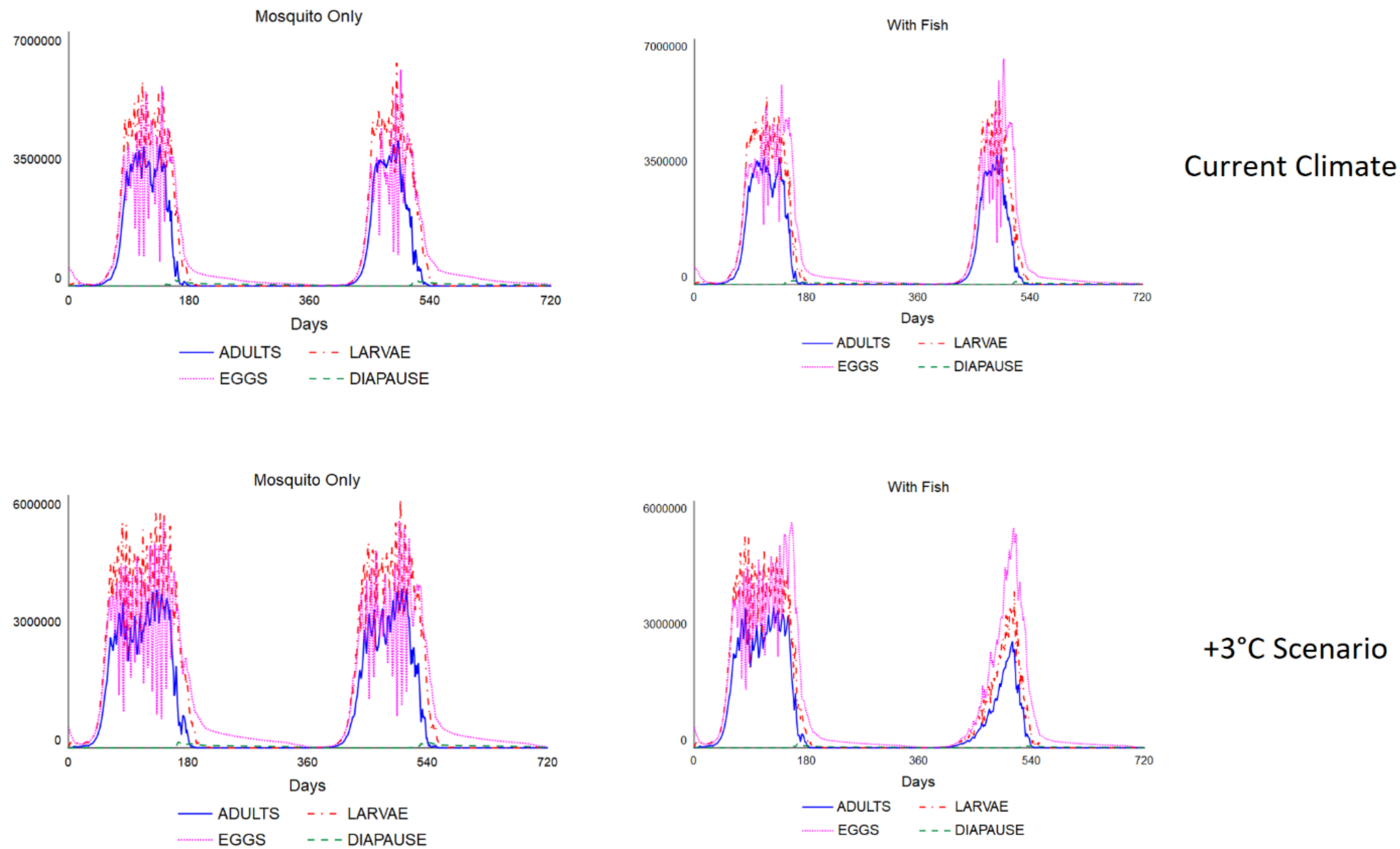


Figure 31 Abundance of mosquitos in different life stages, simulated under the current climate (above) or a scenario in which the temperature has increased by 3°C. The introduction of mosquitofish (right) helps to reduce the mosquito population, ending in smaller peaks during the summer. The effect of predation is more obvious with increased temperature.

7.4.1 Data Collection

The survival of both mosquitos and predators is affected by a wide variety of environmental factors. Being able to gather data related to these factors is essential in the process of evaluating mosquito threats. To minimize human labor and reduce maintenance costs, the preferred method of data collection is through automated sensors. This is especially important in large scale implications when manual survey and counting is impractical. These sensors are connected to a network, reporting their data to the data storage and processing modules.

Water Presence and Water Level. Water is a profound factor that determines the abundance of mosquitos. Mosquito larvae can only survive in the water. The larval stage usually lasts for at least 5 days, and the continuous presence of surface water during the time is required for the larvae to safely emerge into adults (CDC). Ideally, a detention basin that holds water temporarily should be drained within 72 hours after rain to prevent the mosquitos from breeding. Water presence that lasts for more than 5 days can become breeding grounds for mosquitos. Sensors have been developed to detect the water level in wetlands (Danielson, 2002), which can be readily adopted by predatory landscape projects.

Temperature. Temperature can cause great impacts on the survival, growth, and reproduction of mosquitos. The life cycle of mosquitos can vary from as short as a week to more than a month, depending on the temperature (Bayoh & Lindsay, 2004). Moreover, when the temperature is too high or too low, the death rate of mosquitos increases drastically. Low temperature can also put adult mosquitos into diapause, preventing them from reproducing (Diniz et al., 2017). Similarly, many of the predators such as fish, amphibians, and predatory insects are ectotherms. Temperature is a primary determining factor of proliferation as well. Temperature data can be gathered by thermometers within the water, shaded from direct sunlight.

Mosquito Counting. While automatic identification of animals can be very difficult, a recent study by (Kim et al., 2019) has successfully completed a counting of mosquito population by computer. Video clips were recorded then analyzed by deep learning networks to identify and count the mosquitos. This is still a nascent idea, and we have yet witnessed large-scale implications of similar approaches. Nevertheless, a sensor that is capable to count the number of mosquitos in an automatic manner will be exceptionally promising in predatory landscapes.

