URBAN FORESTS AS SOCIAL-ECOLOGICAL SYSTEMS:

THE ROLE OF COLLECTIVE ACTION AND INSTITUTIONS IN SUSTAINABLE URBAN FOREST MANAGEMENT

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This dissertation is dedicated to my family for their love and endless support, and to Elinor Ostrom, for her inspiration.

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FOREST MANAGEMENT

This dissertation portfolio addresses the question: How do institutions and collective action facilitate sustainable urban forest management (UFM)? The research utilizes mixed methods to address this question through case studies in Bloomington and Indianapolis, Indiana, U.S.A. First, the need for institutional analysis in urban ecosystem research is established by arguing that decline of urban forests is related to lack of UFM investment (e.g., free-riding) due to the non-excludable nature of the resource. Such dilemmas are resolved through institutions that adjust individuals' incentives to invest in collective good. The Institutional Analysis and Development (IAD) Framework, a precursor to the Social-Ecological System (SES) framework, is demonstrated for examining the role of institutions in UFM. Secondly, we consider the influence of municipal zoning institutions on canopy cover in Bloomington, finding that high-density residential zones are more like commercial zones than other residential zones in terms of canopy, and that mixed-use zoning is associated with intermediate canopy cover. This refines widely-held theory that residential lands have the most canopy cover and implies that canopy is driven in part by institutions that regulate impervious cover, suggesting bureaucrats consider fine-scale zoning distribution in UFM. Next, we consider collective action and institutions in the survival and growth of planted trees and community cohesion in Indianapolis. We find trees in neighborhoods that collectively water are more likely to survive than those in neighborhoods that

assign watering of individual trees to individual residents. Subsequent collective action is more likely in neighborhoods that collectively water, and institutions such as signed watering agreements and monitoring of watering improve tree establishment. This supports the application of collective action theory and institutional design principles to UFM. Finally, we model parcel-scale tree structure in Bloomington home-owner and neighborhood associations, finding significant differences in parcels by association type and development age, and demonstrating the significant influence of institutions on tree species diversity. This research utilizes the SES framework, extending its use to urban forests, and reinforces the significance of institutional analysis in urban ecosystems research. Practically, the research suggests that community association rules play an important role in structuring parcel-scale urban forests.

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Chapter One

INTRODUCTION AND BACKGROUND

I.   Introduction

The current speed and magnitude of urban expansion is the greatest in history, and by 2050, an estimated 3 billion additional people will live in cities (UNCOB, 2012). While urbanization is frequently linked to environmental degradation, it also offers solutions for global sustainability if cities can move from *consumers* to *generators* of ecosystem services, according to the 2012 draft report, *Cities and Biodiversity Outlook*. A major focus of this report, commissioned by the U.N.'s Convention on Biological Diversity, is sustainable urban forest management, underscoring that trees are one of the most prolific sources of ecosystem services in cities. Indeed, in the conterminous United States, for example, urban trees maintain a gross carbon sequestration rate of 22.8 million tC/yr ($460 million/year value) and annually remove 711,000 metric tons of air pollutants ($3.8 billion value) (Nowak and Crane, 2002; Nowak *et al.*, 2006). And yet, despite such evidence indicating their role in both livable cities and global sustainability, urban forests are on the decline; in the U.S., 4 million trees are lost annually while the average city gains 2.8% impervious cover per year (Nowak and Greenfield, 2012).

Not surprisingly then, a recent study by Wolf and Kruger (2010) determined that urban forestry professionals, academics, and agency-based managers desire a policy-based research agenda to help determine adequate regulations for sustaining urban forests. Moreover, institutional (policy) analysis is now considered a key focus for research in urbanization and human dimensions of global environmental change (IHDP, 2010). Even U.N. Secretary General, Ban Ki-moon warns that many of the Millennium Development Goals, the Aichi Targets of the Convention on Biological Diversity, and other related goals are unlikely to be met without sufficient attention from policy-makers to the implications of an urbanizing world (UNCOB, 2012).

To contribute to this end, the focus of this dissertation, composed of six chapters, is the role of collective action and institutions in sustainable urban forest management. With a focus on urban forests as social-ecological systems (SES), this research addresses the overarching question: Across scales, how do institutions and collective action facilitate sustainable urban forest

management? This question is subdivided into the following research questions, each addressed in a respective chapter:

1. What is the current state of institutional analysis in urban ecosystems research? What can New Institutional Economics and social-ecological systems research contribute to institutional analysis in urban forest management research?

2. Across a city, what is the relationship between zoning institutions and urban tree canopy cover? Do these relationships change between aggregated and disaggregated zone typologies and how might this knowledge improve the sustainability of urban forest management?

3. Across neighborhoods, how does planted tree survival and growth vary by community watering strategy and to what extent does watering strategy influence neighborhood collective action?

4. Across parcels, are there significant differences in urban forest structure and social-ecological system characteristics by community associations? What is the relative influence of these SES factors, particularly social institutions, on tree species richness of private residential parcels?

This chapter briefly introduces the study locations, and explores the concepts, including pertinent theory, used in the subsequent chapters to answer the research questions listed above. Chapter 2 follows and reviews in greater depth the current state of institutional analysis in urban ecosystems research and establishes the core framework and theory for this research (thus the brevity of Chapter 1). Chapter 3 addresses empirically the influence of municipal policy on city-wide canopy cover. Zooming into a finer scale, Chapter 4 addresses the role of institutions at the neighborhood scale in sustainable community tree planting and management. Chapter 5 explores the role of institutions in urban forest management at the finest scale, the private, residential parcel. Finally, Chapter 6 summarizes findings and implications, and addresses future research.

II.   Study locations

This dissertation research is centered in Bloomington and Indianapolis, Indiana, U.S.A. (Figure 1). The state of Indiana is centrally located in the US; the northern portion of the state has been

glaciated, most recently by Wisconsian Glaciations resulting in topography of low relief, composed largely of wetlands until drained for agricultural purposes in the past century. South of the glacial maxima are areas of higher relief composed of Central Hardwood Forests. The city of Bloomington is located within the south central portion of the state in the Interior Plateau. Indianapolis, the state's capital and the largest city is located 50 miles northeast of Bloomington in the Loamy-high lime till plains (Griffith and Omernik, 2008).

Indiana is ranked 15th among states in population with just over 6.3 million residents (US Census, 2008); the state's population is projected to increase through 2040 and most of this growth will occur in counties with urban centers particularly in the metropolitan area surrounding Indianapolis (Indiana Business Research Center, 2008). Additionally, Bloomington is representative of the small and medium-sized cities expected to be the location of the greatest population growth in coming decades (UNCOB, 2012). Sixty-six of the state's cities are Tree City USAs, including Indianapolis and Bloomington; these cities meet the Arbor Day Foundation's definition of sustainable urban forest management (Arbor Day Foundation, 2010). However, recent analysis detected that of the U.S. Forest Service's Northeastern area states, Indiana has the lowest proportion of urban

Figure 1. Study locations of Bloomington and Indianapolis, Indiana, U.S.A.

land in tree canopy cover (22.3%) while maintaining a relatively high proportion of urban land in impervious cover (25.5%) (Shifley, 2012). With increasing populations and development, Bloomington and Indianapolis represent important Indiana cities in which to examine the role of various institutions and community strategies in managing the urban forest.

III.   A review of relevant concepts for this dissertation

Defining urban and community forests and their sustainability

The definition of "urban" and "community" are important concepts for urban forest management research; both can be defined utilizing concepts from the U.S. Census Bureau (Shifley, 2012). While the concept of "urban" may need to be broadened in some research scenarios, here it is meant as all the territory, population, and housing units located within urbanized areas of at least 1000 people per square mile (U.S. Census Bureau, 2007). Communities are geopolitical boundaries including cities, towns, or unincorporated areas including neighborhoods (U.S. Census Bureau, 2000). Urban and community forests are, then, all the trees within urban and community lands, including remnant or emergent wooded areas, as well as planted trees along streets, in parks and in lawns. Some of these trees are actively managed, while many are not; however, all are subject to the active and passive decisions of humans (Zipperer *et al*., 1997). Therefore, urban forests constitute social-ecological systems (SESs), ecological systems intricately linked with and affected by one or more social systems (Anderies *et al*., 2004).

The importance of sustainability associated with urban forests and urbanization is related both to decreasing trees within urban and community areas, but also the decline of rural forests and agriculture lands as urban areas expand. Elmqvist (2012) reports that, globally, the area consumed by urbanization in the next 40 years will be equal to an area three times the size of France. As urbanization displaces other functional lands which provide ecosystem services, including rural forests, the importance of maintaining urban trees to offset forest loss elsewhere is compounded. This distinction is important to ground the focus of this dissertation research in urban forest sustainability but to acknowledge the role of urbanization in influencing non-urban areas.

Sustainable urban forest management is related to the concepts of resilience and robustness, the latter being significant for SESs, in particular (Anderies *et al*., 2004). Robustness is the ability of a system to maintain important functions in the face of external, unpredictable perturbations, or when there is uncertainty about the values of internal design parameters (Carlson and Doyle, 2002). Urban forests are SESs and prime examples of systems that face internal (and emergent),

as well as external disruptions (Wu, 2008), but little empirical research addresses the characteristics of robust urban forest SESs. In fact, it is even difficult to operationalize the broader term of sustainability to urban forests. Shifley (2012) argues that the Montréal Process Criteria and Indicators utilized for temperate and boreal forest sustainability could be applied to urban forests, but much of the data needed are not available, including the links between urban forest structure and function, and the legal, institutional and policy frameworks for sustainable management (Montréal Process Criteria 7).

Nonetheless, theoretical definitions of sustainable urban forest management target the provision of ecosystem services over space and time as well as the community of users and their management strategies (Clark *et al.*, 1997; Dwyer, 2003). Because ecosystem services are provided via functioning urban forests, and a loss in urban forest structure leads to a loss in function (Nowak *et al.*, 2008), sustainable urban forests are managed for robust structure that does not forfeit long-term or distributional provision of ecosystem services. In other words, sustainable delivery of services requires maintenance of structure, the system's natural capital, in the face of disturbance (McPherson, 2006; Lant *et al.*, 2008).

The "ecology *of* cities" approach defines humans not as the sole source of disturbance to urban vegetation but as endogenous components of the ecology that can influence the emergent and dynamic processes of the urban ecosystem and therefore, manage for sustainability (Wu, 2008). Thus, sustainable management of urban forests implicates humans and their institutions as the source of "use" and "decisions" which influence structure. Moreover, the complex matrix of property division and "agents of management" in urban environments implicates the role of collective decision-making and action. Therefore, urban forest sustainability cannot be understood without further research that considers the temporal and spatial distribution of the community of users and their institutions, as well as the biophysical resource system and its flow of goods and services.

The biophysical influence on sustainability of urban forests

Clark *et al.* (1997) and Kenney *et al.* (2011) have defined sustainable structure of urban forests considering relative age distribution, species diversity, canopy cover, and native vegetation. Urban forest management has borrowed from forest ecology and management the concept of an

uneven age mix—a larger population of young trees which tapers to a smaller population of older trees—ensuring that canopy cover remains relatively constant over time (Clark *et al.*, 1997). It has been suggested that the approximate sustainable number of trees to be planted each year for a given species can be determined by dividing the total number of trees expected, when the urban forest is fully stocked, by the maximum age at removal (Thompson *et al.*, 1994). However, age-distribution metrics require basic information regarding the longevity of species in heterogeneous urban environments and lack of this data stifles the ability to ascertain optimal age distributions to operationalize this metric (Personal communication with US Forest Service Ecologist, Paula Peper, 9 Sept. 2012).

Species diversity is a criterion for sustainability as "experience with species-specific pests has shown the folly of depending upon one species" (Clark *et al.* 1997, 22). Dutch elm disease killed millions of elms, the primary street tree across much of the eastern US (Raupp *et al.*, 2006). Invasive pests directly harm common urban tree species like ash (*Fraxinus* sp.) in the case of the Emerald Ash Borer (*Agrilus planipennis*) which was recently confirmed in the study location of Bloomington, Indiana and will surely require the removal of the majority of ash in the city, including those that compose 6% of the street tree population (Fischer *et al.*, 2007). Barker (1975) was the first to suggest diversifying street trees for sustainability, suggesting that no particular kind of tree should exceed 5% of an entire tree population's density; Miller and Miller (1991), realizing the difficulty in supplying proven plant material, eased the goal to no more than 10% of one species; Santamour (1990) coined the "10-20-30 rule," that no more than 10% of one species, 20% of one genus, and 30% of one family compose a city's urban forest population.

Urban forest researchers have come to emphasize the importance of maintaining climate appropriate canopy cover in order to provide sustainable ecosystem services. Canopy cover through leaf area is directly related to a number of ecosystem services: air pollutant removal, stormwater runoff mitigation, and building energy conservation (Walton *et al.*, 2008). The ideal amount of canopy cover will vary by climate and region and by location within a city (Clark *et al.*, 1997) thus American Forests (2009) recommends an average 40% tree canopy across any city east of the Mississippi and in the Pacific Northwest; more specifically, they recommend 50% in suburban residential zones, 25% in urban residential, and 15% in central business

districts to maintain a sustainable flow of ecosystem services. However, it appears these specific recommendations are not based in research.

Native vegetation is a fourth major consideration for sustainable urban forest structure (Clark *et al.*, 1997). Non-native species have the potential to negatively impact urban forest structure through competition for limited resources if they become invasive. Evidence for avoiding non-natives comes from New Orleans where recent studies have shown that native trees are more resilient to disturbance than non-natives (Kollin, 2008). However, Zipperer *et al.* (1997) and Kenney *et al.* (2011) argue that the urban environment may be more hospitable to non-native species given its relatively harsh conditions. If non-natives are not invasive and return ecosystem services like natives, it is worth considering their niche in urban forests.

The human impact on the sustainability of urban forests

Because urban forest systems are SESs, the importance of management in sustaining them is generally understood. In fact, urban forest management researchers Clark *et al.* (1997) and Dwyer *et al.* (2003) include community as an integral factor in sustainable management of urban forests. Likewise, urban ecologists argue that humans at various scales from neighborhoods to governments are part of urban ecosystems and integral to sustaining them (Grove, 2009). This suggests that lessons from SES research can be extended to urban forest management research to test theories related to the characteristics of the community of users and the governance of the resource relating to its sustainability.

However, a lack of association between disciplines has hindered much extension of empirically relevant SES research to urban ecology and urban forest management research, particularly in terms of the mechanisms that facilitate sustainable collective action and institutions, likely due to differences in terminology. For instance, Clark *et al.* (1997) list several groups as members of the community framework that must be engaged in sustainable urban forest management, designating goals for each: Public agencies must cooperate; large private landholders must support management goals; green industries must operate with high professional standards and support city-wide goals; and at the neighborhood level, citizens must understand and participate in management. Among all these groups, a sense of trees as a community resource and willingness to work together are paramount to sustainability according to Clark *et al.* (1997). A

scholar of SES literature would recognize that these points speak to the *salience* of the resource and the need for *polycentric management* in complex resource systems, concepts espoused by such scholars including Ostrom (1990, 2005) for sustainable resource management. The apparent lack of common language likely hinders cross-disciplinary fertilization, which could further add to work that addresses mechanisms that contribute to or detract from sustainable outcomes in urban forest management.

To this end, urban ecologists have taken steps to help define how socio-economic and demographic characteristics co-vary with and influence biophysical outcomes in urban ecosystems, and how spatial patterns affect and are affected by the human-built environment and social processes (Cadennaso *et al*., 2006). For instance, in Baltimore, MD, Grove *et al*. (2006) found a positive association between neighborhood development age and tree abundance, while in Phoenix, AZ, Hope *et al*. (2003) and Martin *et al*. (2004) found a negative relationship between these factors. Hope *et al*. (2003) has also found a link between family income and variation in perennial plant diversity hypothesizing a "luxury effect." Grove *et al*. (2006) suggested an additional mechanism—that homeowners are likely to maintain landscapes similar to their neighbors' to maintain social status. Conway (2009) and Conway and Hackworth (2007) considering the biophysical constraints on urban trees found that compact and mixed-use design principles in cities do not support more vegetation than conventional design.

These approaches advance understanding of the link between the human context and urban vegetation outcomes. And yet, we understand very little about the dynamics of urban ecosystem services (and implicitly, structure) and human preferences over space and time (Grove, 2009). Little research, nor current applications of the Human Ecosystem Framework utilized by urban ecologists, focuses on institutions as a link for understanding what incentivizes management decisions and actions that produce outcomes. Rather, institutions are one factor included in a myriad of influential factors connecting humans and their decisions to the biophysical world. Not surprisingly, characteristics of institutions that lead to sustainable systems have not been explored in depth in either urban forest management or urban ecology literature, thus, recent calls for institutional analysis from Wolf and Kruger (2010), IHDP (2010), and the UNCOB (2012).

The implications of the nature of urban forests as common pool resources

An explicit focus on institutions and collective action is necessary to understand what characterizes sustainable urban forest management in urban SESs because urban forests exhibit characteristics of common pool resources (CPRs) as demonstrated by Loeb (1987), Lant *et al*. (2008), and Fischer and Steed (2008) and their ecosystem services constitute public goods. Without proper management, CPRs are subject to deterioration by their finite nature and difficulty in excluding users (McKean, 2000). This perspective has the potential to uncover significant reasons for the general deterioration of urban forest structure, and explain why some institutions can, despite the vulnerable nature of this resource, help sustain the urban forest.

In fact, the susceptibility of urban forests or any CPR to unsustainable use is not a lost cause, despite Hardin's (1968) argument that CPRs would inevitably be destroyed if not privatized or controlled by government because of the rationally selfish nature of individuals to take from a resource for short term benefits despite long term costs (destruction of the resource). Ostrom (1990) countered that Hardin's was a specific open-access scenario in which resource users had no affiliation with one another, did not communicate, and thus could not collectively arrange socially equitable and biophysically sustainable resource use via institutions. Challenging theory of rational choice human behavior, Ostrom (2005) argued for behavioral theory and has empirically demonstrated that individuals act boundedly rational and fallible, pursue multiple goals for themselves and others, adopt relevant norms of behavior, and adapt from lessons learned over time, and therefore, are capable of creating community governance that facilitates sustainable resource management. Therefore, institutions are tools that "change incentives to enable fallible humans to overcome social dilemmas" (Ostrom, 2005: 125) such as the unsustainable use or management of a resource.

Thus, a better understanding of the characteristics of sustainable management may come to urban forest management via empirical lessons from the study of rural forest communities and other SESs embedded in CPRs. A number of characteristics or design principles were noted by Ostrom (1990) as characteristic of any sustainable natural resource management regime; their application to urban forest management has been limited, used only by Benvie (2005) to our knowledge. Moreover, Ostrom's SES framework (2009) established a diagnostic tool to understand variables

of the resource system, the resource units, the users, and the governance systems to facilitate sustainable SESs, and yet, this framework has not been applied to urban systems, either.

Related to the SES framework and arguably a foundation for it is the IAD Framework first described by Kiser and Ostrom (1982); this framework established the significance of the biophysical condition or nature of a resource, the attributes of a community, and the influential rules in establishing an action situation in which decisions are made and from which, outcomes are produced. Moreover, this framework implicates specific types of rules that one must explore in institutional analysis: operational rules, collective choice rules, and constitutional rules. At the operational level are day-to-day activities that directly affect the world. The collective choice level is where decision-makers create rules that impact operational activities. Decision-makers at the constitutional level determine the collective choice process.

The link between the IAD framework and the SES framework is relatively clear, but interestingly, widely accepted models of sustainable urban forest management by Clark *et al*. (1997) and Kenney *et al*. (2011) reviewed above implicate the same first tier variables in determining how resource users impact sustainability. In fact, all three frameworks characterize the broadest factors affecting resource management to be 1) the biophysical resource, 2) the community resource, and 3) governance which encompasses institutions that affect management decisions (Table 1). Linking these frameworks reveals the relevance of applying SES and CPR theory to better understand the factors that may influence sustainable urban forest management.

Table 1. The table links the major factors affecting management and decision-making in the IAD and SES framework with Clark *et al*.'s 1997 model for sustainable UFM.

IAD Framework Kiser and Ostrom (1982)	SES Framework Ostrom (2009)	Sustainable UFM Framework Clark *et al.* (1997)
Biophysical condition of the resource	Resource system AND Resource unit	Vegetative resource
Community attributes	Users	Community framework
Institutional rules-in-use	Governance system	Resource Management

IV.   A summary of chapters

The literature reviewed here is relevant for the following chapters of this dissertation addressing the overarching question: Across scales, how do institutions and collective action facilitate sustainable urban forest management? Each empirical chapter utilizes a data set that links some form of social-institutional analysis with some form of biophysical analysis to draw conclusions. According to Moran (2005), integrative biophysical-social science must be undertaken if case studies are to be broadly useful to the global change research community. Such is the purpose of this dissertation for informing that research agenda related to urban forest sustainability.

In chapter two, we theoretically demonstrate in greater depth that despite advances in urban ecology that include humans and their institutions in urban ecosystems research, methodologies for capturing and analyzing the role of *institutions* have not been as well developed in *urban* relative to *rural* SESs. This is a disconcerting gap given the overwhelming significance of institutions to sustainable management of rural community forests (Gibson *et al*., 2000) which are akin to urban forests (Nilsson *et al*., 1999). Likewise, resource managers and policymakers are likely to assert misguided institutions that do more harm than good without a coherent understanding of the nature of the resource (Ostrom *et al*., 1994). Subsequently, we suggest the Institutional Analysis and Development (IAD) framework (Kiser and Ostrom, 1982), the precursor to the SES framework (Ostrom, 2009) to structure institutional analysis in urban ecosystems research and specifically, urban forest management research. We undertake this objective by first, demonstrating the variations in definitions and analytical approaches to institutions in urban ecosystems research to make the case that additional structure is necessary for cumulative understanding of their role in sustainable urban ecosystems. Secondly, we review the literature regarding institutional analysis in rural SESs, including community forest management. Finally, by extension, we demonstrate the utility of structuring institutional analysis of urban ecosystems to identify institutional characteristics that facilitate or deter sustainable management.

In the third chapter, we empirically address municipal zoning institutions and their interpretation at various scales as they relate to existing and potential canopy cover in Bloomington, Indiana. We espouse that land use is a primary factor in determining the distribution of tree canopy cover

in urban environments (Nowak *et al.*, 1996) and that municipalities influence land use through urban planning efforts that include zoning institutions (Zhu and Zhang, 2008). We challenge the common conclusions of urban forest management literature that park and residential lands have the highest tree canopy cover among different land uses (Nowak *et al.*, 1996). To achieve this, we classified land cover across the City of Bloomington utilizing 2008 NAIP imagery and ERDAS Imagine software. The classified image is utilized along with a city zoning boundaries shapefile to consider the relationship between zoning and canopy cover. Developing a more precise understanding of this relationship is significant for managing canopy cover, particularly for cities with canopy cover goals, as well as for the relatively recent institutional practice of multi-use zoning for sustainable planning strategies such as new urbanism.

In the fourth chapter, research moves to Indianapolis, Indiana with a focus on neighborhood institutions and collective action. Keep Indianapolis Beautiful (KIB), a local nonprofit, maintains a contract with the city to support urban forest management at local levels, including neighborhoods, through their NeighborWoods Program. NeighborWoods supports neighborhoods, as well as other organized entities, in the planting of urban trees throughout the city. Provided that a neighborhood presents an application and tree preservation plan to plant and manage tree watering, they are freely given at least 20 one-inch caliper trees and supported by KIB in their planting. Post-planting, neighborhoods are given autonomy to fulfill the institutions they constructed to manage watering of planted trees. KIB employees theorized that the trees planted in neighborhoods where collective watering occurs are in better condition and have higher survival rates than in neighborhoods in which watering is undertaken by individuals. Through a reinventory of planted trees and interviews with neighborhood leaders, we test this theory which clearly falls in line with the theory of collective action: Users or managers of CPRs acting independently obtain a total net benefit from a resource that is less than they could have achieved if they had coordinated their strategies in some way (Ostrom, 1990). Further, we examine the impact of collective watering on community cohesion.

The fifth chapter returns to research undertaken in Bloomington, Indiana and focuses on the influence of SES factors on tree structure at the parcel-scale within neighborhood associations (NAs) and home-owner associations (HOA). We focus on this scale because a disproportionate reliance on private, residential property owners to sustain urban forests has been noted (Kenney

et al., 2011) and households represent the finest-scale of decision-making as influenced by institutions. Moreover, NAs have been implicated for their role in the provision of sustainable management of urban forests (Clark *et al.*, 1997); in addition, HOAs, which are growing in numbers, provide a way of preserving, protecting, and enhancing community resources (Le Goix and Webster, 2006). Through household surveys, full parcel-scale tree inventories, analysis of community association by-laws, and application of the SES framework (Ostrom, 2009), we demonstrate variation in parcels by community association type and predict tree species richness through the use of second-tier SES Framework variables that represent first tier variables including biophysical, social, and institutional factors.

Finally, in chapter six, I summarize the findings of these research articles and offer implications for their collective findings. Additionally, I suggest future research plans that draw from this line of inquiry to conclude this dissertation portfolio.

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Chapter Two

STRUCTURING INSTITUTIONAL ANALYSIS FOR URBAN ECOSYSTEMS: A KEY TO SUSTAINABLE URBAN FOREST MANAGEMENT

I. Introduction

More than fifty percent of the world's population now lives in urban areas and that figure is projected to increase to sixty percent by 2025 (Wu, 2008). Increasing urban populations have resulted in tremendous pressure on urban environments including urban forests which provide important positive externalities through ecosystem services. These services—public goods— include clean air and water, energy conservation, carbon storage and sequestration, cooler and more regulated air temperatures, wildlife habitat, recreational opportunities, social, physical, and psychological well-being, and economic stimuli (McPherson *et al*., 1997; Bolund and Hunhammar, 1999; Nowak and Dwyer, 2000; Wolf, 2004, 2005; McPherson, 2006). Yet, urban growth has resulted in a decline in urban forest structure and related function in the United States (Nowak and Walton, 2005; Wu, 2008), elevating the importance of research to understand how the interactions between people, their institutions, and the biophysical environment affect the sustainability of urban natural resources. Such research facilitates the current focus on developing sustainable cities (, 2009) and is a key future focus in urbanization and human dimensions of global environmental change (IHDP, 2010).

To date there has been extensive work developing tools to evaluate coupled human-environment system dynamics in urban ecosystems. Along with comprehensive ecological assessments of vegetative and faunal populations (the ecology *in* cities approach), these methods have included survey instruments to understand how decision-making by individual landowners, non-governmental organization stakeholders and government actors at different scales affect ecological dynamics in urban environments (the ecology *of* cities approach) (Grove, 2009). This work has developed particularly robust methods for linking landowner attributes to landscape outcomes on discrete spatial partitions (Pickett *et al*., 2008), effectively applying analytical approaches from landscape ecology to urban contexts (Wu, 2008). Much of this interdisciplinary research has been accomplished by urban ecologists and social scientists; their multi-method approach has been to study urban ecosystems as coupled social-ecological systems (SESs) in which biophysical and social systems, including human institutions, interact.

However, while frameworks for analyzing human institutions have been applied to many predominantly rural SES domains (e.g. fisheries, forests, irrigation networks), there has been less work formalizing institutional dimensions for the study of social-ecological dynamics in urban systems. To be clear, there have been empirical studies of institutions –the rules and norms that influence human decision-making and action (Ostrom, 1990) — in urban systems, but a formal structure for institutional analysis has not been developed for urban ecosystem research. While urban ecology's Human Ecosystem Framework includes "institutions" as a social ordering mechanism, among its applications, the use and meaning of the term has been varied. Therefore, given the considerable significance of institutions to sustainable management of community forests (Gibson *et al.*, 2000) which are akin to urban forests (Nilsson *et al.*, 1999), an exciting opportunity exists for cross-disciplinary fertilization of institutional analysis from rural SESs to urban ecosystems.

More in-depth and structured analysis of institutions in urban SESs promises to help answer why and how individuals are incentivized to conserve or remove urban trees—an imperative question for declining urban forests. While a strong thread of policy analysis exists in urban systems, additional structure is needed for long-term and cross-site analyses to develop the depth of understanding cumulated in rural systems through structured institutional analysis. Herein, we propose a clear definition of "institutions" and a structure for institutional analysis in urban ecosystems through a general institutional framework (Ostrom, 1990, 2005). We demonstrate the use of this analytical framework by articulating the unique institutional contexts found in urban systems and present approaches for characterizing those institutions and their interactions with urban actors and resources.

II. Background

Urban forests are coupled SESs composed of biophysical components (trees and associated vegetation) and social components, and like other SESs, are complex and adaptive, involving multiple subsystems as well as being embedded in larger systems. SES sustainability is related to both ecological resilience and engineered robustness, the latter being particularly significant for urban forests because they are, in part, engineered by humans (Zipperer *et al.*, 1997; Anderies *et al.*, 2004). A robust SES is able to maintain some desired system characteristic despite changes

in the behavior of its components parts, circumstances that contribute to long-enduring or sustainable systems overall (Anderies *et al*., 2004). Dwyer *et al*. (2003) argue it is the provision of ecosystem services (public goods) over space and time that constitutes sustainable urban forests.

Sustaining ecosystem services from an urban forest means developing a robust population of trees and associated vegetation across the myriad of subsystems that compose the larger system; street trees, park trees, treed preserves and private parcels constitute some of the subsystems to be managed. Trees are one resource within these sub-systems, and can be considered a stock from which a flow of goods and ecosystem services are obtained (see Lant *et al*., 2008). Because important ecosystem services are provided via functioning urban forests, and a loss in urban forest structure leads to a loss in function (Nowak *et al*., 2008), sustainable urban forests are managed for maintenance of structure that does not forfeit long-term or distributional provision of ecosystem services (McPherson, 2006). Thus, a number of structural management challenges facing urban forest managers have been explored including maintenance of a sustainable tree population via tree relative age-distribution, native and diverse species distribution, tree condition, as well as aggregate canopy cover at the scale of interest and equitable distribution of canopy cover across spatial scales (Clark *et al*., 1997).

Because urban forests are human-engineered systems, their structural robustness lies in the actions of urban-dwelling individuals and their collective associations. Humans at various scales or "subsystems" from households to city blocks to neighborhoods to governments are part of an urban ecosystem and make decisions that are integral to sustaining it (Grove, 2009; Roy-Chowdhury *et al*., 2011). Therefore, urban forest management is polycentric and involves operational-level (or on-the-ground) actors as well as policy-makers that influence urban planning, land use, and development. For instance, city governments are frequently charged with managing "the" urban forest and its canopy cover. While governments may produce a portion of city trees, they rely heavily on tree production from individual private-property parcels which contain the majority of urban trees (Clark *et al*., 1997).

This scenario may be partly responsible for the noted decline in urban forest structure as few incentives exist for private individuals to produce public goods at levels that are socially choice.

For instance, since there is no effective market for ecosystem services, property owners often manage land and resources for outputs traded in extant markets (e.g., timber, housing development) at the expense of natural capital (e.g., trees) and the flow of functional public benefits (ecosystem services which generally have no market). Related, the incentive for actors to rely on the provision of others without contributing themselves is known as the free-rider problem. If an individual pays to plant or maintain a tree, other individuals benefit as much as the investor without incurring costs. Further, an investor, acting alone, is likely to argue that one individual's efforts may make little to no difference in "the big picture" provision of ecosystem services. However, involving other individuals to avoid this dilemma is costly in terms of transaction time and effort to facilitate collective action for the sustainable provision of services.

For solutions to these social dilemmas, coordinated strategies of action that define how a community provides for, produces, and manages public goods are required involving rules and norms among actors or resource users in a given situation (Ostrom *et al*., 1994). But for many decades, scholars argued that the appropriate institutions to solve such social dilemmas were property rights alone. More specifically, they argued for privatization (and market solutions) or government command and control, disregarding community management based on the theory that humans are exclusively rational egoists and do not collectively act for the public good as Hardin (1968) defined in the Tragedy of the Commons. But clearly, markets do not always exist where needed—there is no functional market for ecosystem services—and government command and control can fail to incentivize socially beneficial actions, particularly when the target resource setting is as complex and "institutionally thick" as an urban forest (see Hardy and Koontz, 2010). Moreover, humans do not always act as rational egoists but are boundedly rational—able to follow norms and rules and learn and adapt to collectively organize institutional arrangements of resource management (Ostrom, 2005). From this perspective, institutions "change incentives to enable fallible humans to overcome social dilemmas" (Ostrom, 2005: 125). Thus, whether resources reside under private, communal, or government owned tenure regimes, successful institutions in the forms of rules-in-use can help to establish sustainable systems (Moran, 2005). Alternatively, misguided institutions can do more harm than good unless there is a coherent understanding of the nature of the resource itself (Ostrom, 1994), a real concern in the

relatively new field of urban ecology where policy prescriptions abound (Young and Wolf, 2006).

Yet, institutions have received less attention and scrutiny in sustainable urban forest management and urban ecosystems research despite their role in linking actors and the biophysical world through human preferences, decision-making, and ultimately, action. Grove (2009) argues that we understand very little about the dynamics of urban ecosystem services and human preferences over space and time. In other words, the incentives that structure human decisions and actions thus linking community characteristics and biophysical outcomes—including the provision of sustained ecosystem services—are not well understood. Discovering why and how individuals and collective associations of individuals through various social subsystems are incentivized to conserve or remove urban trees is, in part, an institutional inquiry—distinct from a more general social inquiry—because institutions are tools that influence the incentives of humans thereby affecting their decisions and actions. Moreover, urban environments are prone to rapid change through their emergent properties in response to institutions (Benvie, 2005). Thus, to explore the sustainability of urban forests or any urban ecosystem resource, institutional analysis must be undertaken to understand the dynamics affecting resource decision-making.

III. Defining institutions

While "institutions" commonly refer to organizations or entities, the term references an entirely different meaning in several academic disciplines focused on governing forces. And although this idea of institutions dates to Giambattista Vicco's 1725 *Scienza Nuova*, consensus on the definition remains elusive (Ostrom, 2005; Hodgson, 2006). In fact, competing definitions exist both within and among different academic disciplines (see Ostrom, 2005, 178). Across disciplines, however, the majority of explanations can be placed within two approaches: institutions-as-equilibria and institutions-as-norms/rules.

Scholars that advocate an institutions-as-equilibria study how stability emerges from mutually understood actor preferences as well as optimizing behavior (Menger, 1963; von Hayak, 1945, 1967; Schotter, 1981; Riker, 1980; and Calvert, 1995). Proponents of this approach view stable behavior patterns (equilibrium) as a form of institutions. Calvert (1995) finds "there is, strictly speaking, no separate animal that we can identify as an institution…institution is just a name we

give to certain parts of certain kinds of equilibrium" (18). This approach has been critiqued by a number of new institutional economic (NIE) scholars since the definition does not account for the foundations or underpinnings of stable outcomes which tease out different levels of equilibria like "shared advice based on prudence, shared obligations based on normative judgments, and shared commitments based on rules created and enforced by a community" (Crawford and Ostrom, 1995, 583).

The institutions-as-norms/rules approach, supported by NIE scholars, argues that behavior patterns are grounded in what a society believes is both correct and improper behavior given a certain stimulus (Lewis, 1969; Ullman-Margalit, 1977; Coleman, 1988). The study of institutions should not solely be the study of equilibria but go "beyond immediate means-end relationships to analyze the shared beliefs of a group about normative obligations" (Crawford and Ostrom, 1995, 583). This approach stresses that not only norms, but rules such as public policies or laws structure institutions (Hohfeld, 1913; Commons, 1986; Sheple, 1975, 1979, 1989; Sheple and Weingast, 1984, 1987; Plott, 1986; Oakerson and Parks, 1988; North, 1986, 1990; Ostrom, 1986, 1994; V. Ostrom, 1980; Williamson, 1985; and Knight, 1992). Actions outside what a rule proscribes or requires are likely to be discouraged due to sanctions and punishments.

NIE scholars that utilize the Institutional Analysis and Development (IAD) framework originally developed by Kiser and Ostrom (1982) further distinguish between rules, norms, and shared strategies as different kinds of institutions. From this perspective, rules are institutional prescriptions for behavior which require, prohibit, or permit some action or outcome *and* include sanctions (an "or else" statement) if a rule is not abided. Rules can be as formal as law or as informal as common knowledge, so long as the rule also entails an 'or else' clause (Ostrom, 2005). The focus is more specifically on "rules-in-use" as rules followed in practice are not always the same as written rules, resulting in variable outcomes. Norms are the values an "individual places on actions or strategies in and of themselves, not as they are connected to immediate consequences" (Ostrom, 1990, 35); they are institutional prescriptions for behavior *without* defined sanctions that can be just as important as rules in constraining human behavior due to fear of social censure (Ostrom, 1990, 2005). Shared strategies are even less formal as they do not include defined sanctions *or* an operator that "forbids" or "requires" or "permits" an

action; instead, strategies are taken by actors based in part on their knowledge of norms and rules.

IV. Institutions in Urban Ecosystem Research

Within their extensive empirical work that considers biophysical qualities and social characteristics of urban communities, urban ecologists have argued that "institutions" are components of urban ecosystems (Grove, 2009); yet, given the interdisciplinary nature of urban ecology and social-ecological analysis, it is not surprising that some inconsistency exists within the field on how to incorporate this human component into frameworks for urban ecology[1] (McDonnell and Pickett, 1993). While urban ecology scholars recognize institutions are related to behavior patterns and that institutions can take both formal and informal forms, they draw largely from the institutions-as-equilibrium approach, and do not always define "rules" or "norms." This points to a difficulty in basing new institutional analysis in urban ecology on existing work, and highlights the rationale for drawing from the extensive institutional research in rural social-ecological systems and to build upon those urban studies that have utilized more structured institutional analysis.

A number of frameworks, such as the Human Ecosystem Framework (for an example, see: www.beslter.org/one_pagers/pdf/the_human_ecosystem.pdf), have emerged in urban ecology that link human and natural systems (Boyden, 1977; Blood, 1994; Pickett *et al*., 1994; Machlis *et al*., 1997; Pickett *et al*., 1997; Grove, 2009). These frameworks aim to account for connectivity issues and dynamics that govern social-ecological system outcomes. Among them, the prevalence and clarity of "institutions" varies. Those that include institutions frequently embed them as minor exogenous factors. More significantly, variation exists in the definition of institutions. For example, some scholars state that institutions are "rules of the game" and "play a key role in the adjustments or adaption of cities to changing conditions" (Grove, 2009, 287; Birch and Deluca, 1984). However, what is meant by "rules" is not clearly stated. Institutions are also commonly described broadly as "dynamic solutions to universal needs, including health, justice, faith, commerce, education, leisure, government, and sustenance" (Grove, 2009, 288;

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[1] It is important to emphasize here that institutions have been studied in urban areas in general; our point is that in-depth institutional analysis is underutilized and seldom linked to ecological outcomes in research framed as urban ecosystem or urban forest management research, in particular.

Machlis *et al*., 1997)*,* or inclusive of "kinship, economy, religion, polity, governance, and education" (Beddoe *et al*., 2009). Such broad definitions, while arguably appropriate for framing scholarship, make institutions difficult to operationally incorporate into empirical analysis.

Without a working definition of institutions, difficulty arises in conducting cumulative and comparative institutional analyses which otherwise have the potential to yield robust policy prescriptions for sustaining trees and related resources in urban ecosystems. This situation may be the reason for the relative scarcity of empirical analysis clearly linking institutional nuances with urban SES outcomes. Yet, surprisingly, Young and Wolf (2006) find urban ecology literature increasingly engaged in policy prescriptions; nearly three quarters of the papers analyzed specified the set of actors best positioned to effect change but without clarity on how institutions were evaluated within the studies.

Examples of empirical institutional research undertaken in urban ecological studies include analysis outside of the United States. In Stockholm, Sweden, urban green areas that are managed by local user groups have been studied to explain what norms and rules support ecological systems within different landscapes and governance regimes (Colding *et al*., 2006; Barthel *et al*., 2005; Ernston *et al*., 2008). Without explicit analysis of how institutions vary across governance regimes, researchers conclude that "the incorporation of locally managed lands, and their stewards and institutions, into co-management designs holds potential for improving conditions for urban biodiversity […]" (Colding *et al*., 2006, 237). Researchers in China link institutions with outcomes and have noted that where legal regulations are not enforced or understood by local people, urban forests have been destroyed (Ye, 1997 in Li *et al*., 2005). Similarly, Nagendra and Gopal (2011) infer that the lack of consistent and publicly available tree policies in Bangalore, India may be partly responsible for declining tree diversity within city parks.