Vegetation. Vegetation provides food and shelter to both mosquitos and predators, and thus should be considered as a factor in the population dynamics model. The sensing of vegetation in a predatory landscape can be challenging. The spatial resolution of satellite or aerial images is not enough to identify typical wetland plants such as grass and sedges. Fortunately, state-of-the-art technologies in deep learning have brought the possibility to identify plants from close images (Grinblat et al., 2016; Sun et al., 2017). In a predatory landscape, images or videos of the site can be captured by cameras. The presence of vegetation will be then evaluated by image identification.

Other Data Sources. In addition to sensors, other sources of data can also be adopted if they can help in evaluating mosquito threats. For example, there are online portals where residents can report mosquito bites (https://www.nsmad.com/fight-the-bite/report-biting-mosquito-activity/). A connection to these data sources helps us to better understand the big picture of mosquito activities.

7.4.2 Data Processing Framework and a Population Dynamics Model

A population dynamics model is the core of data processing and interpretation. It projects the population trends of both mosquitos and predators based on the environmental factors (such as temperature or water) that affect their survival and reproduction, as well as the predation activities. A simplified example has been introduced in my proposal. The model simulated the population of mosquitos and a common predator, Western Mosquitofish (*Gambusia affinis*), using stock and flow analysis. The major determining factor is the temperature, which will affect the survival and reproduction rates of both species. Predation is imitated as a process that reduces the mosquito population directly while increasing the survival rate of the predator by providing more food sources. The output of the model is the projected mosquito population which can be used to evaluate the mosquito threats. To be applied in a real predatory landscape, this model should be expended, considering the effects of other environmental factors (such as water and vegetation), and the interaction with more predator species (such as frogs and dragonflies).

The Threshold of Mosquito Threats. The goal of a smart landscape is to provide benefits and welfares to nearby residents, aiming to fulfill their requirements and expectations. Therefore, for a predatory landscape, its goal of mosquito control should be set up to meet the requirements of human health, instead of centered on the mosquito itself. Comparing with reducing the mosquito population by a certain percentage, it is more reasonable to keep it below a certain level so the mosquitos will not cause harm to the health and comfort of the nearby residents. Determining this threshold can be a challenging task, and I will do more research on the relationship between the number of mosquitos and the risk of infectious diseases.

Decisions and Actions. When the mosquito population has reached or is projected to reach the level that can pose a threat to the residents' health, pest management actions need to be taken to reduce it to the safe range. The management action should start with measures that are cheap and environmentally friendly. For example, in a detention basin that is designed to hold water temporarily after rain, the drainage valve can be activated if water presence has been detected to last for longer than 72 hours, so mosquito larvae will not have enough time to develop. In ponds and wetlands that hold water permanently, biocontrol agents such as the spores of bacteria *Bacillus thuringiensis* subspecies *israelensis* (Bti) can be released to inhibit mosquitos (Seo et al., 2010). While chemical pesticides can be highly potent, they can also cause collateral damage, thus shall be reserved as a last resort when other attempts fail (Barzman et al., 2015).

In addition to the mosquito control actions, we can also send warnings to the nearby residents when the mosquito threat is high, reminding them to take preventive measures such as insect repellent or protective gear.

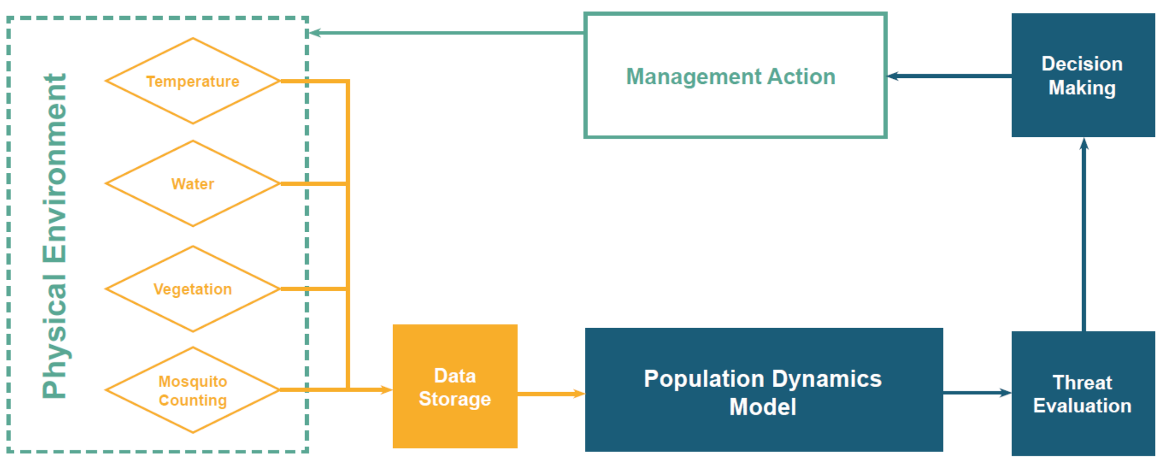


Figure 32 Implication of the smart landscape framework within the context of predatory landscapes.

7.5 BENEFITS OF PREDATORY LANDSCAPES

The construction and maintenance of predatory landscapes can be costly, involving the installation of sensors, the development of the data analysis tools and software, as well as possible human labor in the data collection and processing. Nevertheless, these landscapes can bring multiple benefits, which help to justify the investment. Upon success, the predatory landscape will not only reduce mosquito threats, but also contribute to health, environment, and society. In this section, I will introduce these benefits one by one.

Preventing Mosquito-Borne Diseases. Mosquito-borne diseases have been causing substantial damage to both human health and the economy all over the world. In 2019, the global cases of malaria were estimated to be 229 million, with 409,000 died of the disease. The direct economic cost of treating the disease has been estimated to be at least US$ 12 billion per year, while the indirect costs (such as loss of economic growth) can be many times higher than this value (CDC, 2021). Successful mosquito control not only helps to save lives but also helps to prevent economic loss.

Facilitating the Delivery of Ecosystem Services and Neutralizing the Effects of Ecosystem Disservices. Activities in natural areas will provide chances that humans come to close contact with disease vectors (Hassell et al., 2017). The increased risk of infectious diseases has been identified as a type of ecosystem disservice (Wu et al., 2020). In addition to the health risks, this also leads to prejudice that many people perceive green spaces and natural areas to be unsafe. For example, some of the residents in rural Tennessee wished to be located far away from wetlands (Sims et al., 2016). In my interview with Brent C. Lewis, the University Landscape Architect of UIUC, it was mentioned that many of the residents opposed the plan to restore prairies in the campus because of the concerns of unwelcomed insects and wildlife. Both the health risk itself and the negative impression have prevented many people from enjoying the ecosystem services. Creating green spaces that are free from mosquitos helps to invite more people to visit the places, and thus facilitating the delivery of ecosystem services.

Smart Use of Pesticides. While chemical pesticides have been the dominating method of pest control, they can cause significant collateral damage, killing both the pest and the predators. In addition, overuse of pesticides can lead to the rapid development of resistance in pests, reducing the potency of the pesticide in the future (Shakeel et al., 2017). A problem of the traditional pest control approach is that pesticides are used without proper evaluation of pest threats (E. Birch et al., 2011). In a predatory landscape, the data driven decision making process helps us to determine when it is necessary to apply pesticides. This helps us to reduce the overall usage of pesticides, maximizing their potency, and limit the side effects.

General Ecological Benefits. Predators are at the higher trophic levels in the food web. Their proliferation depends on the abundance of insects and other small animals, which in turn depend on thriving vegetation and healthy microflora. Therefore, a predatory landscape focuses on constructing or restoring a robust ecosystem instead of simply introducing predator species. In addition to predators, it also provides habitats for other wildlife and supports the ecological restoration efforts in a larger context.

Resilience. One major challenge in the control of mosquito-borne diseases is that climate change has facilitated their spreading (Caminade et al., 2019; Semenza & Suk, 2018). The predatory landscapes allow us to detect and monitor mosquito outbreaks that are possibly related to climate change, which will provide information to the studies of the relationship between climate and mosquito-borne disease. The result of these studies can in turn help us to update the system of predatory landscapes, improving its performance under the changing climate.

Public Participation. Public participation and citizen engagement are important features of smart cities that should be adopted by smart landscapes as well. The predatory landscapes will provide opportunities that the residents can engage in the process of mosquito control. They can provide information by reporting mosquito activities. The data processing framework can also give advisories and alerts, increasing public awareness of mosquito threats. Moreover, with data and information shared through the smart framework, the general public can also participate in the decision-making process. A more detailed discussion about the role of the general public in smart landscapes can be found in Section 8.5.

7.6 CONCLUSION

In this study, I propose a smart landscape framework that helps to reduce the risks of mosquito-borne diseases in stormwater facilities and wetlands through biological control. These predatory landscapes, however, should net be considered as mere mosquito control infrastructure. They can be integrated into ecological restoration plans on larger scale, or be connected to the data network of a smart city. From a wider perspective, the predatory landscape can be interpreted as a data driven design and management protocol to create healthy and robust ecosystems that provide services to both humans and nature. This requires smart framework itself to keep evolving over time, being able to respond to future environmental changes. In other words, the smart framework needs to be "adaptive" (**Figure 33**).

One limitation of this study is that the population dynamics model is highly simplified, containing only one species of predator (the mosqitofish). Therefore, the next step of the study is to extend the model, including other predator species such as aquatic Arthropods, to better simulate the predation relationship in the real world. Controlled experiments can also be conducted to investigate the effect of biophysical factors (i.e. vegetation, water depth) on predator and mosquito populations.