Within the U.S., Long Term Ecological Research (LTER) sites situated in Phoenix, Arizona and Baltimore, Maryland, have facilitated a number of studies that link lawn management with social variables, legacies, and institutions offering contributions to understanding this previously unexplored component of the urban forest. While these studies have found institutions to be influential factors apart from more general social variables, they stop short of fully exploring them through structured institutional analysis. Larson *et al*. (2009) have examined lawn

management in Phoenix to understand how social and cultural norms and legacies impact urban landscapes. Martin *et al.* (2003) analyzed how municipal ordinances and home ownership rules shape unique grass-scapes while Jenkins (1994) identified how internalized social censure plays a role in compelling residents to conform to local lawn standards. According to Robbins and Sharpe (2003), aesthetic norms and the fear of neighborhood sanctions are among key drivers affecting front yard maintenance. Larsen and Harlan (2006) explore the power of homeowner association (HOA) rules in influencing front yard landscapes in Phoenix and predict their increasing importance as legal prescriptions for landscape management. In Baltimore, building codes that required minimum set-backs from streets and zoning laws that forbade mixed-use development were determined to be important factors that contributed to residential developments of "seas of green grass" (Boone *et al.*, 2009). Access to green areas—an issue of environmental justice—was impacted by institutional legacies in Baltimore; a neighborhood association that promoted tree planting and a covenant that reserved properties for white occupancy-only led to racially-biased access to trees within the city (Buckley, 2010).

Other researchers, through complimentary work, have argued for in-depth empirical institutional analysis in urban ecosystems. Stewardship mapping and spatial social network analysis in New York, New York is currently characterizing social groups that manage urban forests, with a focus on collective action and social networking (USDA Forest Service, 2009). In fact, Tidball *et al.* (2010) purport that post-9/11 resilience (a concept linked to sustainability) within New York City communities was obtained in part by community civic ecology *strategies*, and argue that SES resilience should be studied not only in rural communities, but in urban areas which are vital to global sustainability. Research is also being conducted in Chicago, Illinois to connect institutions of urban forest restoration groups to potential biodiversity outcomes utilizing a well-developed research program, the International Forestry Resources and Institutions (IFRI) program (see http://www.umich.edu/~ifri/) (Lynne Westphal, personal communication, March 5, 2010). In this case, researchers have made the clear connection between rural and urban institutional analysis and are working to adapt the program's protocols, developed in rural sites, to urban social-ecological systems.

Urban forest ecology and management, arguably a sub-discipline of urban ecology, recognizes the importance of institutions in managing urban trees, but bases many widely accepted policy

prescriptions on best practices espoused by certification programs such as Arbor Day Foundation's Tree City USA or the enduring "Model of Urban Forest Sustainability" (Clark *et al.*, 1997; Van Wassenaer and Kenney, 2010). For example, both define a sustainable urban forest as one governed by a municipal tree care ordinance, a formal institutional arrangement. Clearly, recognition exists that policy is an important tool for sustaining urban forests by adjusting human incentives and actions, but the rich array of institutional types and structures are not well enumerated or studied within the field. Only a handful of empirical works (McPherson, 2001; Conway and Urbani, 2007; Zhang *et al.*, 2009) consider urban tree ordinances in detail.

The IAD framework (detailed in Section VI), which we suggest is useful for urban SES institutional analysis, has rarely been applied in urban ecosystem studies although a few exceptions are notable. Benvie (2005) demonstrated that the presence of institutional design principles (see Ostrom, 1994) supported successful adaptive management via collective efforts of resource users in a degraded and declining urban water outlet channel, the Las Vegas Wash. Additionally, a recent comparative study of rural and urban collaborative watershed groups utilized the IAD framework and concluded that urbanites took advantage of the institutional nature of urban areas to strengthen and add to top-down local policies protecting the watersheds; while the rural watershed group was equally successful, they utilized more bottom-up institutional approaches, encouraging landowners to engage in voluntary institutional arrangements including conservation easements (Hardy and Koontz, 2010). This analysis found that urbanites compared to their rural counterparts incurred higher transaction costs due to the large number of urban actors and overlapping urban jurisdictions, a social dilemma that might not otherwise have been empirically uncovered without structured institutional analysis. This previous research lays the ground work upon which we build our case for further use of the IAD framework.

V. Common Institutions influencing Urban Forest Ecosystems

Clearly, urban forestry and urban ecosystem researchers have acknowledged the role of institutions in their field, thus these settings are well suited for more structured institutional analysis given their biophysical and social complexity and the important role that both formal and informal rules play in land use decisions that have the potential to produce public goods or

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"bads." Specifically, urban forests are fragmented into a multifaceted matrix of property rights and management strategies subject to a myriad of actors and their associated governance regimes. Within many cities, the majority of private property parcels and their trees are owned and managed by individuals, while some private parcels exist under shared ownership and management, and numerous public property parcels are owned and managed by public entities but often heavily used by the general public. Each of these various governance and use regimes is impacted by diverse and nested institutions given the urban context—a context that Hardy and Koontz (2010) reference as institutionally complex.

Beyond the complexity of urban property rights, there are numerous, frequently encountered institutions directly related to and impacting urban forest management at the finest scale of management, the private residential parcel (Figure 1). Municipalities design urban forest conservation codes such as the requirement to obtain a permit to remove trees (even privately-owned ones) above a certain size, the maintenance of a percentage of original canopy cover amidst private development, or tree planting requirements within the public right of way adjacent to private parcels. Besides municipal code, cities often enter into agreements with developers during planning stages that establish conservation easements or tree preservation easements, which may then become covenants of the development and can be enforced or forgotten by the city, homeowners associations or neighborhood associations. Thus, the macro-level force of development legacies, as they have been called (Larsen and Harlan, 2006), may be altered by subsequent governance forces. For instance, collective residential associations often create their own institutions in terms of by-laws that affect the species that are planted, dead tree removal, or even the acceptable height of vegetation. Moreover, utility easements and their related institutions may overlap with any one of these previously mentioned arrangements. To add even more complexity, informal institutions—norms and strategies—can be quite significant in urban forest management, often incentivizing "popular" landscaping species (also affected by tree supply businesses) or management strategies—even poor ones, including tree-topping or monoculture plantings. Green non-profits and advocacy organizations may play a role by attempting to influence strategies and norms related to tree management.
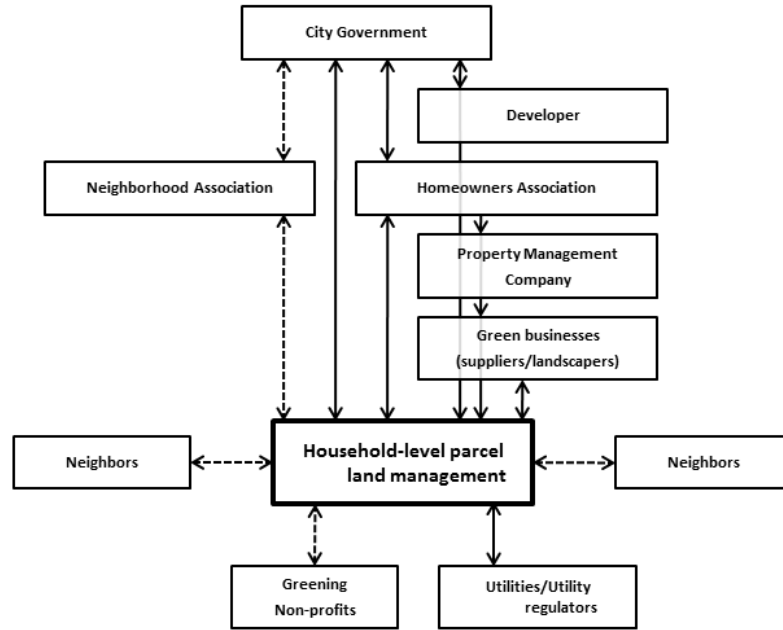
Fig. 1 Common actors and types of institutional arrangements affecting action situations related to urban household land and tree management. Solid lines represent formal institutions or rules while dashed lines represent informal institutions, or norms/strategies. Arrows indicate the primary direction of institutional influence.

Clearly, the nature of the urban forest in terms of governance and institutions is quite complex, facilitating heterogeneous outcomes at the parcel and neighborhood scale across urban landscapes. Unless a community—individual actors, associations, and governments—has established institutions that operate across scales to incentivize sustainable management of urban trees, it may struggle to influence the structure of the urban forest as a whole and its functional provision of ecosystem services. Moreover, unless urban ecosystem researchers have carefully considered institutional forces at play within a research site, their policy prescriptions have the potential to misguide solutions and do more harm than good.

VI. The Institutional Analysis and Development (IAD) Framework and lessons from its application

A wealth of institutional analyses that can be adapted to urban ecosystems research have been undertaken in *rural* social-ecological systems by building upon the IAD framework developed

by Kiser and Ostrom (1982). Through this framework, Munger (2010) finds resolution by an exact definition of institutions as rules, norms, and shared strategies that constrain human behavior, each in precise ways (as detailed in Section III). The importance of institutional analysis in sustaining natural resources is made clear from this perspective: institutions are not minor, embedded components of social systems, but, along with characteristics of the physical world and the community of resource users, are top-tier variables that structure the context in which humans make decisions and act (Ostrom, 2005). In fact, from research in numerous *rural* systems, Ostrom (1990) utilized institutional analysis to categorize institutional characteristics, or design principles, associated with long-enduring natural resource management communities that have proven robust across multiple sectors and over time (Cox *et al.*, 2010). In these systems, in-depth institutional analysis has determined how institutions shift individual incentives and actions, directly affecting sustainable outcomes.

This precise perspective of institutions within the context of the IAD framework (Figure 2) links attributes of the *physical world* (the resource system and resource units), a *community of users,* and the *rules-in-use by that community* as the exogenous variables that incentivize and constrain *actors* in an *action situation,* ultimately leading to *outcomes* (Ostrom, 2005). As suggested, an action situation is a setting where two or more individuals are faced with a set of potential actions that jointly produce outcomes; an example includes users of a natural resource managing resource units, such as a forestry agency managing a forest for timber products. "Management" is a broad term for a number of actions related to governance. Ostrom, Tiebout, and Warren (1961) and McGinnis (2011) define key tasks (or individual action situations) that indicate an effective system of governance—the *provision* for, *production* of, and *consumption* of goods/services, *financing* activities, *rule-making* and *monitoring* related to goods/services, *sanctioning* rule-breakers, *dispute resolution mechanisms, information dissemination,* and *coordination* among all relevant actors. Each of these tasks defines an individual action situation that could be modeled in the IAD framework and for which a set of rules apply.
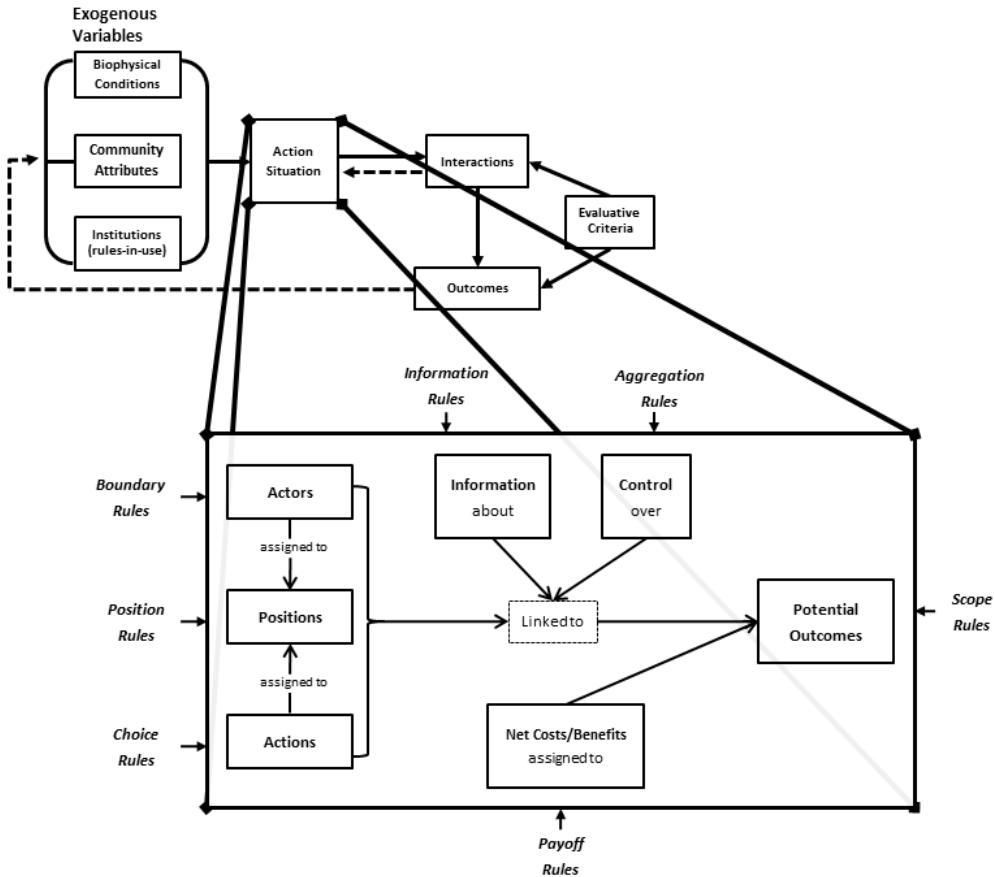
Fig. 2 The IAD framework in the background and the inner-workings of an *action situation* in the foreground. The rules influencing the action situation's component parts are listed around its boundary (Adapted, with permission, from Ostrom 2005: 189).

Within an action situation, actors (individuals or entities) are assigned to positions through which they choose to act in light of information, the control they have over action-outcome linkages, and the benefits and costs assigned to actions and outcomes (Figure 2). Each of these variables is affected by institutions. Position rules define the main role or position an actor can take (in the case of forest management, the position could be a "forest manager"), boundary rules define how actors are eligible for this position and how they enter and exit it, choice rules define the allowable actions that actors can take, and information rules affect the level of information available to actors, affecting their decisions among actions. In addition, aggregation rules define whether a single participant or multiple participants must make decisions in the case that multiple actors hold the same position; payoff rules assign rewards or sanctions to actions, and

scope rules define allowable outcomes (see further details in Ostrom, 2005). This framework clearly "…serves to remind us that each actor's preferences, as well as the choice options available to them, are determined by the institutional arrangements…" (McGinnis, 2011: 2).

This approach further defines institutional arrangements as three nested layers of rules: constitutional, collective choice, and operational rules (Ostrom, 1990). Operational rules govern day-to-day activities (for example, "The forest manager may not cut down any standing tree or else he/she must pay a fine equivalent to the value of the tree"). Collective choice rules determine how operational level rules are set or changed (for example, "The forest trustees must meet annually to review rules related to tree management; rules must be presented to trustees and voted upon utilizing a majority-rule procedure"). Likewise, constitutional rules determine how collective choice rules are set and changed. Typically, the rules in place in a situation of operational choice are assumed to be determined at the collective choice level and collective choice (or policy) actions are governed by rules at the constitutional level; however, adjacent operational action situations impact one another and can be impacted by multiple processes of collective choice or constitutional interactions (*e.g.* increased *financing* may lead to increased *production* of a good/service while both management activities can arise from the same operational level) (McGinnis, 2011). In addition, actors move between levels—if a day-to-day operational tactic is failing, actors utilize collective choice rules to make changes to operational rules. For instance, if *monitoring* of tree removal has not been effective at preserving a forest for the provision of ecosystem services (the outcome of interest), forest managers and concerned forest users (actors) may act in a collective choice arena to adjust rules defining who operationally monitors the forest and the qualification they must have (boundary rules) and the tactics they use to monitor (choice rules).

The IAD framework has been utilized empirically in a myriad of seemingly disparate settings—across fisheries, forests, and irrigation networks, for example (Ostrom *et al.*, 1994)—and through these studies, design principles (Ostrom, 1990) were built upon the IAD framework that demonstrate key characteristics that yield sustainable SESs. Many of these principles relate directly to institutions: 1) resource users must be able to communicate; 2a) boundaries of the resource and 2b) the roles of the resource users must be clear; 3a) rules governing the use of the good must fit the local needs and conditions and 3b) benefits are proportional to inputs; 4)

individuals affected by the rules can modify the rules; 5) external authorities respect the rights of users to devise rules; 6a) monitoring of the resource occurs and is undertaken by users and 6b) monitoring of the users occurs; 7) a graduated system of sanctions is used for rule breakers; and 8) a system of nested enterprises and institutions are necessary for complex resource systems (Ostrom, 1990; 2005; Cox *et al.*, 2010). Although arising largely from *rural* resource management studies, the design principles have recently been deemed useful for analysis of collective resource management in *urban* settings (Benvie, 2005).

Empirical work has confirmed the importance of these institutional characteristics and the IAD framework to research. For instance, Chhatre and Agrawal (2009) used data on 80 forest study sites in 10 countries to show that greater rule-making autonomy at the local level is associated with greater carbon storage and livelihood benefits, outcomes which facilitate sustainability. Another study found that for 220 forest communities, regular, institutionalized monitoring was significantly and positively correlated with forest condition (Gibson *et al.*, 2005). Although not in the realm of social-ecological systems research, the framework was also utilized in urban studies related to the provision of policing as a public good in Chicago and Indianapolis (Ostrom and Baugh, 1973; Ostrom *et al.*, 1974) which is not dissimilar from the concept of the provision of ecosystem services, as both are essential public goods in urban communities.

VII. Application of Institutional Analysis to Urban Forest Ecosystems

As a concrete illustration of the application of this kind of institutional analysis to urban ecosystem management, consider one sub-system of the urban forest: street trees and an operational (Figure 3) and collective choice (Figure 4) action situation affecting their managed structure, and thus, their functional provision of services. In the United States, street trees are frequently defined formally in a city's municipal code as the trees in the public right-of-way (PROW). The PROW is also generally described in city code as land beyond the surface of the road pavement to some distance measured from the center of public streets. As described above, municipal governments and adjacent private property owners are generally responsible for the maintenance of street trees in accordance with broad city goals like public safety and aesthetics, or specific aims to plant all available tree spaces. Such information is generally included in code, as well. For example, some cities require adjacent property owners to obtain a permit to remove,

plant, or prune trees in the PROW even if the responsibility of its maintenance is relegated to the adjacent property owner while its ownership is relegated to the city. These institutions in city code constitute operational level rules (Figure 3), impacting the day-to-day management of street trees.
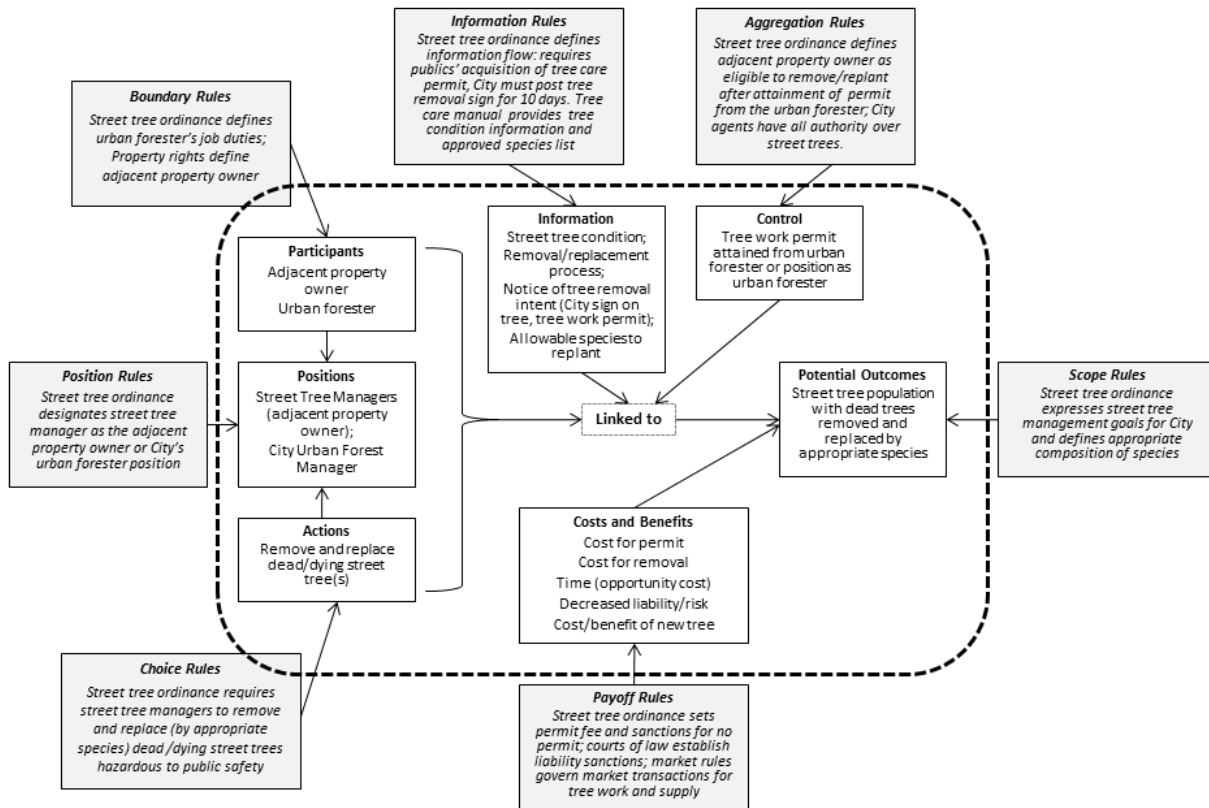
Fig. 3 The institutions impacting the inner workings of an action situation related to operational-level street tree management, specifically, street tree removal and replacement. Adjacent property owners are responsible for street trees but may only take action to remove or replace them when they are dead or dying and after attainment of a tree work permit from the City's urban forester. City foresters may remove or replace street trees but most post signage of the intent for the public unless the situation constitutes an emergency.