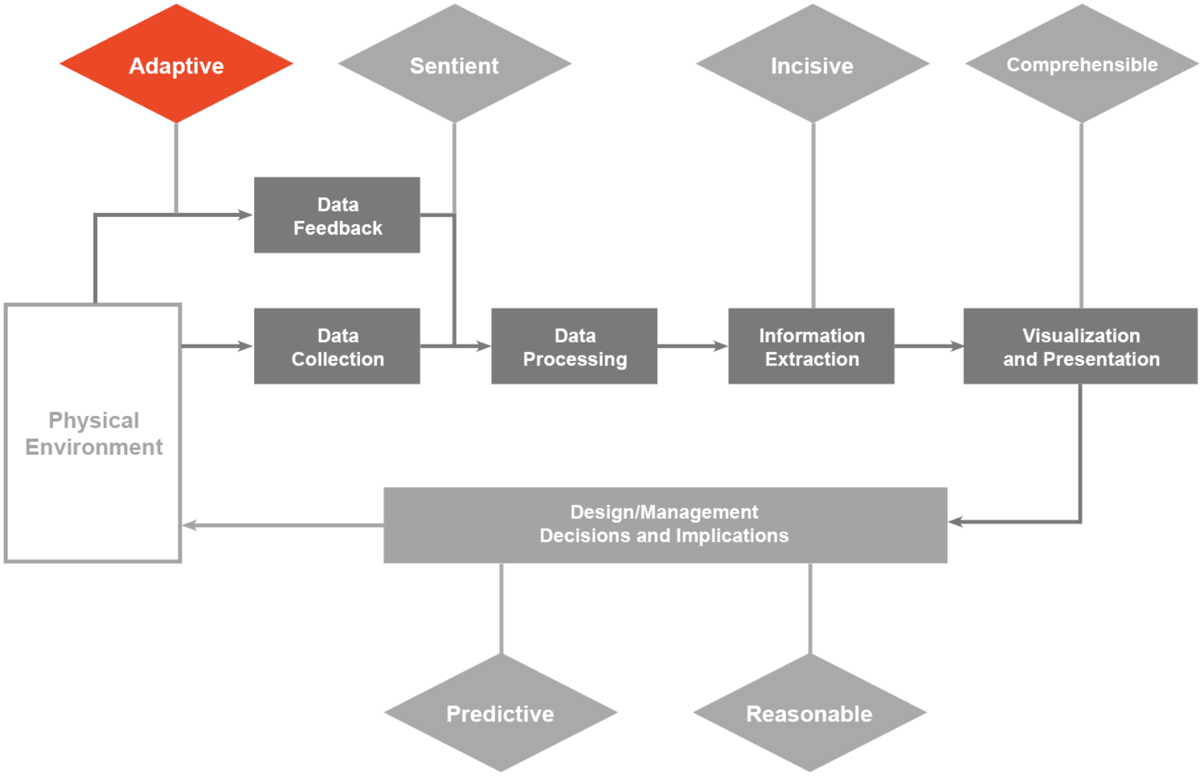


Figure 33 Contribution to the smart landscape framework, Chapter 7

CHAPTER 8: SUMMARIZING SMART LANDSCAPES

The concept of ‘smart landscapes’ was initiated as a simple idea that in-depth modeling and data processing should have a more important place in landscape design. With the experiences and lessons learned from the research above, the concept has become more complex and nuanced for me. In this chapter, I attempt to work through those nuances. I provide a summary of the smart landscape concept and ideas for how it might reshape the landscape design process, with emphasize toward my main interests in ecologically based information and designs. I do this in the following 5 sections: (8.1) lessons learned from the above research and how these works have helped the smart landscape concept to advance and lead to, (8.2) an advanced smart landscapes framework, (8.3) which technologies can be most readily adapted toward inclusion in smart landscape design processes, (8.4) how smart landscapes can help to fill the gaps between ecological principles and current landscape design practices including the challenges in implementing ecological design ideals, and smart landscapes’ role in overcoming these obstacles, and (8.5) the role of the general public in smart landscapes.

8.1 LESSONS LEARNED

The 5 research projects presented in this dissertation have been carried out in different scales and contexts. Nevertheless, they all followed the same philosophy: identify gaps in what we know, discern the costs of not knowing, summarize fragmented data and sources, and translate it into design strategies form a smart landscapes perspective – i.e. how we might approach the problems using ICT and advanced models. Chapter 3 - impact assessment, Chapter 4 - comparative study, and Chapter 5 - ecosystem services delivery have all been carried out at a city scale. In these projects, I collected data in different categories that reflected urban structure (such as land use, transportation network, or vegetation). The data was then interpreted with different ‘smart’ technologies and models (such as transportation or urban growth models) to delineate how current urban structure and future development may affect ecosystem services from parks and green spaces. Suggestions were then given on how to preserve urban ecosystems in a developing city and facilitate the delivery of ecosystem services. In addition to data processing that has been described in the chapters, it is also planned that both the results and the models to be available on an online platforms. Planners, designers, other decision makers, as well as general public are allowed to access the data online, or even customize variables in the model and create their own forecasts.

The estimation of forest carbon sequestration potential (Chapter 6) combined a forest stand structure model with a growth model (for single tree). In other words, it manipulated data in both macro and micro scales. This helped to translate the CTAP survey data (fragmented, collected at numerous sites) into knowledge of a bigger picture, such as carbon flux into the forests and suggestions on the preservation of carbon sinks. While existing data processing has been completed manually, there is the potential that monitoring and evaluation of forest stand structure can be completed automatically thanks to technologies such as LiDAR and image identification.

The Predatory Landscapes (Chapter 7) has summarized data from different sources (literature review, model simulation, and field observation) to evaluate how biophysical factors in stormwater facilities can affect the level of mosquito threat. While guidelines of mosquito control in wetlands, ponds, and detention basins already exist, they may conflict with each other because of the complexity of ecosystems, and thus provide limited information for designers. For example, thinning of vegetation has been found to reduce shelter areas that hide mosquito larvae from predators (Russell, 1999; Thullen et al., 2002). On the other hand, predation of adult mosquitos can happen in dense vegetation by arachnids (J. Medlock & Snow, 2008). Moreover, the presence of native plant species contributes the abundance of *Bacillus thuringiensis*, a bacterial pathogen that is lethal to mosquito larvae (Gardner, 2016). Therefore, a smart system supporting the design of mosquito-free stormwater facilities need to be location-specific, being aware of the design context and the prioritized goals of the project.

In sum, the concept of smart landscapes is still in its nascent stage. Experiences from the research projects have provided vital insights and promoted the development of the smart landscape concepts. A sophisticated framework of smart landscape, however, can only be streamlined through numerous real-world applications. Similar to smart cities, smart landscapes should also be a flexible and adaptive concept that evolves with changing environments and advancing technologies.

8.2 AN ADVANCED SMART LANDSCAPE FRAMEWORK

With lessons and experiences learned from the research projects, an evolved framework of smart landscapes can be described in the following steps (**Figure 34**): data collection, data processing, information extraction, visualization and presentation, design/management decision and implications, and data feedback.