These operational rules are often recommended by tree commissions or boards to executive councils or administrators for enactment in accordance with collective choice (Figure 4) and constitutional rules that determine policy-making and its procedure. For example, city ordinances frequently define that a city tree board write legislation regarding operational management of

city trees, that a city council consider that legislation and take public comment regarding it, amend it as necessary, and vote to reject or pass it as city code.
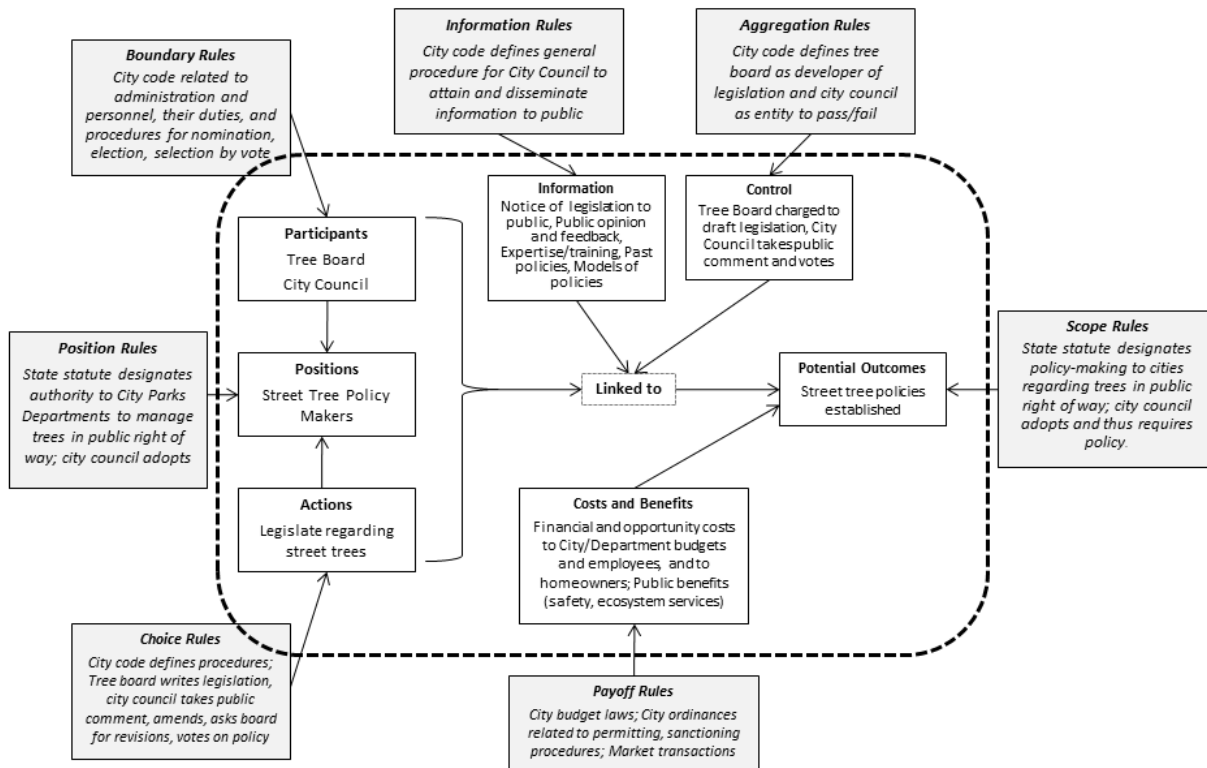
Fig. 4 The institution impacting the inner working of an action situation related to collective choice or policy-making regarding street tree management. State statute designates city parks departments (inclusive of the urban forester) as responsible for street trees in the public right-of-way (PROW). Cities generally turn over legislative development to tree boards or commissions where parks department employees hold one or more seats. The tree board is further composed of volunteer experts that develop street tree management legislation. Draft legislation is then presented to city councils for public feedback, requests for revisions, and ultimately voting.

Unfortunately, perverse incentives and unintended outcomes may result from such common urban forestry institutions meant to aid in the management of street trees and may be linked to the general decline of urban forests, causal links that may not be empirically derived without structured institutional analysis. The failed governance task in this exemplary circumstance is the *production* of healthy street trees (an action situation) which requires urban forest managers

(defined by *position rules*) to remove dead or dying trees (with a permit) and replant appropriate species (defined by *choice rules*); the main actors involved are adjacent property owners and urban foresters (*boundary rules*).

Fischer and Steed (2008) argue that the definition of street trees is often unclear to the general public, as is the role of adjacent property owners in their management; attempts to clarify the definition of a street tree, particularly by defining it in the PROW, have led to increased confusion because PROW definitions are embedded in municipal code, which is often unclear or less accessible to the general public. Thus, *information rules* impact the production of street trees, as well. Moreover, when ownership is claimed by the public municipality but management responsibilities are assigned to the adjacent property owner, increased confusion arises about which entity (adjacent property owner or city) is responsible, indicting *aggregation rules*.

If a permit is required of the adjacent property owner to manage trees in the PROW, *payoff rules* are imposed and often increase transaction costs for the actor (in this case, in terms of time and potentially money spent obtaining the permit). If only particular species of street trees are allowed to be planted, *scope rules* are imposed. If municipal code is unclear, unknown or unjustified, actors may not understand the benefits of street trees and cannot consider all the costs of not obtaining a permit to remove and plant trees (potentially including legal sanctions which are classified as *payoff rules*). Thus all benefits and costs are not weighed within the action situation, potentially resulting in inappropriate removal or planting of the wrong species of street tree in the PROW and/or the absence of a tree in a plantable space.

As mentioned, many action situations are influenced by actors within adjacent action situations, as is the case in the above scenario. Even if urban tree production arrangements are understood, outcomes are affected by adjacent action situations including *monitoring* and *sanctioning* related to street tree management. In fact, these adjacent action situations often fail to feed positively into the action of street tree production: limited urban forestry staff (as is commonly the case resulting from *financing* action situations) or uninformed neighbors may be unable to *monitor* street trees, and thus, do not *sanction* rule-breakers, adjusting their costs and benefits (*payoff rules*) in the production of street trees. Why follow a rule to plant or maintain a tree in the

PROW if there is no cost to breaking the rule, but there is a cost to following it (the cost of the tree, the cost of the time and effort to obtain a permit and to plant, etc.)?

As demonstrated through this example, institutions are important variables influencing the probability of sustainable actions in street tree management. Because street trees and their associated rules and norms are only one component of an urban forest and its many management policies and strategies, this example underscores that sustainability, as Clark *et al.* (1997) argues, can only be achieved in comprehensive urban forest management if policies and institutions are well-crafted and respected. More importantly, it points to the fact that structured, repeatable analysis is required of institutional arrangements in order for urban forestry and urban ecosystems scholars to empirically derive a deeper understanding of the roles institutions play in sustaining (or distressing) urban ecosystems and to appropriately advise on policy prescriptions.

VI. Conclusion

There has been much work to test several human ecosystem frameworks and develop a better understanding of urban ecology through institutional exploration (McIntyre *et al.*, 2000). While institutions are recognized as integral to the study of urban ecosystems, theoretical and empirical analysis has been limited in both urban ecology and urban forest management. We argue that the lack of research is not indicative of institutions' relative importance in urban ecological systems; rather, it appears to stem from a lack of clarity regarding how institutions should be defined and analyzed. This has prevented cumulative and comparative research regarding the role of institutions in sustaining urban ecosystems.

Urban ecologists and urban forest management researchers could benefit from applying a working definition of institutions that uniquely defines rules, norms, and strategies, by recognizing the nested nature of operational, collective choice, and constitutional institutions, and by applying the IAD framework to urban SESs such as urban forest systems. Although the IAD framework has not often been applied to urban SESs, it is an ideal paradigm to extend to the urban ecology field given its proven usefulness in understanding the relative role of institutions amidst social and biophysical factors in rural SES sustainability (Hardy and Koontz, 2010). Research framed in this manner has provided a foundation of institutional design principles (Ostrom, 1990) that work as hypotheses when extended to urban ecosystem research.

A recent study by Wolf and Kruger (2010) determined that urban forestry professionals, academics, and agency-based managers desire a policy based research agenda to help determine adequate policy, code, and regulations for urban forest management. By detailing the role of institutions in the sustainable management of urban forests, we hope to offer direction for analysis in urban ecosystems research that will meet this need and facilitate a key future focus—institutional analysis—in urbanization and human dimensions of global environmental change (IHDP, 2010). Opportunity exists for cross-disciplinary fertilization of institutional analysis from rural SESs to urban ecosystems and promises to help answer why and how individuals and their collective associations are incentivized to conserve or remove urban trees.

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<h1 style="text-align:center">Chapter Three</h1>

<h1 style="text-align:center">ZONING, LAND USE, AND URBAN TREE CANOPY COVER:
THE IMPORTANCE OF SCALE</h1>

I.   Introduction

As urban populations grow, the significance of urban tree canopy cover, its provision of public benefits, and the uncovering of forces that shape and change it grows for city managers. This is particularly pertinent as tree cover in urban areas of the United States is on the decline at a rate of 7900 ha yr-1 or 4.0 million trees per year (Nowak and Greenfield, 2012). The loss of urban tree canopy equates to a loss in ecosystem services including regulating and supporting services (e.g. air pollutant removal, stormwater runoff reduction, building energy conservation), provisioning services (e.g. wildlife habitat and mast), and cultural services (e.g. mental health and well-being, community cohesion, economic stimuli) (Kuo and Sullivan, 2001; de Vries *et al*., 2003; Wolf, 2005; Elmendorf, 2008; Walton *et al*., 2008). Sustainable urban forest management is the maintenance of these ecosystem services over space and time (Dwyer, 2003), and given that ecosystem services are important for sustainable cities (Grove, 2009), preserving existing canopy cover and locating areas of potential canopy cover is imperative for city managers.

Municipal governments often attempt to manage the sustainability of urban forests through institutions—policies or strategies used to organize repetitive structured interactions (Ostrom, 2005). Institutions directly aimed at sustaining urban forests are apparent in municipal urban tree canopy cover goals. Despite use of such institutions, urban canopy cover continues to decline—a collective action problem likely linked to the fact that the majority of a city's urban trees are privately owned and managed by households and sub-municipal governances often incentivized to manage land for private gain, rather than manage trees for public benefits (Clark *et al*., 1997).

In fact, land use itself is a primary factor in determining the distribution of tree canopy cover in urban environments because different land uses experience different degrees of development; not surprisingly, variation in canopy cover exists across land-use types (Hill *et al*., 2010). Thus, because municipalities establish land-use policies via zoning (Zhu and Zhang, 2008), they influence the amount of canopy cover across a city notwithstanding tree-specific ordinances

meant to directly influence canopy cover. However, little empirical analysis has tested the relationship between land-use institutions and canopy cover (Conway and Urbani, 2007; Hill *et al.*, 2010). Those studies which address this relationship are generally focused on existing canopy cover and are conducted at aggregated zoning scales, potentially drawing conclusions that fail to hold at finer scales.

This study explores the relationship between zoning and canopy cover using the city of Bloomington, Indiana as a case study. Our research specifically addresses the following questions: What is the relationship between zoning and existing, potential, and relative canopy cover? Do these relationships change between aggregated and disaggregated zone types and how might this knowledge improve the sustainability of urban forest management? To answer these questions, we first review the literature regarding municipal canopy cover goals and policies, followed by consideration for how zoning institutions impact canopy cover. Our methodology for this study is then detailed followed by results and discussion of our analysis.

Municipal urban tree canopy cover goals and policies

Many cities aim to maintain high urban tree canopy cover for the provision of ecosystem services by setting canopy cover goals. A recent study found that 38.9% of 329 cities with populations of 50,000 or greater have adopted an urban tree canopy cover goal (Krause, 2011). Frequently informing these municipal goals, American Forests (2009) has recommended cities east of the Mississippi River and in the Pacific Northwest maintain an average of at least 40% canopy cover, and underscoring the variability of canopy cover within cities, these recommendations vary by land use; suburban residential lands should maintain 50% canopy cover, urban residential, 25%, and central business districts, 15%.

Analysis of urban tree canopy cover provides the information necessary for determining canopy cover goals. While existing canopy cover (CC) is ubiquitously defined as the areal extent of canopy across a city and is generally determined through classification of remotely sensed aerial imagery, potential CC has various definitions and calculation methodologies within the literature. Traditionally, potential CC is defined as the percentage of total land area that is pervious and without tree canopy cover (Wu *et al.*, 2008). However, Wu *et al.* (2008) used GIS-based decision rules to automate the elimination of potential CC sites too small or too close to

urban infrastructure for more precise measurements. Similar methods have been used by Kenney (2008) and Kirnbauer *et al.* (2009). Overall, Kenney *et al.* (2011) argue that high-quality potential CC assessments should not only determine available plantable space, but take into consideration aboveground growing space for future canopy expansion, current and future land uses, regional climate and soils, and other key variables with the potential to affect tree growth and longevity. Kenney *et al.* (2011) also define relative CC which allows for the comparison of existing CC to maximum potential CC, in which case potential CC is composed of pervious area without tree canopy cover plus existing CC; we adopt and utilize this latter definition of potential CC and relative CC herein.

Once canopy cover metrics are established, municipalities frequently enact strategies or policies that influence urban tree planting or removal, a tactic for meeting canopy cover goals. Krause (2011) found that of the 329 cities studied, 56.1% provided education and outreach regarding privately owned trees and 74.7% had adopted a tree ordinance specifying tree planting or removal requirements for developers. Hirokawa (2011) reports that the most common tree protection ordinances require application for tree removal to determine the effect of tree loss and mitigation, offering the example of Atlanta, Georgia where a permit is required for the movement, destruction, or injury of any tree larger than 6 inches in diameter at breast height (DBH). Seemingly less common tree protection institutions directly address the effect of land use on urban trees. In Jackson, New Jersey, tree removal applications may be denied if proposed land-use activities are deemed to have a negative effect on trees (Hirokawa, 2011).

Zoning ordinances and canopy cover

Zoning ordinances, legally binding regulatory tools, are commonly used instruments for regulating land-use and intensity, building densities, and other land development issues. Typical aggregated zoning designations include residential, commercial, industrial, agricultural, and recreational zones (these are often disaggregated for land-use policies; residential zones are often composed of single-family and multi-family residential; commercial zones may be composed of downtown, arterial, or general). Parcels within a zone category are subject to zone-specific institutions that may set forth restrictions on height and size of buildings, square feet of building space, the proportion of building area per lot, minimum lot size requirements, and setback and

side-yard dimensions (SBA, 2011). Developers must comply with zoning specifications or seek permission from city planning departments for approval of zoning variances. Planned Unit Development (PUD) zones represent a zone designation through which developers and planners establish uniquely appropriate land-use regulations for individual sites that cannot be reconciled through variances of other zone designations; they have been associated with "new urbanism" which encompasses a return to traditional planning with a focus on creating walkable communities, among other goals (Duany *et al.*, 2000; Ohm and Sitkowski, 2003).

Municipal zoning ordinances may act directly or indirectly to influence urban tree canopy cover. Some municipalities utilize zoning regulations to go beyond typical land-use requirements to directly influence tree canopy cover. Hartel (2003) reports that the municipality of Chesapeake, Virginia required through its zoning ordinance the maintenance of 10% canopy cover on parcels in non-residential zones, 15% canopy cover in multi-family residential zones, and 20% canopy cover in single-family residential zones. In Athens-Clarke County, Georgia, similar zoning ordinances have been developed (for example, there the target tree canopy for residential zones is 45%) (Hartel, 2003).

 In the absence of direct regulation of canopy cover via zoning ordinances, typical zoning regulations can have indirect impacts on canopy cover by influencing land use (Hill *et al.*, 2010). Urban planning, including zoning, and preservation of undeveloped spaces (and thus, potentially treed spaces) in cities is clearly linked (Hollis and Fulton, 2002; Begnston *et al.*, 2004). Direct regulation of other land cover proportions is not atypical. For example, some communities regulate the maximum proportion of impervious land cover through zoning ordinances, policies often meant for the management of stormwater (Arnold *et al.*, 1996), but with impacts on the amount of land area with the potential for urban tree growth.

The extent to which land-use policies such as zoning actually relate to canopy cover is not well-understood and little empirical analysis has tested this relationship at multiple scales of analysis (Conway and Urbani, 2007; Hill *et al.*, 2010). Analyses are complicated by the absence of high-quality data that can be used to evaluate how land-use planning impacts land use and land cover (Bengston *et al.*, 2004). Among existing studies that contend with such difficulties, case analysis is generally confined to aggregated zoning units in relatively large communities. Nowak *et al.*

(1996) analyzed 58 US cities with a minimum population density of 386 people km-2 and found that park and residential lands, along with vacant lands in forested areas of the country, generally have the highest existing canopy cover among different land uses. However, it is not clear from the study how "land use" was derived. Similarly, Wilson *et al*. (2003) used moderate-resolution (30m x 30m) ETM+ imagery to report NDVI values (the Normalized Difference Vegetation Index; a gross measure of vegetated land cover which could be utilized to identify areas suitable for potential CC) in Indianapolis, Indiana among park and residential lands. NDVI varied significantly by aggregated zoning types (residential, commercial, etc.) as well as specific residential zoning districts (single-family residential, multi-family residential, etc.). The researchers encouraged future use of finer resolution imagery for similar analysis. Landry and Pu (2010) employed a fine resolution (4m x 4m) urban tree canopy cover classification of Tampa, Florida to determine the effects of the city's 1974 tree care ordinance on parcel-level tree canopy, incorporating zoning as a means of categorizing data analysis. They reported canopy cover for all aggregated zone types and found residential zones among those with the highest cover.

Common among such studies is the supposition or conclusion that urban communities with a higher percentage of residential, undeveloped, or parkland will have higher levels of existing tree canopy coverage. While this does not appear to be disputed among researchers, it may be an overgeneralization for planners and urban forest managers who must practically consider the relationship between tree canopy and fine-scale (rather than aggregated) land use and related policies. Moreover, it does not inform our understanding of the relationship between zoning and potential CC or relative CC. With the exception of the analysis of Wilson *et al*. (2003), few studies consider the variation of canopy cover among specific types of otherwise aggregated zones. This analysis is particularly lacking in mid-sized U.S. cities, such as our study site of Bloomington, Indiana.

II.   Methods

Study site

Bloomington is located in Monroe County, Indiana, approximately 50 miles southwest of the state capital of Indianapolis (Figure 1). As of 2010, Bloomington's population reached 80,405

individuals and 33,239 housing units, making the site fairly representative of a mid-sized (19.9 square miles), Midwestern city (US Census Bureau, 2010). The city is located within the temperate Central Hardwoods region in the Mitchell Plain and the Norman Upland Plateau physiographic regions of Indiana where topography is variable as is typical of the unglaciated portion of the state (Hill, 2011). Consequently, the region is heavily forested for Indiana, and the city itself has one of the highest proportions of canopy cover in the state at 49.7% (Heynen and Lindsey, 2003). These results, however, were based on coarse resolution AVHRR data (1 km x 1 km) which can limit the classification accuracy of heterogeneous urban environments, thus there is room for improvement in estimating the canopy cover of Bloomington using finer-resolution imagery.
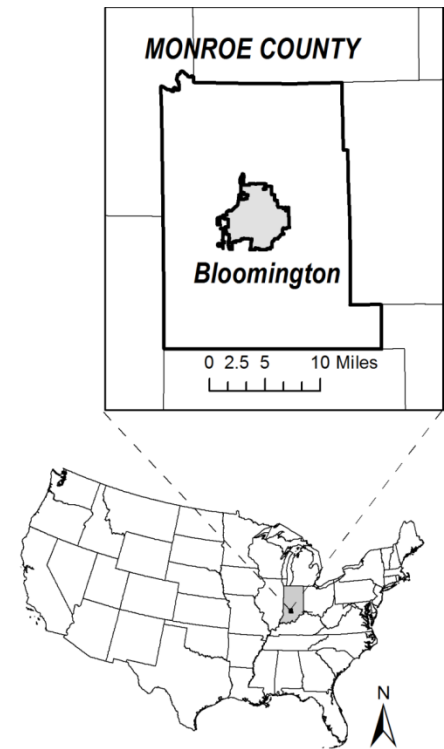
Figure 1. The study site of Bloomington, Indiana, USA.