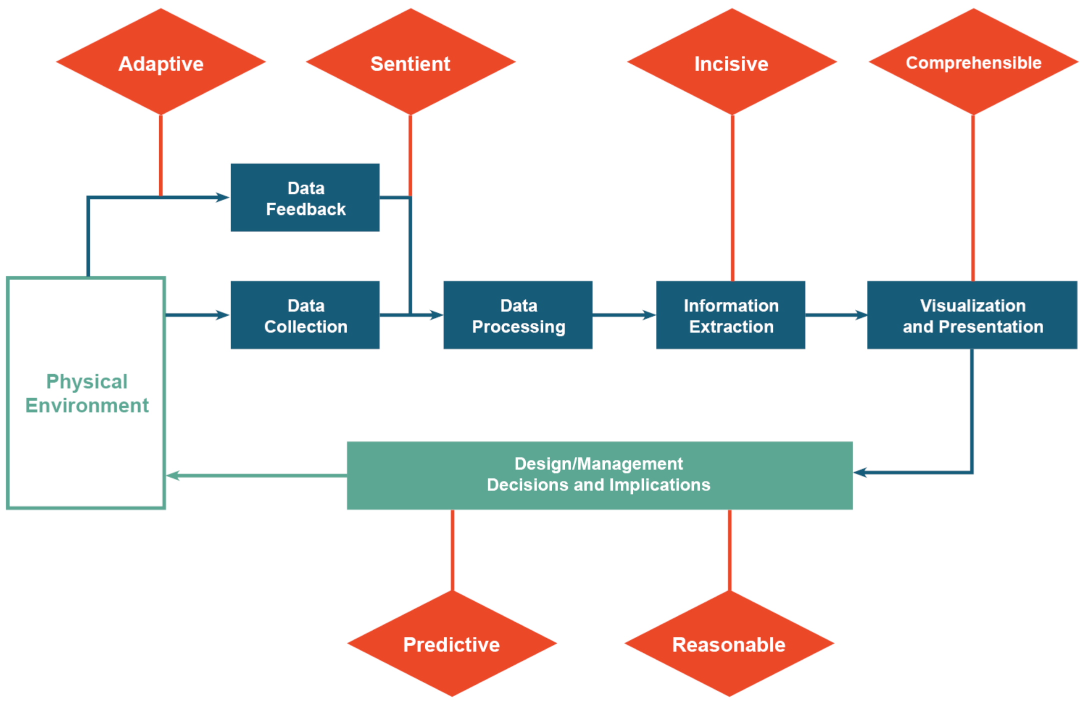


Figure 34 The evolved smart landscape framework and its six features

Data Collection. The first step of the smart system is to acquire data from both the project site and its context. The advance in technologies such as remote sensing and Big Data has enabled plenty of data sources, while the search process has also been made easier. Nevertheless, the most important part of data collection is to identify which type of data is related to the design project.

Data Processing. The collected data will be analyzed using models and PSS/DSSs. This allows us to both evaluate the current situation and predict what will happen in the future. This is especially important for ecological design. Because of the complexity of ecosystems, even a small disturbance can cause chain reactions and end up changing the entire system. A reasonable ecological design should rely on model simulation instead of intuition to predict its outcomes.

Information Extraction. After the data processing, the smart system look for information that may be helpful in the design decision making.

Visualization and Presentation. Results of the data processing are visualize in a way that can be easily understood by designers and other clients who do not possess professional knowledge in ecology.

Design/Management Decisions and Implications. Design and/or management decisions are made based on the information provided by the smart system.

Data Feedback. A landscape needs to evolve over time to provide better services in an ever changing environment. This asks for continuous monitoring of both the project and its context after construction. The design implication and management plan adjusted according to the timely data. In addition, models used in the smart system can also be calibrated and improved during this process. Ideally, the smart system and the design project will form a feedback loop that is evolving constantly.

8.2.1 Six Features of Smart Landscapes

In smart city applications, a common mistake is to define smartness by the number of technologies applied, which can lead to technocracy and urban labeling (Caragliu et al., 2011; Hollands, 2008). Similarly, the smartness of a landscape should not be measured by the number of technological applications. Instead, a landscape can only be considered as smart when technologies have facilitated the design and management processes and led to more successful landscape design. To evaluate the smartness of landscapes, here I propose criteria of 6 features: reasonableness, incisiveness, sentience comprehensibleness, predictiveness, and adaptiveness. A smart landscape is expected to meet all the conditions simultaneously.

Sentient. A smart data-processing system has to be aware of its target users and context of application. It should aim to meet the expectations of the designers and produce context-specific results, instead of general data analysis that may not fit the specific design environment.

Incisive. Without proper interpretation, raw data can provide little knowledge. The smart landscape framework should include in-depth data processing and analysis, translating data into useful information that can support decision-making.

Comprehensible. The results of the data processing should be presented and visualized in a way that is easily understandable to non-professionals. While the data processing is usually conducted by professionals and researchers, the results will be eventually presented to designers, stakeholders, and general public who are not experts in the field of study. It is important to create a data presentation interface that is friendly to these non-professional users.

Reasonable. A common limitation of the traditional design process is that decisions are usually made out of mere creativity instead of reasoning. In a smart landscape, designers are expected to provide adequate evidence for their decisions, such as data and information collected from the real world, or scientific findings.

Predictive. Landscapes are rarely temporary installations. In most cases, a landscape is expected to provide services for decades if not centuries after construction. Therefore, possible changes in biophysical (such as climate or ecosystem types) and social-economic (such as demographic or industry) factors in the context of a landscape need to be considered during the design process. Model simulation can assist in forecasting how changes in the environment can affect the performance of the landscape through complicated mechanisms.

Adaptive. Landscapes are dynamic mediums that are constantly evolving. Similar to the landscape itself, the design framework should also be dynamic and evolve over time to reflect the changes in the landscape. This requires feedback in the flow of data. The landscape should be monitored continuously, collecting information that reflects landscape performance, such as biodiversity, stormwater mitigation efficiency, or comments from stakeholders. Management policies should be adjustable based on the situation at hand. Moreover, the framework is also expected to be flexible and extendable, ready to adapt new technologies.

8.2.2 Relationship between the Smart Landscape and Existing Concepts

Concepts such as responsive landscapes (Cantrell & Holzman, 2015), GeoDesign (Steinitz, 2012), and evidence-based design (Ulrich et al., 2010) have all provided vital inspirations for this dissertation. The most important idea I have learned from responsive landscapes is that landscapes are dynamic and temporal mediums, which tend to change over time. Therefore, landscape architects should aim to design protocols and methodologies rather than static objects. The design process itself needs to evolve over time so it can keep up with the changing landscape (Cantrell & Holzman, 2015). This has helped me to develop concepts that a smart landscape needs to be dynamic, be aware of its surrounding environment (sentient), be prepared for possible environmental changes in the future (predictive), and be able to evolve through its life span (adaptive). Nevertheless, while the concept of responsive landscapes itself has emphasized the importance of computer modeling, many of the real world examples have been designed intuitively without proper reasoning and reliable evidence. GeoDesign and evidence-based design, on the other hand, have provided powerful tools for in-depth data processing (incisive). The data processing results can be presented on user-friendly interfaces (comprehensible), and help to back design decision making with scientific evidence (reasonable).