Bloomington's tree preservation and zoning policies

The city of Bloomington is a Tree City USA as designated by the Arbor Day Foundation which requires a city to maintain a tree care ordinance among other obligations. The Tree Ordinance's stated policy goal is to maintain and increase city tree canopy, although a desired proportion of tree cover is not stated. This general policy goal is imposed upon all city departments likely to influence canopy cover as, "decisions by the board of zoning appeals, the plan commission, or the common council that impact trees subject to these provisions shall be made in accord with the policies and principles of urban forest management set forth [in the Tree Ordinance]" (Bloomington Municipal Code 12.24.0705, Section 5). In addition to the Tree Care Ordinance, the city's Unified Development Ordinance (UDO), enacted in 2007, established protection for canopy cover by requiring that all wooded lands subject to development retain a proportion of the original canopy cover during land-disturbing activities. Moreover, the ordinance required the establishment of tree preservation easements where contiguous areas of at least one-half acre of tree cover are required to be preserved (UDO, 20.07).

To understand the influence of municipal institutions on tree canopy cover in Bloomington, we conducted interviews with the city's Director of Planning, its Environmental Planner, and the Urban Forest Manager, and we examined city policies related to tree canopy preservation. Specifically, we examined the city's Unified Development Ordinance (UDO, 20.02; 20.05) and the Bloomington Tree Ordinance (Bloomington Municipal Code 12.24, Trees and Flora) to determine city-level and zoning-specific policies regarding urban tree canopy cover and land-use zoning. The extent to which the urban forest management-related policies of the Tree Ordinance and the UDO have influenced canopy cover is difficult to study at the present time given a lack of accurate geospatial information available from the city regarding the location of tree preservation sites and easements. More importantly, such research may be premature as these protection measures were introduced in 2007 and effective as of 2009, only one year prior to this study.

A more mature institution whose outcomes are likely more demonstrable on the landscape of Bloomington is the city's general zoning policies also outlined in the 2007 Unified Development Ordinance (UDO) but relatively unchanged over the last several decades. While variances have been approved and zoning designations have adapted over time with the passage of zoning ordinances in 1973 and 1996, the majority of Bloomington land has been maintained in the same functional zone over this time (personal communication with the Director of Planning, Bloomington, Indiana, 9/16/2010). The UDO defines 16 different zones, including commercial arterial, commercial downtown, residential single-family and residential multi-family, for example. These 16 zones are referenced as "zone districts" throughout the remainder of the article. These 16 zones districts can be logically aggregated into a coarser thematic scale of five "zone types," including commercial, industrial, institutional, residential, and planned unit development (PUD) (Table 1). For each zone district, multiple observations are spatially distributed throughout the city except for the commercial downtown and medical industry districts for which only one observation or conterminous area is present (Figure 2).

Table 1. Description of zoning districts and their aggregations to zone type. Zoning districts highlighted in grey were excluded from statistical operations given small sample sizes (n<5). For consistency, they were also excluded from aggregation into zone types.

Zoning districts	Zone *district* code	Zone *type* code	Maximum impervious surface[1] per parcel	Average zone size (acres)	Number of zone observations (n)	% of city area
Business Park	BP	--	60%	70.0	4	1.82
Commercial Arterial	CA	COM	60%	52.0	12	4.07
Commercial Downtown	CD	--	not applicable	294.7	1	1.92
Commercial General	CG	COM	60%	8.4	32	1.76
Commercial Limited	CL	COM	50%	2.3	25	0.37
Industrial General	IG	IND	70%	8.6	13	0.73
Institutional	IN	INST	60%	106.1	40	27.64
Medical	MD	--	60%	75.4	1	0.49
Mobile Home	MH	RES	65%	13.7	7	0.62
Planned Unit Development	PUD	PUD	varies	66.1	52	22.39
Quarry	QY	--	not applicable	28.8	4	0.75
Residential Core	RC	RES	45%	99.6	9	5.84
Residential Estate	RE	RES	15%	11.7	7	0.53
Residential High Density Multifamily	RH	RES	50%	19.1	36	4.48
Residential Multifamily	RM	RES	40%	11.2	47	3.44
Residential Single Family	RS	RES	40%	126.9	28	23.15
					318	**100.0**

[1] City of Bloomington Unified Development Ordinance: Zoning Districts. Chapter 20.02. Effective December 7, 2009.
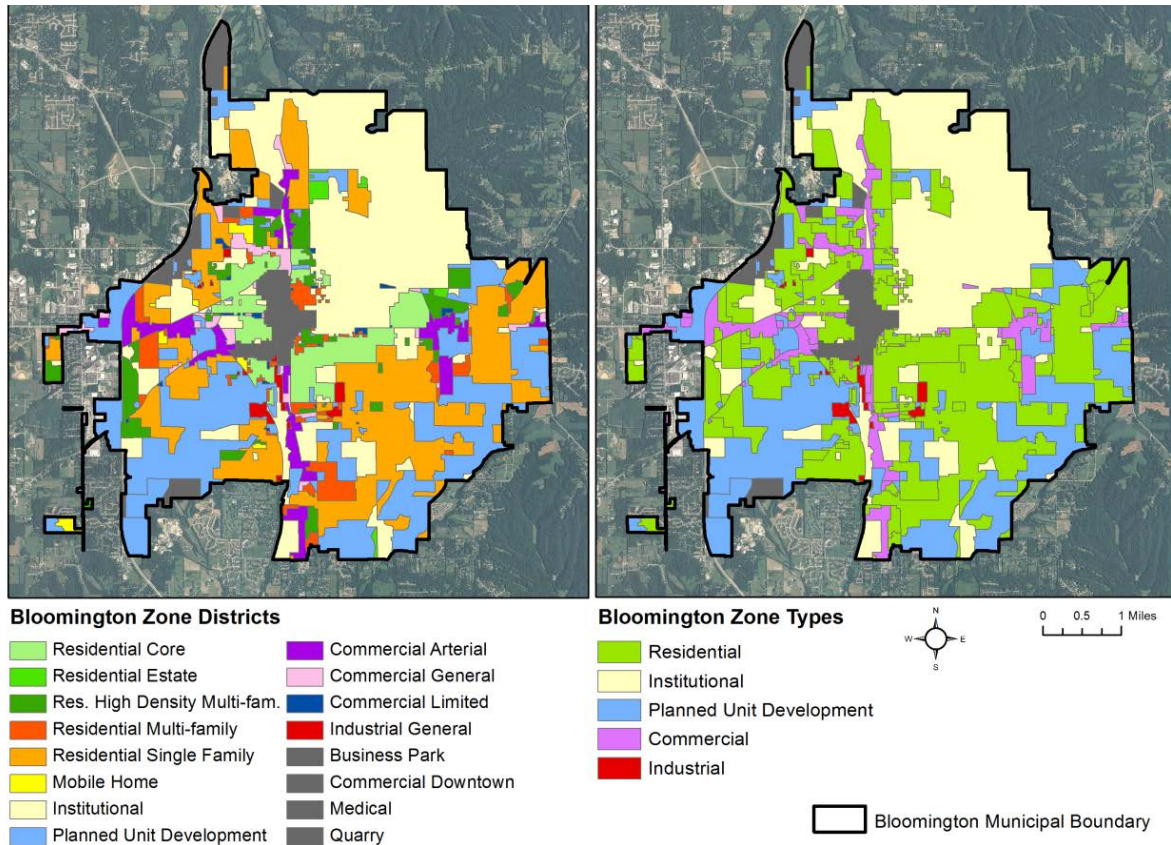
Figure 2. Aggregated zone types and disaggregated zone districts as defined by the 2007 Unified Development Ordinance for Bloomington, Indiana. Zone districts and types represented in gray were not included in analysis because of small sample sizes (n<5).

Classification of remotely sensed imagery and spatial analysis of canopy cover

To consider the influence of zoning policies on tree canopy cover in Bloomington and to offer such analysis with fine resolution data, we classified land cover for Bloomington using 2010 National Agricultural Imagery Program (NAIP) aerial photos. Photos were mosaicked without enhancements then clipped to a modified and buffered Bloomington city boundary. Boundary modifications include (1) exclusion of the noncontiguous, annexed sections of the municipality, and (2) inclusion of non-incorporated, but zoned, areas in the central and northwest portion of the city.

A hybrid decision-tree / maximum likelihood classification procedure was used to map land cover classes across Bloomington. The decision-tree classification procedure relies on manual

selection of spectral thresholds for selected bands and land cover classes (Goetz *et al*., 2003; Wilson *et al*., 2003; Yang *et al*., 2003; Lu and Weng, 2009). Decision-tree analysis was used to select optimal bands by comparing band-wise boxplots of inter-class (signature class) pixel values. An unsupervised classification (ISODATA algorithm in Erdas Imagine) of NAIP spectral bands was used to separate the landscape into three land cover categories; (1) vegetation (forest and non-forest), (2) dark land cover objects (dark impervious surfaces and structures and shadows), and (3) other land cover types (impervious surfaces, structures and soil). A maximum likelihood, supervised classification was then used to classify forest and non-forest classes from the vegetation class. Resulting classes from the maximum likelihood classification include (1) existing CC (e.g. trees / shrubs), (2) potential CC (e.g. trees and shrubs + grasses / turf), and (3) other (e.g. impervious surface, soil, water). To assess classification accuracy a traditional point-based accuracy assessment and contingency table were constructed for forest canopy and other land cover classes.

The percent existing CC, potential CC (the grass/turf area + existing CC) and relative CC (existing CC / potential CC) were calculated for each zoning district utilizing the 2007 Unified Development Ordinance (UDO) zoning polygons for Bloomington. These zone district polygons were aggregated to "zone types" for which existing, potential, and relative CC percents were calculated as well. While analysis at the parcel scale would have added additional information, city cadastral boundaries did not align well with zoning district boundaries, thus the finest scale of analysis defined canopy coverage at individual observations of zoning districts.

Statistical analysis

We utilized SPSS to conduct ANOVAs to examine differences between existing, potential, and relative CC across zone observations by zone district and zone type. No spatial autocorrelation was detected within our dataset (Moran's Index = 0.038; Z-score = 1.44 or better) and a one-sample Kolmorogov-Smirnov test of normality retained the null hypothesis of normal distributions. Levene's test of homogeneity of variance for zone districts proved variances of canopy cover proportions were significantly different, thus the conservative Dunnet's C test, measuring significance at $\alpha = 0.05$, was used for post-hoc comparisons. As stated previously, four zone districts were removed from analysis given their small sample sizes (n < 5

observations) including business park, commercial downtown, quarry, and medical zones. This resulted in 308 different zone district observations, 12 zone districts, and 5 zone types for analysis.

Results

Tree Canopy Classification

NAIP image classification accuracy was calculated based on a standard point-based accuracy assessment. In total, 283 points were used to assess the classification accuracy, including 79 points for trees / shrubs, 38 points for grasses / turf, and 166 points for 'other' (Table 2). The overall accuracy achieved equaled 93.3%, with classification errors predominantly stemming from confusion of grasses / turf with 'other' (Table 2). Of the 38 grasses / turf points, four were misclassified as 'other' resulting in a user's accuracy (UA) of 81.6%. For trees / shrubs and 'other', the UA and PA (producer's accuracy) were greater than 92% suggesting our classification adequately captured and classified the heterogeneity of the urban landscape.

Table 2. Error matrix for the classified image of Bloomington, Indiana.

Classified data	Reference data				
	Trees / Shrubs	Grasses / Turf	Other	Row total	UA%
Trees / Shrubs	73	3	1	77	94.8
Grasses / Turf	1	31	5	37	83.8
Other	5	4	160	169	94.7
Column total	79	38	166	283	
PA%	92.4	81.6	96.4		

Note: PA and UA represent producer's accuracy and user's accuracy for each land cover class.

Zone Type and Canopy

Differences in average existing, potential and relative CC at the most coarse scale of analysis, the zone type, are demonstrated in Table 3. Overall, average existing CC and potential CC followed the same pattern of highest-to-lowest canopy proportion for all zone types. Institutional and

residential had the highest proportions of these canopy metrics, followed by the PUD and industrial zone types. The commercial zone type had the lowest averages.

Table 3. Descriptive statistics for ECC, PCC and RCC by zone type (n=5). Zoning districts excluded from aggregation given their small sample size (n<5) include: BP, CD, MD, and QY.

Zone Type	N	Existing Canopy Cover (%)				Potential Canopy Cover (%)				Relative Canopy Cover (%)			
		$\bar{x}$	s	min.	max.	$\bar{x}$	s	min.	max.	$\bar{x}$	s	min.	max.
COM	69	19.1	14.0	0.0	67.3	30.4	18.3	0.6	82.2	60.3	17.7	0.0	88.5
IND	13	23.8	14.3	2.6	56.1	31.4	16.2	4.7	64.3	73.7	13.6	52.3	92.4
PUD	52	30.9	20.9	0.1	88.9	44.4	24.3	0.1	98.0	65.0	17.1	22.8	96.1
RES	134	37.8	19.6	0.5	92.3	51.4	21.9	2.6	98.6	70.5	12.6	18.5	96.7
INST	40	40.8	20.7	2.8	84.3	63.2	24.8	4.7	93.5	62.4	14.1	28.5	90.4

Note: $\bar{x}$ = sample mean; s = sample standard deviation; min. = minimum; max. = maximum

The ANOVA results demonstrated the same relative differences between zone types in terms of existing and potential CC, thus we report results only for existing CC here (Table 4). The difference in average existing CC between zone types was significant overall (F = 14.315; p = 0.000) and post-hoc comparisons showed significantly less existing CC in commercial zone types compared to all others with the exception of the industrial zone which, itself, had significantly less existing CC than the institutional and residential zones.

Table 4. Test of significance in existing canopy cover (ECC) among zone types. Values in bold indicate pairings where the mean difference exceeded critical value at α = 0.05. Table should be read from column to row.

	COM	IND	INST	PUD
IND	-4.67			
INST	**-21.76**	**-17.09**		
PUD	**-11.81**	-7.15	9.95	
RES	**-18.67**	**-14.00**	3.09	-6.86

Relative CC for zone types followed a different pattern than corresponding existing or potential CC metrics; overall, average relative CC values were much greater and less variable across zone

types (Table 3). Additionally, the industrial zone type moved from low existing/potential CC to the high relative CC, meaning these zones utilized more potential CC for actual tree canopy than other zone types. The institutional zone type fell from having high average existing/potential CC to having low average relative CC. Thus, despite having more potential CC, institutional zones used less of this area for trees.

ANOVA results for average relative CC by zone type were significant overall ($F = 6.976$, $p = 0.000$) and here, the commercial and institutional zone types had significantly less relative CC than the residential zone (Table 5). Thus, the proportion of potential CC available to these zone types is utilized for trees far less in commercial and institutional zones than residential.

Table 5. Test of significance in relative canopy cover (RCC) among zone types. Values in bold indicate pairings where the mean difference exceeded critical value at $\alpha = 0.05$. Table should be read from column to row.

	COM	IND	INST	PUD
IND	-13.34			
INST	-2.09	11.25		
PUD	-4.67	8.67	-2.58	
RES	**-10.14**	3.19	**-8.05**	-5.47

Zone Districts and Canopy

At a disaggregated scale of analysis, each zone district's existing, potential and relative CC values varied within zone districts as well as across them. Again, potential CC was higher than existing CC for all zone districts and the two metrics generally followed the same pattern from highest-to-lowest cover (Table 6). The average existing CC for the residential estate district was the highest at 78.2% while the lowest average (15.5%) was observed for the commercial arterial district. Residential estate districts also had the highest average potential CC (94.4%) while commercial arterial had the lowest (25.64%). The largest variations of existing/potential CC among district observations were found in the institutional and PUD districts.

Table 6. Descriptive statistics for ECC, PCC and RCC by zoning district. See Table 1 for a description of each zone. Zoning districts excluded from aggregation given their small sample size (n<5) include: BP, CD, MD, and QY.

Zone Type	N	Existing Canopy Cover (%)				Potential Canopy Cover (%)				Relative Canopy Cover (%)			
		$\bar{x}$	s	min.	max.	$\bar{x}$	s	min.	max.	$\bar{x}$	s	min.	max.
CA	12	15.5	9.1	2.6	26.8	25.6	13.7	4.8	40.7	58.8	9.0	44.6	74.8
CL	25	16.5	12.0	2.1	57.1	29.1	18.4	6.6	80.5	57.4	19.9	15.4	86.5
RH	36	20.8	11.0	0.5	49.9	31.8	13.1	2.6	65.9	62.4	15.9	18.5	95.0
CG	32	22.5	16.3	0.0	67.3	33.1	19.7	0.6	82.2	63.3	18.3	0.0	88.5
IG	13	23.8	14.3	2.6	56.1	31.4	16.2	4.7	64.3	73.7	13.6	52.3	92.4
PUD	52	30.9	20.9	0.1	88.9	44.4	24.3	0.1	98.0	65.0	17.1	22.8	96.1
MH	7	35.3	16.0	20.1	64.3	46.7	15.3	29.0	73.0	73.7	10.1	56.8	88.0
RM	47	37.1	17.0	4.9	91.8	49.7	18.1	10.3	94.9	72.5	9.2	48.0	96.7
RC	9	39.8	5.5	32.1	47.0	53.4	5.6	46.0	60.4	74.3	3.6	68.3	80.0
IN	40	40.8	20.7	2.8	84.3	63.2	24.8	4.7	93.5	62.4	14.1	28.5	90.4
RS	28	50.6	13.0	21.4	74.6	69.7	13.1	35.7	98.4	72.3	10.7	27.6	87.0
RE	7	78.3	11.6	60.8	92.3	93.4	6.1	80.9	98.6	83.5	8.3	72.9	93.6

Note: $\bar{x}$ = sample mean; s = sample standard deviation; min. = minimum; max. = maximum

At the zone district scale, ANOVA results determined significant differences for existing and potential CC (because they followed similar patterns, only existing CC is reported). The difference in existing CC between zone districts was significant overall (F = 16.181; p = 0.000) and post-hoc comparisons demonstrate significant differences (α = 0.05) largely between commercial and residential districts (Table 7). However, the residential high density district with low existing/potential CC was significantly different than all other residential districts (with the exception of the mobile home district) while not significantly different than any commercial, industrial, or PUD district. Average PUD CC was significantly different from two residential districts (those with the highest averages) and two commercial districts (those with the lowest averages). The institutional district, with fairly high average existing/potential CC, significantly differed from all commercial districts and the high density residential districts, all with low averages and the residential estate district with the highest average.

Table 7. Test of significance in existing canopy cover (ECC) among zone districts. Values in bold indicate pairings where the mean difference exceeded critical value at α = 0.05. Table should be read from column to row.

	CA	CG	CL	IG	IN	MH	PUD	RC	RE	RH	RM
CG	-6.93										
CL	-0.93	6.00									
IG	-8.21	-1.29	-7.30								
IN	**-25.31**	**-18.38**	**-24.39**	-17.09							
MH	-19.73	-12.80	-18.80	-11.52	5.58						
PUD	**-15.36**	-8.43	**-14.4**	-7.15	9.95	4.36					
RC	**-24.24**	**-17.31**	**-23.32**	-16.03	1.07	-4.52	-8.88				
RE	**-62.74**	**-55.81**	**-61.81**	**-54.52**	**-37.43**	**-43.01**	**-47.38**	**-38.50**			
RH	-5.21	1.70	-4.30	3.00	**20.09**	14.51	10.14	**19.03**	57.51		
RM	-21.56	-14.63	-20.63	-13.34	3.75	-1.83	-6.20	2.68	**41.17**	**-16.34**	
RS	**-35.04**	**-28.11**	**-34.11**	**-26.83**	-9.73	-15.32	**-19.69**	-10.80	**27.69**	**-29.82**	-13.48

A few variations between existing and potential CC must be noted. Namely, the industrial general district had significantly less potential CC than institutional and residential core districts although no difference was detected in existing CC (Table 7). Similarly, the institutional district had significantly more potential CC than the PUD district, and residential had significantly more potential CC than residential single family, but neither set were different in terms of existing CC. These variations demonstrate that while there may be little difference between some districts in terms of existing CC, those same districts differ in terms of their potential CC and thus their ability to obtain higher actual CC.

Relative CC averages across zone districts were higher and exhibited a smaller range compared to existing/potential CC metrics at this scale (Table 6). The lowest relative CC was obtained for a commercial general district (0.00%) and the highest for a residential multi-family district (98.69%). The lowest relative CC average was observed for the commercial limited district while residential estate had the highest.

ANOVA results determined significant differences between districts in terms of relative CC (F = 4.611, p = 0.000) (Table 8). Compared to existing/potential CC ANOVAs, the same general differences between residential and commercial districts are observed in relative CC, as is the

same exception—the relative CC of the residential high density district is significantly lower than half of the other residential districts including residential estate and residential core, while not unlike all commercial districts. Thus, the proportion of potential CC area in trees in commercial and residential high density is far less than in most residential districts. Additionally, whereas the institutional district had significantly more existing/potential CC than all commercial districts, it did not differ in terms of relative CC, suggesting institutional zones utilize relatively less potential CC for trees. Similarly, the industrial general district, which had significantly less existing/potential CC than residential districts, did not differ from them in terms of relative CC, suggesting these districts utilize relatively more potential CC for trees.

Table 8. Test of significance in relative canopy cover (RCC) among zone districts. Values in bold indicate pairings where the mean difference exceeded critical value at $\alpha = 0.05$. Table should be read from column to row.

	CA	CG	CL	IG	IN	MH	PUD	RC	RE	RH	RM
CG	-4.47										
CL	1.44	5.91									
IG	-14.89	-10.41	-16.33								
IN	-3.64	0.83	-5.08	11.25							
MH	-14.89	-10.42	-16.33	-0.00	-11.25						
PUD	-6.22	-1.75	-7.66	8.66	-2.58	8.67					
RC	**-15.49**	-11.01	**-16.93**	-0.60	**-11.84**	-0.59	-9.26				
RE	**-24.72**	**-20.24**	**-26.16**	-9.82	**-21.07**	-9.82	**-18.49**	-9.23			
RH	-3.60	0.86	-5.05	11.28	0.04	11.28	2.62	**11.88**	**21.11**		
RM	**-13.67**	-9.20	**-15.11**	1.21	**-10.03**	1.21	-7.45	1.81	11.04	-10.06	
RS	**-13.51**	-9.03	-14.95	1.38	-9.86	1.39	-7.28	1.98	11.21	-9.91	-0.17

III.   Discussion

Land use and canopy cover metrics

Interpretation of this study's results rest on an understanding of existing, potential and relative CC metrics. As expected, potential CC values at both thematic scales of analysis were always higher than existing CC because potential CC (termed maximum potential CC by Kenney *et al*.,

2011) includes existing CC as well as grass/turf area with assumed potential to accommodate trees. Thus, potential CC likely best reflects constraints to land cover from zoning institutions given that UDO zoning regulates impervious surface coverage, the primary rule-based land cover constraint. Alternatively, relative CC, as the proportion of potential CC in existing CC, reflects indirectly the effects of zoning institutions but also unknown constraints on the establishment of trees in potential CC locations. These constraints are largely the active and passive decisions regarding pervious land cover made my urban land managers, including, for example, where to maintain mowed lawns for recreation, where to avoid tree growth due to utilities or where commercial signs and buildings require visibility. With this in mind, it is also important to note that existing CC values in this study represent only a snapshot in time and should not be interpreted as tree abundance; it is possible to have many small statured or recently planted trees and exhibit low existing CC.

In this context, canopy calculations make sense. Lower potential CC values for zones reflect the physical infrastructure required for land-use purposes, and generally, this leads to lower values for existing and thus relative CC. We reason that highest-to-lowest order among the three metrics is generally maintained for zone types and districts because, for less potential CC area, land management constraints likely increase, lowering existing relative CC as well. For instance, commercial zones' low potential CC values make sense given the amount of impervious surface allowed and required for business parking lots and building area for inventory. Likewise, low existing and relative CC values for commercial zones make sense given the assumed desire of businesses to maintain visibility to potential customers (although such reasoning brings to mind the role trees play in increasing time and money spent in "tree-friendly" businesses per Wolf (2005)). With less potential CC area with which to work, commercial zones are understandably lower in existing and relative CC than other zones.

The two exceptions to the trend that canopy metrics follow a general order are important, particularly for city managers. Industrial zones, while maintaining relatively high potential CC, have relatively low existing and relative CC, likely due to the fact that industries generally utilize a large portion of land area for factory buildings or other infrastructure. However, industrial zones in this study had the highest relative CC values, which we reason is likely due to the fact that there is little function for mowed grass/turf areas in these zones, and thus they are heavily

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treed, because most human activity takes place inside buildings or on impervious surfaces related to industrial work. On the other hand, institutional lands, which demonstrated some of the highest existing/potential CC values, had lower relative CC. In this case, land uses such as golf courses or other sporting fields require few to no trees to function, thus relative CC is low.

Our results regarding canopy metrics across institutional and PUD districts bring to light additional considerations for city planners and urban forest managers. These districts had some of the largest variation in canopy cover across their observations, an expected finding given that these zone districts are some of the broadest in purpose (containing schools, parks, and cemeteries in institutional, for example) with more variation in form compared to districts with one purpose (commercial or residential, for example). Despite variability among observations, the institutional zone had relatively high existing CC, on average, supporting previous findings that recreational or parklands have high canopy cover. However, our findings offer the caveat that these lands have low relative CC and, while they could accommodate more canopy (high potential CC), we believe land management for recreational uses likely negates this possibility.

The PUD type/district, with the purpose to encourage a harmonious and appropriate mixture of uses (UDO, 20.04.01, District Intent), had mid-range canopy metrics likely due to its mixed-use nature. Not surprisingly, these developments, which follow new urbanism patterns of mixed residential and commercial use, exhibit canopy cover values that fall between these zones' values. While new urbanism development aims to increase walkability from residences to commercial locations and is often touted as sustainable development relative to conventional development, it is worth consideration among urban planners that this dense form of development may result in a localized trade-off in canopy cover and impervious cover as Stone (2004) suggests.

Zoning scale

In terms of understanding the relationship between zoning analysis scale and canopy, our results can be generalized to draw similar conclusions to other studies that consider the relationship between urban land use and existing CC—that, in general, residential lands have higher levels of existing CC than many other zoned land uses. And in general, potential and relative CC are among the highest for residential zones. However, it is important to recognize that such

generalizations break down when land uses are specified at the disaggregated analytical scale of zone districts versus zone types in our study, and we suspect the same is true for other studies that analyze canopy cover by aggregated zone classifications only.

Disaggregated zone district-scale findings extend somewhat on Wilson *et al*. (2003)—that residential lands with lower densities have higher existing CC. The fact that districts zoned for high density residential use have significantly less existing, potential and relative CC compared to other residential districts (excluding the mobile home district) and that they are more akin to commercial districts in these terms is likely due to the fact they face similar constraints regarding physical infrastructure. The need for large parking areas and buildings to accommodate residents in high density zones is reflected in their potential CC values and likely links to impervious coverage allowances which are similar for commercial zones (see Table 1). Existing and relative CC values of high density and commercial zones are likely low for similar reasons such as the need for visibility for attracting customers or renters/owners. Alternatively, and as suggested above, the relationship between constraints on land management and potential CC are likely non-linear; land managers with smaller potential CC proportions may desire many uses of that land, but may be unable to fulfill them given certain minimum area requirements for each use (e.g., only so many trees can be located in a given area and still maintain some area for recreation). Thus, existing and relative CC are constrained.

These results demonstrate an important exception to the oft-cited theory that "residential lands" have higher canopy cover, a conclusion that our data supports only at the analytical scale of aggregated zone types. The generalization does not apply when our data is disaggregated to the zone district scale, pointing to the well-established problem of modifiable areal units (Openshaw, 1984) and underscoring its importance for city planners and managers. In fact, our finest scale of analysis—the zone district observation—does not rid this study of the same shortcoming. Better fit among cadastral and zoning datasets would have allowed comparison of zone district canopy cover to parcel-scale canopy cover which would support the most robust analysis.

Given that Bloomington's zoning policies do not include direct regulation of canopy cover by zone districts as in Chesapeake, Virginia, our results suggest the importance of zone-specific impervious cover regulations for attaining canopy cover goals, as noted above. Urban planners

and urban forest managers must recognize the potential, albeit indirect, significance of alternative land cover regulations on potential CC, and existing and relative CC if land-use constraints grow as potential CC decreases. Particularly relevant are regulations of land cover that can detract from urban potential CC, factors which may drive variation in canopy cover across zone districts, although further analysis is needed to make that determination.

Additionally, our findings suggest the importance of institutional scale. While American Forests' recommendations for urban canopy cover goals vary by land use, they are still at a coarse application scale and vary more by location (suburban residential versus urban residential) than by urban form (e.g. housing and population density). Planners and urban forest managers may be wise to apply canopy cover goals at finer scales of land use—the parallel to zoning districts as opposed to zone types in this case—because canopy cover varies significantly at this level of analysis as can policies related to other, potentially influential land covers. Likewise policies meant to incentivize meeting these goals should be scaled to this level as institutions associated with sustainable resource management have been characterized as those that best fit local conditions (Ostrom, 2005).

Limitations of our research are related to the lack of information about parcels within the zone district observations. As stated earlier, canopy cover analysis at the parcel scale while controlling for zone district and type would have yielded further information to resolve the modifiable areal unit problem. Moreover, parcel scale analysis would allow for consideration of the relationship between zone districts/type and age of development which may be an additional determinant of canopy cover. Additionally, variances are granted at the parcel-scale, and the extent to which they may influence land cover outcomes was beyond the scope of this research but an important future consideration. Even so, our results shed light on the importance of scale for urban forest managers and planners concerned with the influence of municipal land-use institutions on urban tree canopy cover.

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Chapter Four

THE INFLUENCE OF WATERING STRATEGY ON THE SURVIVAL AND GROWTH OF COMMUNITY-PLANTED TREES

I.   Introduction

Because urban forests provide a myriad of ecosystem services that constitute public goods, municipalities and nonprofits interested in the provision of public benefits are increasingly engaged in a variety of tree conservation and planting strategies. In fact, a recent study found that 127 of 329 (40%) U.S. cities have adopted an urban tree canopy cover goal and 246 (74.7%) have adopted a tree ordinance specifying tree planting requirements for developers (Krause, 2011). Less is known about the number of nonprofits engaged in tree conservation and planting strategies, although approximately 200 organizations are members of Alliance for Community Trees (ACTrees), a U.S.-based organization that supports grassroots, citizen-based nonprofit organizations dedicated to urban and community tree planting and care (ACTrees, 2012). Many of these member organizations serve a dual mission to improve the provision of urban trees and to engage and build communities, working primarily through private citizen groups and neighborhood associations rather than developers. To this end, some provide free or reduced-cost trees to groups that apply and have developed a plan to water and maintain trees through their establishment; after all, it costs approximately $150 to simply plant a street tree, while both planting and providing two years of maintenance costs approximately $250 (ACTrees, 2012). With maintenance pledged by applicants, tree-planting organizations cover tree stock and planting expenses through donated funds, partnerships with municipalities, or through cost-share agreements with local neighborhoods or homeowners receiving trees. Thus, non-profits working synergistically with private citizen groups *co-produce* the urban forest.

Despite these efforts and a community's best laid plans, planted trees, if not watered, may fail to become established in relatively harsh urban environments and, instead of a *source* of public goods (i.e., the benefits of trees), constitute a *sink* of public or charitable funds (Appleyard, 2000). Unfortunately, little research exists to define the magnitude of this problem. Moreover, little is known about the variety of neighborhood watering and management strategies and their relative success or failure in terms of tree establishment, l*et al*one community-building. Such information could greatly serve nonprofits' efforts (and by extension, municipalities) in

supporting successful community tree-planting projects and the co-production of public goods, such as the ecosystem services generated by urban trees.

This paper addresses the dearth of information regarding community-planted tree management through a case study of Indianapolis, Indiana's Keep Indianapolis Beautiful (KIB), Inc. and their NeighborWoods tree-planting program. Specifically, we examine the variety of tree watering plans across Indianapolis neighborhoods and ask: *How does tree survival and growth vary by community watering strategy and to what extent does watering strategy influence neighborhood collective action?*

Theoretical background

A. Water availability and tree growth and mortality

The availability of water is arguably the most important biophysical parameter influencing the establishment of transplanted trees. In natural environments, water is one of the most limiting factors on plant growth and a determinant of ecological community type; however, in urban areas and managed landscapes, human irrigation has the ability to overcome local water constraints (Whitlow *et al.*, 1992). Irrigation practices can therefore influence both the establishment and growth of planted urban trees, particularly in times of drought which were experienced during several summers studied within this research. The process of tree establishment in the landscape depends on root growth (Nilsson *et al.*, 2008). Lack of adequate water decreases cell turgor and temporary water stress causes wilting, while long-term water stress can decrease both root and shoot growth rates (Kozlowski and Pallardy, 1997). Water stress can limit root elongation through dry, compacted soils, inhibiting tree stability and ability to take up needed nutrients (Kozlowski and Pallardy, 1997; Kramer, 1987). Water stress also limits aboveground growth: Gilman (2004) found that live oak trees (*Quercus virginiana*) watered 3 times per week for the entire first growing season after transplanting grew twice as fast as trees watered only during the first 12 weeks. Ultimately, if trees face sufficient constraints to establishment, they will not survive or grow in the landscape (Appleyard, 2000).

B. Why use the vocabulary of institutional theory and collective action?

Conversations with urban forestry nonprofits have led us to conclude that these organizations are often very much aware that they are operating in social-ecological systems of both trees and the communities of people in which these trees are planted. However, they often struggle to find a language to describe the types of change their tree planting activities aim to inspire in communities. The vocabulary of collective action, social capital, and institutions used in this paper can help fulfill this need. We also believe that extending the application of institutional theories of collective action to the fields of urban forestry and urban ecology is a crucial step in improving understanding of urban neighborhoods in the context of sustainability (Mincey *et al*., in review). In the remainder of this section, we review these social and institutional theories related to collective action and co-production of resources.

C. Collective action and co-production

Collective action and social capital are important concepts in understanding the coproduction of urban services (Ostrom, 2009, 1996; Adger, 2003; Marschall, 2004). The Oxford Dictionary of Sociology defines collective action as *action taken by a group (either directly or on its behalf through an organization) in pursuit of members' perceived shared interests* (Scott and Marshall, 1994). Theories of collective action in the provision of conventional urban services (e.g., policing, education) are much more developed compared to theories of urban vegetation distribution and provision. For instance, studies by Ostrom and colleagues in the 1960s on urban policing demonstrated that citizen involvement in the provision of policing services (a form of *co-production* of these services) yielded enhanced service delivery (cited in Ostrom, 2009). Marschall (2004) found that citizen participation in the co-production of public safety and schooling efforts was related to involvement in both formal and informal associations (collective action). While not directly related to tree growth or survival, this literature suggests that communities that engage in more collective tree-planting activities may produce more successful tree-planting projects (i.e., with higher tree survival rates).

The mechanisms behind collective action and successful outcomes lie largely in institutions—the rules, norms, or strategies that groups use to structure repeated behavior (Ostrom, 2005). Co-provisioning is costly in terms of the time and effort required to come to agreement and to engage in mutually agreed-upon actions. Institutional mechanisms change the costs and benefits of actions, overcoming such social dilemmas because, for instance, they can be utilized to

impose sanctions on cheaters (of agreed upon actions) that aim to outweigh the perceived benefits of cheating. In fact, a number of institutional mechanisms beyond sanctioning have been associated with sustainable outcomes in community resource management and were specified in the Design Principles (Ostrom, 1990; Cox *et al.*, 2010). For example, the Design Principles point to the local relevance of rules, the clarity of resource boundaries and responsibilities, and effective monitoring and sanctioning for sustainable community resource management (Table 1).

Table 1. Ostrom (1990) Design Principles as modified by Cox *et al.* (2010).

Principle	Description
1A	<u>User boundaries:</u> Clear boundaries between legitimate users and nonusers must be clearly defined.
1B	<u>Resource boundaries:</u> Clear boundaries are present that define a resource system and separate it from the larger biophysical environment.
2A	<u>Congruence with local conditions:</u> Appropriation and provision rules are congruent with local social and environmental conditions.
2B	<u>Appropriation and provision:</u> The benefits obtained by users from a common-pool resource (CPR), as determined by appropriation rules, are proportional to the amount of inputs required in the form of labor, material, or money, as determined by provision rules.
3	<u>Collective-choice arrangements:</u> Most individuals affected by the operational rules can participate in modifying the operational rules.
4A	<u>Monitoring users:</u> Monitors who are accountable to the users monitor the appropriation and provision levels of the users.
4B	<u>Monitoring the resource:</u> Monitors who are accountable to the users monitor the condition of the resource.
5	<u>Graduated sanctions:</u> Appropriators who violate operational rules are likely to be assessed graduated sanctions (depending on the seriousness and the context of the offense) by other appropriators, by officials accountable to the appropriators, or by both.
6	<u>Conflict-resolution mechanisms:</u> Appropriators and their officials have rapid access to low-cost local arenas to resolve conflicts among appropriators or between appropriators and officials.
7	<u>Minimal recognition of rights to organize:</u> The rights of appropriators to devise their own institutions are not challenged by external governmental authorities.
8	<u>Nested enterprises:</u> Appropriation, provision, monitoring, enforcement, conflict resolution, and governance activities are organized in multiple layers of nested enterprises.

After studying multiple community management scenarios including fisheries, forests, and irrigation systems, Ostrom (1990) concluded that the Principles were common among long-enduring, or sustainable, systems, and argued that when individuals working together are able to

establish such institutions about shared responsibilities and have means to monitor and sanction those rules, they are more accountable to one another and accomplish more than individuals who do not (Ostrom *et al*., 1994). While not yet applied to urban forest management, these principles have emerged in recent studies related to urban vegetation management: for example, Robbins and Sharpe (2003) report that upholding aesthetic norms and the fear of neighborhood sanctions are key drivers to front yard maintenance. Thus, given the Design Principles theory and related evidence, *we hypothesize that neighborhoods that engage in collective tree-planting and maintenance activities (co-production) will produce more successful tree-planting projects.*

D. Collective action and social capital

Successful collective action has been linked to the existence of social capital both within- and across-groups (Adger, 2003; Ostrom, 1996). Social capital, as a measure of the strength and networks of human relationships, involves trust and reciprocity (Adger, 2003), elements that are also key to current and future successful collective action (Ostrom, 1996). In other words, working well together builds social capital which increases the capacity to continue working together. Suggestive of this link, Sommer *et al*. (1994) found that residents who engaged in tree plantings with a group of people were more satisfied with the outcome than residents who planted a tree by themselves. This same research group has also measured the attitudes of tree-planting program participants and non-participants toward trees and neighborhoods finding that participants were more satisfied with tree location, staking, maintenance quality, and neighborhood quality than non-participants in tree-planting programs (Summit and Sommer, 1998).

Outside of this research group, no systematic, quantitative research has been done to evaluate urban tree-planting programs from a social perspective. Elmendorf (2008) cites an extensive literature from urban planning and community development research, outlining the theoretical linkages between trees, tree planting and community capacity building; yet, to our knowledge, no research has empirically analyzed the effects of tree-planting programs on community collective action. However, the existing literature indicates reason to suspect that participation in collective tree-planting and management activities may have positive effects on other types of collective engagement. Consequently, *we hypothesize that communities that engage in collective tree-*

planting and management activities are likely to engage in additional collective efforts through built social capital.

Ultimately, collective action, social capital, and co-production of the benefits of trees in neighborhood-initiated tree-plantings should theoretically lead to more consistent and frequent efforts to maintain trees. Because of the known importance of watering to planted tree establishment and survival, *we hypothesize that any neighborhood activities that lead to frequent and consistent watering throughout the first few seasons after transplanting will lead to greater likelihood of tree establishment and therefore survival.*

II. Methods

Study site

To examine if tree survival or growth varies between tree watering strategies and to what extent strategy may influence neighborhood collective action, we partnered with Keep Indianapolis Beautiful, Inc. (KIB) to study a sample of neighborhoods throughout Indianapolis and Marion County, Indiana (Figure 1) that have participated in the organization's NeighborWoods program. Founded in 1976, Keep Indianapolis Beautiful is a 501c(3), private, not-for-profit organization, and an award-winning affiliate of Keep America Beautiful, Inc., the national organization dedicated to preserving the

Figure 1. Study location in Marion County and Indianapolis, Indiana, U.S.

natural beauty and environment in American communities. KIB's NeighborWoods program, begun in 2006 as part of the larger Alliance for Community Trees NeighborWoods initiative, is an urban forestry effort to strategically plant 100,000 trees throughout the City of Indianapolis and Marion County. (Indianapolis and Marion County are under a single metropolitan government unit.) Indianapolis is the state capital, with a metropolitan area population of just

over 800,000 and growing (US Census, 2010). The total area of the consolidated city covers 373.1 square miles (966 km^2) and is located in a flat, glaciated plain; thus, climatic and edaphic conditions are relatively consistent across the study site.