The limitation of these practices, however, is that there is usually a lack of dynamicity and adaptiveness that can be found in responsive landscapes. The smart landscape framework aims to combine the advantages of both responsive landscapes and geo/evidence-based designs.

8.3 SMART TECHNOLOGIES IN THE CONTEXT OF SMART LANDSCAPES

With more than a decade of development, technologies involved in smart city approaches have formed a complex ecosystem. Many of them have been linked to or integrated with others to better perform in the demanding smart framework. Inspired by several published reviews (Lea, 2017; H.-P. Lu et al., 2019; Stratigea et al., 2015), I will classify the technologies based on their roles in the smart cities framework, then evaluate their possible contribution to smart landscapes.

Sensing and Monitoring. Since both smart cities and smart landscapes are data-driven approaches, sensing and monitoring play an essential role in the acquisition of raw data. Devices from street cameras to satellite sensors all provide valuable information during the planning, construction, maintenance, and management of smart cities. Similar implications of sensors can also be seen at the landscape level. One example was the park benches that were designed to track human activities in the park by collect anonymous location data of nearby cell phones and other Wi-Fi enabled mobile devices (Barth, 2017).

Landscape design and urban planning are different approaches that focus on different scales and have different philosophies. Therefore, landscape designers can ask for a very different data set comparing with urban planners. Sensing technologies that are commonly implied in smart cities may provide little useful information to smart landscapes unless they are optimized to support landscape design as well. For example, satellite imagery from Landsat or Sentinel is frequently used to monitor urban structures. With a spatial resolution of 25 by 25 or

30 by 30 meters, changes in roads, buildings, and vegetation in a city can be easily identified from these images. Nevertheless, many of the elements in landscape design, such as trees and paths, can be much smaller comparing with even a single pixel from the satellite images. In other words, the data is not detailed enough when monitoring a park or green space. Aerial images from aircraft or drones have a much higher spatial resolution, but these images can be expensive and are not usually available to the public. To support the smart landscape approaches, a new data infrastructure (including both data collection and sharing) needs to be established to provide high-resolution remote sensing images that are affordable and accessible. In addition to optimizing the existing sensing technologies, landscape designers may ask for the development of new sensors as well. Being an essential element of outdoor landscapes, biological and ecological metrics can be especially hard to acquire, for the sensing of living organisms is very difficult to automate (MacLeod et al., 2010). A recent approach by (Kim et al., 2019) developed a deep learning-based sensing system that was able to monitor the population of adult mosquitos automatically. Their device attracted mosquitos with CO_2 and UV light as baits. The presence of mosquitos was recorded by a surveillance camera. The number of mosquitos in the video clip was then counted using multiple deep learning networks. Similar sensors can be highly valuable in supporting ecological design by providing real-time data of the pest population.

Big Data and Big Data Analytics. Cities are constantly generating enormous amounts of data. Without the ability to collect, store, and process big data, the smart city is not possible. Landscapes, while smaller than cities in most cases, are still highly complex and dynamic systems. This means the data generated in landscapes also exhibits the classical characteristics of big data: volume, variety, and velocity (De Mauro et al., 2015). In other words, a smart landscape approach needs to be able to handle data that is large in quantity (volume), comes from different sources and in different formats (variety), and changes constantly (velocity).

The interdisciplinary nature of landscape architecture means that landscape designers are trained to consider a wide variety of information during their design processes (Gazvoda, 2002). The volume of big data, however, is far beyond the capacity a human can handle. Therefore, without proper analysis by the computer, big data can simply overwhelm the designers instead of supporting them to make design decisions. Proper processing and analysis need to be carried out to extract useful information from the data. Royds (2018) evaluated three different approaches of presenting big data to landscape architects. The first method simply showing the data as it was, which provided little help. The second or the "GeoDesign" approach combined data from different sources and created a gradient map. While it promoted a much better understanding of the site, the method was only suitable to represent geospatial data and could answer only one question at a time. The third approach conducted an in-depth statistical analysis of the data and visualized them into an infographic. Despite it was considered to be the most efficient way to present big data, this approach was demanding on the analytical process. Improper analysis or processing could end up in exhibiting misleading information. In sum, while big data itself is an essential element of the smart landscape approach, proper analysis and visualization of the data can be a more important and more challenging part. It is possible that new software and tools need to be developed (or modified from existing products) to accommodate the needs of the designers.

Cyber Physical Systems and the Internet of Things (IoT). The cyber-physical systems, notably the Internet of Things (IoT), allow physical devices to be represented on the internet (Lea, 2017). The implication of IoT in smart cities enables direct communication between sensors and actuator devices (Gubbi et al., 2012). IoT helps to promote the cooperation and integration between different devices and has shown its potential in applications such as smart energies and smart transportation. Similar approaches can be adopted by the smart landscape. Because of the scale of landscapes, however, the IoT in a smart landscape needs to extend beyond the landscape itself, connecting to its context. For example, a rain garden is more effective when it can be synchronized with upstream and downstream drainage facilities. In addition to linking the devices within a landscape, it is more important for the IoT to integrate a series of landscapes and related facilities so they cooperate towards the same goal.

Open Data and Communication. In the context of smart cities, open data refers to the policies that require or encourage public agencies to publish their data in a way that is easily accessible for the residents (Lea, 2017). Almost all of the modern cities are sharing their data to some extent. For example, real time data of traffic and road conditions can be acquired from different map applications. The sharing of landscape data, however, is much less sophisticated. In many places, it is difficult to find a map of paths and trails, not to say the real time condition. To overcome these shortages, better platforms and interfaces need to be developed so the data from parks, natural areas, and other landscapes will be available to the public.