KIB NeighborWoods requires submission of an application demonstrating a group's self-organized plan to aid in the planting and establishment of ~1-2 inch caliper trees through a structured watering strategy of the neighborhood's choosing; strategies generally fall along a continuum from "collective watering," in which neighbors gather together at a specified time to water all trees, to "individual watering," in which individual neighbors are responsible for one or more trees (usually near their home) and water them individually at any time. Regardless of the specific type of watering strategy chosen, neighborhoods are instructed to give each planted tree at least 15 gallons of water every week during the summer (April-October) during which it does *not* rain at least 1 inch.

We selected a stratified sample of neighborhoods and homeowners associations (subsequently referred to as "neighborhoods") throughout Indianapolis that participated in KIB's NeighborWoods program between 2006 and 2009 based on a list provided by KIB of projects by watering strategy. Watering strategies were dichotomized and project selection was based on balancing the sample between the two types: neighborhoods where there was some form of collective watering compared to neighborhoods in which only individual watering was conducted. Thirty-six projects (in 25 distinct neighborhoods) were identified and selected in which a minimum of ~20 trees were planted, and, for purposes of access, where trees were planted in or near the public right of way or in a community's common areas.

Social data collection

Upon project selection, KIB employees provided contact information for at least one individual from each neighborhood that was involved in helping manage one or more of the tree planting projects in their neighborhood. (Because the same individual often managed multiple projects, in our analyses, all the tree-planting projects in a single neighborhood are considered part of the same sampling unit.) In the summers of 2011 and 2012, these individuals were contacted for an interview at a location of the interviewee's choosing in Indianapolis. In two cases, interviews were conducted over the phone as the individuals had moved away from the study site. Of the 25

neighborhoods for which tree data was gathered, a total of 19 were represented by one to four individuals per neighborhood who were interviewed. In the summer of 2011, two interviewers (one lead, one note-taker) were present for interviews that were not audio-recorded (n=7). In the summer of 2012, audio recording was approved by the researchers' institutional review board; subsequently, one interviewer was present for each interview which was recorded and later transcribed (n=12). In all cases, interviews were approximately one hour in length and semi-structured in nature. Detailed interview notes and transcriptions were used to code interview data relevant for analysis, including whether or not watering was undertaken and for how long, whether the community formalized the agreement that neighbors water (e.g., a signed commitment), whether neighbors monitored and/or sanctioned (e.g., reminded, prodded) one another regarding watering and whether that purportedly changed watering behavior.

Additionally, to determine the effect of watering strategy on collective action, we asked interviewees to enumerate and describe collective efforts that took place in their respective neighborhoods both before and after their initial NeighborWoods project. This question was open-ended in nature, although we offered the example of community crime watch to help define "collective efforts." A complete list of such activities was generated *a posteriori* and coded with 1s and 0s (presence/absence) for each neighborhood. The percents of all potential collective activities occurring in each neighborhood before and after tree planting were generated, and an index of "change in collective activities" was created by subtracting the percent of activities after planting from the percent of activities prior to planting for each neighborhood. This yielded a metric of change in collective activities after planting.

Biophysical data collection

Data about the survival, growth and condition of individual trees in each neighborhood project was collected using a planted tree re-inventory protocol developed by the authors for the purpose of this study (Vogt *et al.*, 2012). Tree inventory methods were modified from those of the Urban Forestry Data Standards initiative (www.unri.org/standards) for use with young, recently planted trees and for use by high school aged youth. The authors and members of KIB's Youth Tree Team collected data during the summer months (June-August) of 2011 and 2012, identifying all dead or missing trees from the total planted per project, and systematically sampling 20-30 trees

per project upon which the full suite of variables in the planted tree re-inventory protocol was collected if the tree was alive. Data collected via the re-inventory protocol was combined with data collected by KIB at time of planting (namely, caliper at planting) and average annual caliper growth rates were calculated for each sampled tree. The analyses here focus on tree survival, growth rates, and condition ratings in neighborhoods with different tree maintenance strategies.

Statistical analysis

Most statistical analyses were performed using the individual tree as the unit of analysis, with different types of neighborhoods serving to divide the trees into categories for comparison of growth and mortality rates. This allows us to take advantage of the relatively large sample of trees (n=1492 trees) instead of deferring to the smaller sample of neighborhoods (n=18). Analyses of collective action pre- and post-planting were conducted at the neighborhood scale. All statistical analyses were performed in SAS statistical analysis software (SAS Institute, Cary, SC). PROC FREQ with the CHISQ MEASURES option was used to test for significant differences in mortality rates (assignment to 'Dead' or 'Alive' category) and tree condition ratings across neighborhoods with different characteristics. PROC TTEST, PROC ANOVA (for balanced samples), and PROC GLM (for unbalanced samples) were used to test for significant differences in average growth rates between trees in neighborhoods with different characteristics (e.g., signed commitment to watering or no signed commitment). For all analyses, a p-value of less than 0.05 was considered statistically significant, and p-values between 0.05 and 0.10 were considered marginal.

III.   Results and Discussion

Of the 6366 trees planted in NeighborWoods projects by KIB between 2006 and 2009, 1462 trees occurred in the sampled neighborhoods and 1304 (89%) were found to be alive, while 158 (11%) were found to be dead or missing upon sampling. Of the 1304 living trees in sampled neighborhoods, growth rates were measured for 663 sample trees. Overall, average caliper growth rate was 1.11 cm/year (standard deviation = 0.61). Eighty-five point four percent of sampled trees were in good condition, while 11.2% were in fair condition, and only 2.5% were in poor condition (0.9% of trees were in shrub form and condition was not assessed).

Watering strategy (collective versus individual) and signed watering agreements

Our results demonstrate the importance of watering strategies on the survival and growth of community planted trees. Collective watering strategies in sampled KIB NeighborWoods appear to be associated with more consistent and thorough watering given that trees in neighborhoods with some collective watering were more likely to be alive than trees in neighborhoods with no collective watering (Table 2). Community watering likely increases individual accountability to follow-through with planned watering given the physical presence of other neighbors at the specified watering time. The accountability mechanism at play is institutional in nature. The fact that neighbors can see one another watering leads to the perception that there is an increased chance that one could be "caught cheating" (not watering) and face social sanctioning. Even if social sanctioning seems unlikely, feelings of guilt or self-sanctioning related to known institutions can motivate action. Interviewee statements support this reasoning, with such phrases as, "[when] the dedicated few [are watering]…the others, out of guilt, will walk outside and start helping…[or] come back later." For this neighborhood and others in our sample, neighborhood choice of collective watering strategy is related to consistency and frequency of tree watering activities, and, thereby, to tree survival.

Table 2. Significant differences in tree biometrics between neighborhood watering strategies were observed for survival and condition, but not for growth rates in sampled KIB NeighborWoods projects. Percentages may not add up due to rounding error.

		No collective watering	Some collective watering
SURVIVAL[*]	*Alive*	86.5%	93.7%
	Dead	13.5%	6.3%
	Total n	908	554
TREE CONDITION RATING[**]	*Good*	88.3%	81.2%
	Fair	9.5%	15.4%
	Poor	2.2%	3.4%
	Total n	358	298
ANNUAL CALIPER GROWTH[***]	*Mean*	1.11 cm/yr	1.15 cm/yr
	Std dev	0.62 cm/yr	0.61 cm/yr
	Total n	357	302
[*]Signif. diff.: Frequency table analysis: n=1462, X^2=18.652, p<0.0001, df=1. [**] Signif. diff.: Frequency table analysis: n=656, X^2=6.402, p=0.041, df=2 [***] No signif. diff.: T-test: n=659, t=-0.83, p=0.409, df=657			

The living trees in neighborhoods where at least some collective watering occurred were more likely to be in worse overall condition (Table 2). This makes sense because higher rates of tree mortality in individually watered neighborhoods also results in better average tree condition, since those individual trees in the poorest condition have succumbed to death and dropped out of the sample of living trees given condition ratings. In other words, more consistently watered trees are better able to "hang on," albeit in worse condition, in collective watering scenarios.

In contrast to tree mortality and condition rates, tree growth rates did not differ significantly by neighborhood watering strategy (Table 2). This is not unreasonable as watering newly-planted trees is much more tightly coupled to their survival and root growth than early aboveground growth. For instance, Gilman (2004) found no differences in shoot growth between frequently and infrequently irrigated trees after the trees were established. Thus, we suggest that mortality is more strongly linked to the institutional and management variables than aboveground growth.

Although collective watering had no impact on tree growth rates independently, the effect of the *interaction between choice of collective watering strategy and having a signed watering agreement* on growth rates was significant (Table 3). The positive effects of signed agreements appear to have significantly improved tree growth rates even in the absence of our theoretically important variable of collective watering. Additionally, either a signed watering agreement or collective watering or both can significantly improve mortality outcomes over trees in neighborhoods with neither signed agreements nor collective watering.

Table 3. Annual caliper growth and survival rates for trees in sampled KIB NeighborWoods projects with and without signed watering agreements and with no or some collective watering.

	ANNUAL CALIPER GROWTH[*]	No signed agreement	Signed agreement	SURVIVAL RATES[**]	No signed agreement	Signed agreement
No collective watering	Mean (cm/yr)	0.96 [a]	1.34 [c]	Alive	80.6% [a]	90.7% [b]
	Std dev	0.49	0.55	Std dev	39.6%	29.1%
	n	128	78	n	304	215
Some collective watering	Mean (cm/yr)	1.10 [a b]	1.22 [b c]	Alive	94.2% [b]	92.5% [b]
	Std dev	0.58	0.66	Std dev	23.5%	26.4%
	n	189	113	n	394	160
ALL TREES	Mean (cm/yr)[***]	1.04 [*]	1.27 [*]	Alive[****]	91.5%	88.3%
	Std dev	0.55	0.62	Std dev	32.2%	28.0%
	n	317	191	n	375	698

[abc] Means with the same letter are not statistically different from one another.
[*]Signif. diff. in growth for interaction between collective watering and signed agreement: GLM for unbalanced samples: n=508, F=5.75, p=0.0169, df=1
[**] Signif. diff. in survival for interaction between collective watering and signed agreement: . GLM for unbalanced samples: n=1073, F=12.46, p<0.0001
[***] Signif. diff. in growth for signed agreements: T-test: n=508, t=-4.32, p<0.001, df=506
[****] Marginal diff. in survival for signed agreements: Frequency table analysis: n=1073, X^2=2.655, p=0.103, df=1

Signed watering agreements alone appeared to positively moderate growth regardless of watering strategy (Table 3). Trees in neighborhoods that instituted signed watering agreements had significantly greater growth rates and were marginally more likely to be alive (Table 3), although no significant differences in condition ratings were evident (Frequency table analysis: n=505, X^2=0.688, p=0.709, df=1; not shown in Table 3). These results suggest the importance of signed agreements as a formal (written) institution for ensuring any type of watering occurs and establishing accountability to neighbors and perhaps also to KIB. Other studies have emphasized the importance of *written* rules for building social capital and influencing outcomes; Svendson and Svendson (2010) attribute the written form of cooperative farming association rules in Denmark to social capital building that led to economic growth in poor rural areas.

Monitoring and sanctioning

Monitoring positively moderates tree survival, while monitoring and sanctioning that reportedly changed watering behavior appears to have had a positive impact on both survival and growth (Table 4), findings supportive of the Design Principles (Cox *et al*., 2010; Ostrom, 1990; Table 1).

Trees in neighborhoods that monitored whether or not trees were watered were significantly more likely to be alive than trees in neighborhoods with no monitoring. However, monitoring alone appears to have no influence on tree growth or overall tree condition (Table 4). Monitoring occurs naturally in collective watering scenarios because a group of individuals can generally see one another watering. However, many interviewees we spoke with referred explicitly to monitoring outside of watering times while driving or walking in their respective neighborhoods. Parallels can be drawn from community forestry research; better forest conditions have been associated with the use of monitors that are local resources users themselves (Banana and Gombya-Ssembajjwe, 2000; Ostrom, 2005).

Table 4. Differences in tree biometrics by existence of monitoring activities in sampled KIB NeighborWoods projects. Percentages may not add up due to rounding error.

		No monitoring activities reported	Monitoring activities reported
SURVIVAL[*]	*Alive*	73.0%	91.3%
	Dead	27.0%	8.7%
	Total n	111	962
TREE CONDITION RATING[**]	*Good*	90.4%	83.4%
	Fair	7.7%	13.5%
	Poor	1.9%	3.1%
	Total n	52	453
ANNUAL CALIPER GROWTH[***]	*Mean*	1.12 cm/yr	1.13 cm/yr
	Std dev	0.55 cm/yr	0.59 cm/yr
	Total n	51	457

[*]Signif.diff.: Frequency table analysis: n=1073, X^2=35.079, p<0.0001, df=1
[**] No signif. diff.: Frequency table analysis: n=505, X^2=1.689, p=0.430, df=2
[***] No signif. diff.: T-test: n=508, t=-0.08, p=0.937, df=506

Monitoring alone and unknown to neighbors would have little impact if not for subsequent sanctioning that effectively changes behavior. Trees in neighborhoods that reported that monitoring and sanctioning changed watering behavior were significantly more likely to be alive and experienced higher tree growth rates than trees in neighborhoods that monitored without a subsequent reported behavior change (Table 5). Monitoring and sanctioning activities that change behavior also appear to affect tree condition in the same way that collective watering strategies do: trees in neighborhoods where monitoring and sanctioning affects behavior are more likely to be in poorer condition than trees in neighborhoods where monitoring and sanctioning does not change behavior (Table 5).

Table 5. Differences in tree biometrics in sampled KIB NeighborWoods by whether or not monitoring and sanctioning changed watering behavior. Percentages may not add up due to rounding error.

		Monitoring and sanctioning *did not change* behavior	Monitoring and sanctioning *changed* behavior
SURVIVAL[*]	*Alive*	89.2%	93.5%
	Dead	10.8%	6.5%
	Total n[+]	472	429
TREE CONDITION RATING[**]	*Good*	88.7%	76.4%
	Fair	8.5%	19.7%
	Poor	2.8%	3.9%
	Total n[+]	177	208
ANNUAL CALIPER GROWTH[***]	*Mean*	1.00 cm/yr	1.27 cm/yr
	Std dev	0.58 cm/yr	0.59 cm/yr
	Total n[+]	176	209

[*]Signif. diff.: Frequency table analysis: n=901; X^2=5.142, p=0.023, df=1
[**] Signif. diff.: Frequency table analysis: n=385, X^2=10.347, p=0.0057, df=2
[***] Signif. diff.: T-test: n=385, t=-4.47, p<0.0001, df=383
[+]Note that the smaller number of trees (*n*) reported in the monitoring and sanctioning analyses represents only the trees in neighborhoods out of the total that reported any monitoring (14/18).

Interviewees generally reported mild sanctioning, such as confronting and prodding neighbors who were not watering or leaving reminder notes on doors, and it appears to have been effective; for instance, one interviewee stated, "So as long as people aren't afraid to call their neighbor on it, and say, come help me, it all works." In such a situation, the cost of social "punishment" likely overcomes the costs of time and energy to water. These findings that informal monitoring and mild sanctioning improves biophysical outcomes are consistent with findings of institutional researchers in other resource systems (Table 1, Principles 4A and 4B). For instance, Gibson *et al*. (2005) examined 178 forested communities and found that regardless of the formality of an organization, regular monitoring and sanctioning of whatever rules are actually in place is related to better forest conditions and that irregular monitoring and sanctioning of rules was associated with poorer forest conditions.

Collective action

Watering strategy may play an important role in bringing neighborhoods together for subsequent collective action. Neighborhoods that chose to water trees collectively had marginally lower

levels of collective activities *before* their tree planting compared to neighborhoods that watered individually (T-test: n=18, t=2.08, p=0.0541, df=16), suggesting inexperience working together upon tree-planting. However, neighborhoods that watered at least some trees collectively experienced a positive change in collective action (i.e., more collective activities reportedly occurring after the tree planting compared to before), while neighborhoods choosing only individual watering experienced a negative change in collective action (i.e., fewer collective activities were reported to occur after the tree planting compared to before) (T-test: n=18, t=-3.23, p=0.0052).

The fact that neighborhoods were marginally more likely to work together after tree planting and maintenance compared to neighborhoods that individually watered may suggest that by watering and maintaining trees together, these groups learned from the experience and built social capital, supporting their undertaking of additional collective efforts. This theory is strongly underpinned by the sentiment of one interviewee from a neighborhood that collectively watered: "I feel like all of this [the NeighborWoods tree-planting project] has certainly brought our neighborhood closer together. It helps with the crime watch, too, because more people know each other…and it really all started with our first NeighborWoods project. It's been good – huge success."

Working through the collective operation of watering appears to give a sense of accomplishment in community and the feeling of accountability to one another, characteristics that one might argue demonstrate trust. Trust allows individuals to feel as though they won't be cheated or end up as the "sucker" doing all the work (Ostrom, 2000); thus additional efforts seem less costly. Another interviewee explained that more regular neighborhood board meetings resulted in "…more of a community feel, and [we started to] take ownership for what was going on." Other neighborhoods echoed the sense that tree-planting project ownership could actually lead to greater ownership in the neighborhood as a whole.

There are several reasonable causal pathways for why neighborhoods that water individually had higher levels of initial collective watering activities and lower post-tree planting/watering activities. Such neighborhoods could have experienced several unsuccessful collective activities prior to tree planting and therefore decided to undertake fewer collective activities through time and to water trees individually. Alternatively, such neighborhoods could have built great trust

and goodwill via prior collective activities and therefore feel they can water individually and take on subsequent civic operations individually with no detriment in outcomes. To tease out the relevance of these causal pathways requires additional future research.

IV.   Conclusions and Implications

The findings here suggest the importance of watering strategies for the survival and growth of community planted trees as well as for building subsequent collective efforts in neighborhoods and homeowners associations. Such findings may have implications for nonprofits and the management of tree planting projects. However, prior to considering these practical implications, it is important to recognize two important contextual issues: first, a word of caution about analyzing institutional variables in isolation of biophysical or social variables is necessary; and second, the institutional context in which these community strategies operated as observed in this study is relevant.

We recognize that the watering and management strategies discussed here do not occur in a vacuum: there is social and biophysical variation both between and within neighborhoods that may affect how trees survive and grow. Variation in socio-economic status may affect the ability of a neighborhood to care for trees. Differences in species composition, biophysical growing conditions, and at-planting parameters, such as nursery of origin or packaging, both within and between neighborhoods may impact the success of individual trees. However, because systematic variation in these parameters is unlikely between neighborhoods, and water availability has been shown to be one of the most important factors influencing tree establishment (Kozlowski and Pallardy, 1997), we believe that the results shown regarding watering strategy hold despite neighborhood variability. We also recognize the possibility of multiple causal pathways and do not purport that there is necessarily a direct causal relationship between watering strategies and tree outcomes. Future analyses will consider the simultaneous influence of watering strategy and other neighborhood-level social and institutional factors as they vary with tree-level biophysical parameters.

The watering and management strategies discussed here also occur within an organizational context that can influence their success. Keep Indianapolis Beautiful has established a NeighborWoods tree-planting program that creates an excellent institutional environment for

enduring and effective community tree management; in fact, their practices echo the Design Principles (Table 1). By working alongside neighborhoods to plant the trees and providing them with the information and some means to undertake projects, KIB and communities are working as nested enterprises—sharing the burden of a complex resource management undertaking (Table 1, Principle 8). By allowing communities to collectively choose their own watering strategies, KIB is also supporting the Principle that rules should fit local environmental and social conditions (Table 1, Principle 2A). Furthermore, by offering autonomy to the communities in the management of their watering strategies, KIB, as a "higher authority" is recognizing the rights of the communities to devise their own rules (Table 1, Principle 7). Given the average rate of tree success in NeighborWoods communities falling within this context (an average 5-year tree survival rate of 89%), it appears that KIB's program management, which unintentionally follows multiple Design Principles, is rather effective.