Similar to IoT, data sharing and communication should not be restricted within a single landscape. In Los Angeles County, there is an ongoing project which is described as "park prescriptions" (UCLA Luskin Center for Innovation, 2017a). The project aims to share the information of local parks with medical providers. As a result, patients can be directed to proper parks where their desired exercises and recreational activities can be conducted. By making landscape data transparent and easily accessible, services provided by the landscapes can be better connected with the persons in need.

Modeling. Models are useful tools in interpreting data as well as predicting the outcomes of design or management decisions. Current smart city approaches have incorporated different models such as those simulating transportation (Masek et al., 2016) or energy consumption (Hayashi et al., 2018) in cities. Similar to many other tools, however, models developed to support urban planning may not be suitable for landscape designs because of the difference in scales and design elements. The same problem can be found in other models that have the potential to be adopted in landscape design. For example, when ecological models may help the designers to create more robust ecosystems in landscapes, the majority of these models have been developed by ecologists who are working in remote, well-protected natural areas that are free of human disturbances (Marjo van Lierop et al., 2011). As a result, the human aspect is usually lacking in these models, rendering them incapable to simulate how the ecosystem can interact with the visitors and provide services to them. While models from related disciplines can be adopted in the smart landscape approach

8.4 CONNECTING ECOLOGY WITH LANDSCAPE DESIGN

The basic component of an ecosystem, be it living (such as vegetation) or inorganic (such as soil and water), makes essential design elements in landscape architecture as well. From the perspective of ecologists, a landscape can be defined as a mosaic of different ecosystems. Because of this close relationship, I argue that a significant portion of the services provided by designed landscapes is originated from ecosystem services. Failing to follow ecological principles in the design process can result in a landscape that is incapable to provide its services as expected, or lead to unexpected consequences. For example, mowed lawns have been a staple of American landscapes since the 19^{th} Century (Byrne, 2005). People appreciate lawns in their backyards, parks, and public green spaces for providing great recreational opportunities and aesthetic values. These lawns, however, are crippled ecosystems with very low biodiversity, thus unable to deliver as many services as healthy ecosystems. Lacking enough flowering plants, mowed lawns fail to provide enough food sources for pollinators such as bees, resulting in a decline of their populations, which in turn damages agricultural production and causes economic losses (Lerman et al., 2018). It is only when ecological principles have been adopted in the landscape design process that concepts like pollinator gardens can be proposed. Now we can see more heterogeneous and robust ecosystems such as low-mow zones and restored prairies in our cities, ready to support the proliferation of pollinators. In summary, unawareness of the ecological consequences can lead to design mistakes that make the landscape that provides inadequate services, sometimes causes more harm than benefits. Models that predict the ecological impacts of the design should be incorporated in the design process to avert these mistakes.

8.4.1 Challenges in Ecological Landscape Design

We have witnessed successful examples of landscape designs that follow ecological models. The implication of ecological design strategies in the real world, however, is not as common as the rhetoric in literature (Calkins, 2005). Common obstacles preventing the adoption of ecological principles in landscape design includes: lack of information, lack of time for research, lack of evidence that helps to justify the performance, and lack of support from the public and stakeholders (Calkins, 2005; Marjo van Lierop et al., 2011). These problems need to be addressed before we can see broader applications of ecological landscape design.

Lack of Information. One of the most prominent reasons that prevent landscape architects from adopting ecological models is that the ecological data itself can be extremely hard to find and use (Poisot et al., 2019). Ecological models, as well as the concept of ecosystem services, emerged from the field of natural sciences (Marjo van Lierop et al., 2011). Studies in these topics typically follow the classical scientific approach of reductionism. Experiments are designed and conducted to test small, discrete, and reduced ideas (Creswell & Creswell, 2017). As a result, information and data generated by ecological studies can be highly fragmented. It does not help that there is a lack of a universal standard of data structure in the field of ecology, which further impedes the attempt of data integration (Poisot et al., 2019). To facilitate the transdisciplinary adaption of ecological data in landscape architecture, the data needs to be better organized so it is easier to search and review. Platforms to present domain specific can be established (Poisot et al., 2019).

Another gap in the availability of ecological data is that the leading philosophies in ecological study and landscape design are different, especially in their views of humans. The concept of ecosystem services was developed from an eco-centric perspective, in which humans are not usually considered as an element of the ecosystem. Landscape design, on the other hand, are more anthropocentric, aiming to provide services to humans. The majority of ecosystem services studies take place in well preserved natural areas that are far from the cities (Marjo van Lierop et al., 2011). Models developed in these studies often lack the ability to evaluate the interaction between humans and the ecosystem, thus provide limited help to landscape design. To develop ecological models that are capable to support the landscape design process, we need to encourage communication and cooperation between ecologists and landscape architects. There should be more ecological studies conducted in densely populated areas to evaluate the effects of human interferences.

Lack of Research. Most frequently, landscape architects are trained as designers rather than researchers. While many landscape architects are fully aware of the benefits of adopting ecological principles, they can find these principles too difficult to understand, or do not have enough time to do the research (Calkins, 2005). A survey of architects, interior designers, and landscape architects by (Szenasy, 2002) revealed that the majority of the respondents (93%) had expressed their interest in green design. Nevertheless, 70% of them were concerned that they might not possess adequate knowledge and skills to participate in a real green design project, which required a thorough understanding of ecology. Therefore, better training and education should be provided to the landscape architects so it is easier for them to participate in landscape based ecological studies.

Lack of Validation Processes. Theoretically, ecological design can help to increase the performance of built landscapes, benefiting both humans and nature. In real life, however, there is not enough evidence to justify these effects. More independent studies that help to validate the cost benefits, as well as case studies of successful projects should be conducted to provide adequate evidence (Calkins, 2005; Cassidy, 2003).

Lack of Support. Ecological design can lead to increased costs in the process of design, construction, and maintenance. The rise in the price, as well as the lack of evidence to prove the benefits, resulting in many stakeholders and common (non expert) people viewing ecological designs with a suspicious attitude, and thus unwilling to invest in these projects (Calkins, 2005). To promote the market interest, better communication needs to be made so the potential clients will have a better understanding of the long-term reward of ecological designs. The market value of ecosystem services should also be provided to prove the investment is profitable.

8.4.2 Overcoming the Obstacles in Ecological Design with Smart Technologies

Many of the challenges in ecological design are caused by inefficient handling of information and lack of communication. Smart technologies described above can help to resolve these problems.

Lack of information. The first obstacle preventing the delivery of ecological data to landscape architects is that the data itself is highly fragmented. Fortunately, modern technologies have enabled the storage and organizing of big data that comes from a wide variety of sources and formats. The data can be integrated and cataloged to facilitate searching attempts. Platforms such as TreeBASE and GenBank have been developed to share ecological data (Poisot et al., 2019). These platforms also imply data visualization technologies so it is easier for non professionals to understand the presentation. As a result, ecological data is now much more accessible to those without a background in ecology.

Lack of Research. Learning about ecological principles can be a time-consuming process. The development of ICTs has provided resources such as online courses, allowing additional opportunities for education and training. Communication technologies also help to facilitate the cooperation between landscape architects and ecologists. Moreover, tools such as Planning Support Systems (PSSs) can provide services of automatic data organizing, analysis, and visualization. So landscape architects no longer need to conduct all the research, calculation, and analysis by themselves.

Lack of Validation Processes. The key to validating the effectiveness of a landscape is to keep monitoring after construction. Sensors, as well as other measures of data collection such as online comments and text mining, have provided effective ways for the designers to track the situation of the landscape continuously, and gather reflections and feedback from the visitors. The data collection technologies, combined with proper data analysis, can provide solid evidence that help to approve the benefits of ecological design.

Lack of Support. As described before, data sharing and data visualization have helped to make ecological data easily accessible to non experts. The data portals, as well as communication technologies, can promote public participation in ecological design, encouraging more people to engage in the process. This not only offers chances that designers can explain their ideas to the public, but also allows the residents' expectations of the landscapes to be heard by the designers.

8.5 THE ROLE OF INDIVIDUALS IN SMART LANDSCAPES

While the design, construction, and management of smart landscapes will involve experts and professionals from different disciplines, the majority of the visitors to these landscapes are expected to be non-expert people living within or nearby the landscapes. Their involvement will be vital for any designed landscape to succeed. In my opinion, the public plays at least three roles in the smart landscape framework: the client, the decision-maker, and the evaluator.

Client. The majority of visitors in a landscape is likely to be constituted by common folks, especially local denizens. As a result, they should be considered as part of the clients of smart landscape project, regardless of the names on the project contract. In other words, a smart landscape needs to be designed *for* the residents nearby, or anyone who will visit the landscape and benefit from it, aiming to provide services that will meet their expectations.

Decision Maker. A recent trend in landscape architecture is to encourage community engagement in the design process. As early as the 1990s, landscape architects had started to challenge the top-down design and planning process by inviting more community members to the decision-making process (Hester Jr, 1999). Common people can review the design ideas from the user's perspective, as well as provide valuable information of the site that can only be gathered in daily lives. Comparing with traditional design processes, the smart landscape approach is science- and data-intensive. This can bring a challenge to public engagement, for the design decisions and the reasoning behind them can be difficult to understand by non-experts. To solve this problem, data, models, and theories need to be translated into a form that can be interpreted by common people without obstacles. Luckily, the implication of ICTs can facilitate the communication between the designers and the community (Ruggeri & Young, 2016), offering platforms in which large groups of common people can join the decision-making process without difficulty. In other words, a smart landscape must be *comprehensible* to encourage public participation.

Evaluator. A smart landscape is dynamic and evolves over time. It needs to be adjusted and improved based on its performance. While common people can make up the majority of visitors to a landscape, their satisfaction shall be taken as a primary indicator when evaluating the performance of the landscape. Feedbacks from the visitors can be accessed from sources such as online comments. Surveys and text-mining can also be carried out when more specific information is needed.

8.6 LAST THOUGHTS

The development of smart landscape concepts is closely related to my intellectual experiences. Before I started my doctoral study, I have completed an undergraduate study in biological sciences and a Master of Landscape Architecture. When I started my master's study, I experienced a "cultural shock", for the philosophy of designing bear few resemblances to scientific approaches I was familiar with during my undergraduate study. While the instructors encouraged me to exert my biological knowledge and “bring science to the class”, I found interdisciplinary cooperation can be challenging sometimes. Participants need to overcome the gaps in knowledge, experience, perspectives, or even languages to communicate efficiently. Despite not having a name of “smart landscapes” yet, my initial thoughts started to emerge that philosophies and methods needed to be developed to facilitate the application of ecological principles in landscape design.

Fortunately, I have been working in an interdisciplinary team during my doctoral study, and thus have accumulated experiences in communicating with people from different majors. I found that an interdisciplinary communication can be much more effective when an expert is willing to accommodate the non experts, translating professional knowledge into easy to understand language. Under an opposite situation, when everyone talks in his/her own language and leaves the others to interpret by themselves, communication will be nearly impractical. Therefore, in the case of ecological design, it is the duty of ecologists to express their ideas in a way that is easily understandable to designers. The developments in ICTs have brought new, highly effective tools to facilitate communication and data sharing. With proper implications of these tools, it will be much easier for landscape architects to adopt ecological knowledge, theories, principles, and models in their design process.

One limitation of this dissertation is that while the feedback loop of the smart framework is proposed, it has not been tested in the real world. Research projects in this dissertation have reached the stage of providing suggestions to design decision making, but not further. Therefore, the feedback part of the framework becomes an expected extension of my study. In the coming years, I hope I will be able to participate in real world landscape design projects, monitor the performance of constructed landscapes, interpret the landscape dynamics, and provide suggestions to management policies in addition to design. This will fill the last piece of the conceptual framework.

Similar to smart cities, smart landscape is a comprehensive concept, and the development of which is far beyond the capability of one single person. I wish ecologists, data scientists, landscape designers, and professionals from other fields will adapt the smart landscape concept and participate in its development. I also wish this dissertation can serve as a foundation stone of a new field.

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