In keeping with these ideas, the observed variation in watering strategies between neighborhoods is not only expected but desired. Not only do the Design Principles imply the desirability of diversity, but tenants of adaptive management suggest the same. Variation in management strategies within a resource system or between like resource systems allows for natural experimentation (of which this study has taken advantage) and bolsters robustness of the resource sector as a whole (Holling, 1995). By examining trees in neighborhoods with different watering strategies, one can determine which strategies work well and which work poorly in various contexts, and avoid losing all planted trees in the worst case scenario that one single applied strategy fails. Thus, it is important to recognize that there are no panaceas in terms of one best community management strategy (Ostrom *et al.*, 2007). While we find higher tree survival among communities that collectively water, we observe other strategies that support survival and growth, including the presence of signed watering agreements regardless of watering strategy. For example, some neighborhoods may be unable to find one time to water together, but may be better equipped with other mechanisms to maintain accountability, such as utilization of social media to "check in" when individuals water. Effective strategies can be developed from the bottom-up (neighborhoods) with support and relevant information-sharing from the top (KIB).

Thus, we advocate that tree-planting nonprofits working with neighborhoods and homeowner associations should simply share the results of this study and the relevant theory addressed herein

with communities devising their own tree management strategies. We conclude that for communities interested in both tree survival and building social capital, collective watering strategies may be most effective. Yet, regardless of whether individuals water together or individually, some form of signed watering agreements should be utilized as it appears to improve accountability to water and therefore survival of trees. Moreover, monitoring and subsequent sanctioning, even in the form of prodding a neighbor that appears to be slacking in watering, is an effective tool for improving both survival and growth of community-planted trees. These strategies, and those undertaken by nonprofits which offer autonomy and support to communities, may best set the stage for efficient co-production of urban forests.

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Chapter Five

THE RELATIVE INFLUENCE OF SOCIAL INSTITUTIONS ON THE DIVERSITY OF RESIDENTIAL URBAN FORESTS

I.   Introduction

The bulk of urban trees are located on private, residential parcels where there is a disproportionate reliance upon landowners—homeowners and community associations alike—to maintain this substantial portion of the urban forest for its provision of ecosystem services (Clark *et al.*, 1997; Kenney *et al.*, 2011). Sustained provision of ecosystem services from urban trees has been credited with supporting sustainable urban communities, and been linked to maintenance of robust urban forest structure (Clark *et al.*, 1997, McPherson, 2006, Nowak *et al.*, 2008, Kenney *et al.*, 2011, UNCOB, 2012). However, little is known about private urban forest structure or the social factors that drive it, in part due to difficulty in obtaining access to privately owned plots, and, because many studies of urban ecology tend to isolate the behaviors of individuals from the institutions that influence their actions (Foster, 2000). This is unfortunate, as the abundance of institutions affecting private tree management grows, in part due to the increase in homeowner associations in the U.S. (Stabile, 2000; Low, 2003). As evidence mounts that urban forests are declining as a whole, a better understanding of what factors influence urban forest structure on private, residential lands increases in importance (Nowak and Greenfield, 2012). To that end, this study utilizes full tree inventories of private residential parcels across homeowner and neighborhood associations in Bloomington, Indiana to address the relative influence of social-ecological system (SES) factors, particularly social institutions, on private, residential parcel tree diversity in terms of species richness.

Urban Forest Structure: Metrics and Drivers

Clark *et al.* (1997) and Kenney *et al.* (2011) define metrics of sustainable urban forest structure considered important for robust provisions of ecosystem services. Mutually, they point: (1) to an uneven-aged tree distribution which ensures enough young trees will persist to replace older trees as they die; (2) maximized, climate-appropriate canopy cover which ensures adequate ecosystem services linked to leaf area; (3) high species diversity at fine scales to ensure sustained

services despite species-specific pests and pathogens; and (4) use of native species adapted to local conditions with recognition that in cities, non-natives can be better than no trees at all.

Utilization of these structural metrics and their application for determining drivers of private residential urban forests is varied in the literature. Canopy cover analysis is common given the relative ease of remote sensing data collection despite lack of detailed information such as tree species and size distributions. For instance, it was utilized to determine the positive influence of neighborhood development age on tree abundance in Baltimore, Maryland (Grove 2006). Age-distribution metrics have principally been utilized at the city-scale, and lack of basic information regarding the longevity of species in heterogeneous urban environments stifles the ability to ascertain optimal age distributions to operationalize this metric at finer scales (Personal communication with US Forest Service Ecologist, Paula Peper, 9 Sept. 2012). Studies of native versus non-native tree species are lacking and somewhat controversial as to their importance (Kenney *et al.*, 2011), but at least one study using data from Syracuse, New York and Baltimore, Maryland found more than half of residential tree species to be non-native and their occurrence and prevalence to be driven by patch history and site disturbance (Zipperer, 2010). Various measures of species diversity have been used in several urban studies given their role in maintaining vegetation populations that are robust to species-specific pests and pathogens and in influencing the composition and abundance of associated biota (Matson *et al.*, 1997). However, analyses of tree species diversity have been limited by the costs of obtaining access to these private, residential lands.

Those studies that have focused on forest structure, particularly species diversity, at the private, residential parcel scale have offered important information regarding its drivers. Hope *et al.* (2003) and Martin *et al.* (2004) found in Phoenix, Arizona that newer neighborhoods had higher plant abundance and species richness in front yards, but also tended to be wealthier, suggesting income as a confounding covariate with age. However, Hope *et al.* (2003) concluded that in addition to elevation and current and former land use, family income explained variation in perennial plant diversity, hypothesizing a "luxury effect." Grove *et al.* (2006), having found, in contrast, a positive association between development age and tree abundance, suggested an alternative hypothesis—that homeowners are likely to maintain landscapes similar to their neighbors' because of social status or norms. Indeed, in multiple cities in Michigan, Nassauer *et*

al. (2009) found community norms regarding landscape appearance to influence *exurban* household preferences for *front* yard ecological design, likely linked to species composition. Underscoring the importance of full-parcel data, Larsen and Harlan (2006) found significant differences between *front* and *back* yard landscaping behaviors and landscaping preferences as reported in a household survey, finding income as the only significant predictor of front yard preferences, time since development and homeowner preferences as significant predictors of front yard landscaping, and only homeowner preferences as a significant predictor of backyard landscaping. A possible relationship between family life stage and urban vegetation heterogeneity has also been hypothesized, but not tested (Grove *et al*., 2006; Troy *et al*., 2007). These studies offer important hypotheses to test regarding private residential tree species richness with complete parcel–scale tree inventories, a specific focus unaddressed in the literature.

The Role of Institutions in Urban Forest Structure

An additional limitation to our understanding of the relative importance of drivers of private residential urban forest structure stems from a historically limited focus on the role played by governance and institutions—the rules, norms, and strategies that structure human decision-making and action (Ostrom, 2005). Humans at various scales, from households to city blocks to neighborhoods to governments, are part of an urban ecosystem and make decisions that are integral to sustaining it (Grove, 2009; Roy-Chowdhury *et al*., 2011). At each level, human decisions and actions are based on incentives which are adjusted by institutions (Ostrom, 2005). As a result, an explicit and clear accounting of institutions at various scales is necessary in urban SES research.

A framework that establishes the relative importance of various institutional mechanisms on equal footing with biophysical and social factors is the Social-Ecological Systems (SES) Framework (Ostrom, 2009) (Table 1). The framework links attributes of the physical world (*the resource system* and *resource units*) with those of institutions *(governance systems)* that affect the incentives and constraints of the actors (*users*) in the system and produce outcomes. The framework unpacks these first tier components to reveal second tier variables, at scales relatable to decision-making arenas (e.g., households to neighborhoods to city government, etc.).

There are several types of institutional objects contained in the framework. Norms (U6) are the values an "individual places on actions or strategies in and of themselves, not as they are connected to immediate consequences" (Ostrom, 1990, 35); they are institutional prescriptions for behavior *without* defined sanctions but can constrain human behavior due to fear of social censure (Ostrom, 1990; 2005). Rules are institutional prescriptions for behavior which require, prohibit, or permit some action or outcome *and* include sanctions if a rule is not abided; the focus is more specifically on "rules-in-use" as rules followed in practice produce outcomes. This approach further defines three nested layers of influential rules: constitutional rules (GS7), collective choice rules (GS6), and operational rules (GS5). Constitutional rules set groundwork for formation, governance, adjudication, and modification of the collective choice arena. Collective choice rules regulate how collective decisions are made on management or policy. Operational rules establish allowable or required day-to-day actions.

These distinctions are important, particularly for urban ecological research, because they allow researchers to parse out specific institutional arrangements of influence. For instance, Larsen and Hall (2008) determined the powerful influence of homeowner associations (HOAs) and their institutions on the structure of residential front yards in Phoenix. HOAs, in conjunction with Neighborhood Associations (NAs), represent an important set of constitutional arrangements for residential parcels. HOAs, as legally incorporated entities, maintain formal rules (covenants, codes, and restrictions (CCRs)) that influence parcel-scale management. NAs do not govern parcel-scale management legally, although institutions in the form of norms may influence parcel management. This constitutional rule variation influences collective choice and operational rules as the three are linked. For instance, Larsen and Harlan (2008) reported operational rules of HOAs actively regulated the composition and maintenance of residential landscapes; such rules could not be enforced by an NA without constitutional groundwork for governing such decisions. Moreover, research in rural SESs has found that collective-choice arrangements that allow most resource users to participate in the decision-making process regarding operational rules are associated with more sustainable outcomes in community resource management (Ostrom, 1990). Thus, this framework offers a means for exploring urban forest structure as an outcome of interactions within an urban SES composed of complex institutional arrangements that can influence parcel-scale management.

**Table 1.** Second level SES variables per Ostrom (2009).

Social, economic, and political settings (S)	
S1 Economic development. S2 Demographic trends. S3 Political stability.	
S4 Government resource policies. S5 Market incentives. S6 Media organization.	
Resource systems (RS)	*Governance systems (GS)*
RS1 Sector (e.g., water, forests, pasture, fish)	GS1 Government organizations
RS2 Clarity of system boundaries	GS2 Nongovernment organizations
RS3 Size of resource system	GS3 Network structure
RS4 Human-constructed facilities	GS4 Property-rights systems
RS5 Productivity of system	GS5 Operational rules
RS6 Equilibrium properties	GS6 Collective-choice rules
RS7 Predictability of system dynamics	GS7 Constitutional rules
RS8 Storage characteristics	GS8 Monitoring and sanctioning processes
RS9 Location	
Resource units (RU)	*Users (U)*
RU1 Resource unit mobility	U1 Number of users
RU2 Growth or replacement rate	U2 Socioeconomic attributes of users
RU3 Interaction among resource units	U3 History of use
RU4 Economic value	U4 Location
RU5 Number of units	U5 Leadership/entrepreneurship
RU6 Distinctive markings	U6 Norms/social capital
RU7 Spatial and temporal distribution	U7 Knowledge of SES/mental models
	U8 Importance of resource
	U9 Technology used
Interactions (I) → outcomes (O)	
I1 Harvesting levels of diverse users	O1 Social performance measures
I2 Information sharing among users	(e.g., efficiency, equity, accountability, sustainability)
I3 Deliberation processes	O2 Ecological performance measures
I4 Conflicts among users	(e.g., overharvested, resilience, bio- diversity, sustainability)
I5 Investment activities	O3 Externalities to other SESs
I6 Lobbying activities	
I7 Self-organizing activities	
I8 Networking activities	
Related ecosystems (ECO)	
ECO1 Climate patterns. ECO2 Pollution patterns. ECO3 Flows into and out of focal SES.	

This research utilizes the SES framework and full tree inventories of private residential parcels to address the questions: How do parcel-scale SES characteristics vary by development age and community association type (homeowners or neighborhood association)? What is the relative influence of these and other SES factors on private, residential urban forest diversity in terms of species richness? Given the literature reviewed here, we are particularly interested in the effects of parcel age, income, and community institutions on within-parcel tree species richness.

II. Methods

Study Location

This research was carried out in the City of Bloomington, Indiana, located in Monroe County approximately 50 miles southwest of the state capital of Indianapolis (Figure 1). As of 2010, Bloomington's population had reached 80,405 individuals and 33,239 housing units (US Census Bureau, 2010), representative of where most future urban growth is expected to happen—small and medium-sized cities (UNCOB, 2012). The city is situated in the Norman Upland Plateau physiographic region of Indiana, with variable topography typical of the unglaciated portion of the state (Hill, 2011). The surrounding landscape is heavily forested for Indiana, and the city itself has been recognized as having one of the highest proportions of canopy cover in the state, at approximately 50% (Heynen and Lindsey, 2003).
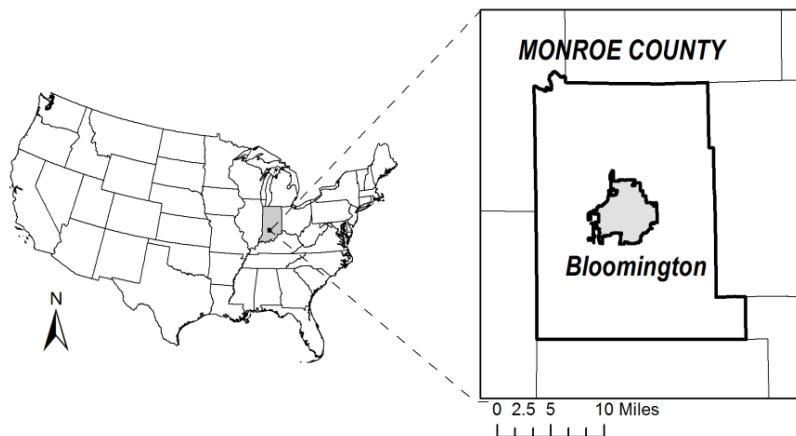
Figure 1. The study site of Bloomington, Indiana, located in Monroe County.

Bloomington's Unified Development Ordinance (UDO, 20.02; 20.05) and the Bloomington Tree Ordinance (Bloomington Municipal Code 12.24, Trees and Flora) determine city-level policies regarding urban tree management and influence all parcels within the city. The UDO, passed in 2007, stipulates that a specific percentage of tree canopy cover must be retained during land development depending on the baseline canopy of a site. The section also includes a provision for preferencing native trees, undisturbed or virgin woodlands, and older forest growth over younger stands of trees by the placement of such stands in tree conservation or preservation

easements[2]. In addition to the tree canopy cover provisions in the UDO, Bloomington's Tree Ordinance requires that abutting property owners manage street trees in the public right of way (PROW) and private boundary trees (those that influence public space). Owners are required to remove all dead, diseased, or dangerous trees, prune the branches of such trees to avoid obstructing traffic signs, street intersections, or street lamps, and provide clearance of tree branches above streets and sidewalks. The ordinance further outlaws topping of any public or private boundary tree. For all management affecting these trees, the city requires a free permit.

Data Collection: Sample Design, Household Survey, and Field Inventory

The City of Bloomington has 49 Neighborhood Associations (NAs) and approximately 65 Homeowners Associations (HOAs) (collectively referred to as communities). The HOAs, as legally incorporated entities, maintain formal rules (covenants, codes, and restrictions (CCRs)) that influence parcel-scale management. NAs do not govern parcel-scale management. Beyond institutional differences, communities vary widely in their age of development, a theoretically important factor for which we can attempt to control in research design.

In an effort to capture the heterogeneity of communities and their rules and norms associated with tree management, we employed a stratified random sampling approach in which similar combinations of HOA and NA, and 'old' (1950 - 1986) and 'new' (>1986) communities were represented. The age of communities was determined by randomly sampling approximately 10% of parcels within community boundaries from the Monroe County Online GIS website which details year of building construction. In total 14 communities (7 HOAs and 7 NAs) were selected and CCRs were obtained and analyzed from communities for which they existed (largely HOAs). In generating this sample, we avoided the core of the city (defined by a radius of approximately 2 kilometers from the city center) where residential lands tend toward high-density and selected communities in which owner-occupied, single–family housing dominated (determined by mapping Monroe County tax assessor shapefiles and city zoning boundaries).

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[2] Unfortunately, incomplete and inaccurate easement location data made it impossible to utilize systematically within our data analysis, although conversations with city officials elucidated general locations of easement for consideration in interpretation of our results.

In the spring of 2011, household surveys were sent to ~1100 households (the total number of owner-occupied parcels within the 14 communities selected). These were later followed by reminder postcards and a replacement survey per the Dillman Method (Dillman, 2007). The survey asked households questions regarding their tree management and how it is influenced by neighborhood- and city-level rules, the purported benefits and costs of their trees, and household demographics. A total of 420 surveys were returned (38% response rate) from which 230 respondents cited their willingness to have their parcel inventoried. From this set of 230, we randomly selected 106 parcels to be inventoried in the summer of 2011, ensuring at least 10% of all communities' owner-occupied parcels were represented. One exception was a very large community (>400 parcels) in which we sampled ~3% of parcels due to time/budget constraints.

At each sampled parcel, a census of all trees was conducted. All trees with diameter at breast height (DBH) $\geq$ 2.5 cm were identified to species level (including those in the public right of way (PROW) adjacent to the sample parcel), geo-located using a Trimble GeoExplorer 2008 GeoXH GPS unit, and measured for their DBH (at 1.37 m above surface) and height (via a Nikon Forestry 550 laser rangefinder/hypsometer). Additional information regarding the maintenance of each tree (e.g. presence of mulch, pruning), condition of each tree, and presence or absence of infrastructure conflicts was collected.

Data analysis

Per the SES framework, we utilized tree inventory and household survey data to operationalize variables of theoretical significance to the outcome of interest, tree species diversity at the residential parcel scale. Table 2 displays the variables selected for analysis. First tier variables represented include the Resource System (RS), Resource Units (RU), the Governance System (GS), the Users (U), and Outcomes (O). Within the first tier variable, RS, two second tier variables were included: the size of the system (RS3) interpreted as parcel size and the human constructed facilities (RS4) interpreted as buildings on parcels. Together, these variables produced an approximation of "plantable area," needed to determine tree density.

Table 2. The Social-Ecological System (SES) variables (per Ostrom 2009) included in analysis of private residential parcels in Bloomington.

First Tier Variables	Second Tier Variables	Variable Operationalized	Source
Resource System (RS)	Size (RS3) & Human constructed facility (RS4)	Plantable area (m^2 / acres)	House footprint/Size of parcel (City shapefiles)
	History of system (not in SES framework)	Age of parcel (years)	Monroe County Online GIS
Resource Unit (RU)	Number of units (RU5)	Tree density	Trees/Plantable area (Field data/City shapefiles)
Governance System (GS)	Constitutional rules (GS7)	Association type: HOA/NA	City shapefiles
	Operational rules (GS5)	Importance of compliance with city and community rules regarding landscape management	Survey: 1 = Not important, 2 = Somewhat important, 3 = Very important
	Collective Choice rules (GS6)	Perceived ability to solve problems together in community	Survey: 1 = Disagree, 2 = Neutral, 3 = Agree
Users (U)	Number of users (U1)	Number of children in household	Survey (number)
	Socio-Economics (U2)	Resident's education level	Survey: 1 = High school or equivalent, 2 = Some college/technical, 3 = College grad or higher
	Socio-Economics (U2)	Resident's income level	Survey: 1 < $25,000, 2 = $25,001-50,000, 3 = $50,001 – 75,000, 4 > $75,000
	Socio-Economics (U2)	Resident's age category	Survey: 1 < 35 years, 2 = 35-44 years, 3 = 45-54 years, 4 = 55-64 years, 5 > 65 years
	Knowledge of SES (U7)	Resident time at parcel	Survey (years)
	Use history (U3)	Resident's level of tree management (planted and/or removed or neither)	Survey: 0 = no planting or removal, 1 = planting or removal, 2 = planting and removal
	Norms/Social Capital (U6)	Importance of fitting in with neighbors in terms of landscape management	Survey: 1 = Not important, 2 = Somewhat important, 3 = Very important
Outcomes (O)	Biodiversity/Sustainability (O2)	Tree species richness	Field data

History of the system operationalized as parcel age was considered a second-tier variable within RS, but is not a component of the original SES framework[3]. It was included given its relevance in previous research. Within RU, number of units (RU5) was operationalized as tree density. Second tier variables of the GS included constitutional rules (GS7), interpreted as association

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[3] Vogt *et al*. (2012), a manuscript in progress, argues for the addition of this variable.

type (HOA / NA); collective choice rules (GS6), interpreted as resident-perceived community problem solving abilities; and operational rules (GS5), interpreted as resident-perceived importance of city or community landscaping rule compliance. Second tier U variables included several socio-economic (U2) variables: the number of users (U1) (operationalized as number of children in household given its theoretical importance related to family life stage), their use history (U3) (residents' level of tree management activity) and knowledge of the SES (U7) (residents' time at parcel), as well as the importance of norms (U6) (importance of fitting with neighbors' landscaping). Finally, the outcome of interest, species richness, as a metric of urban forest sustainability, was interpreted as biodiversity (O2) operationalized.

Using IBM SPSS Statistics 19 software, we conducted descriptive statistical analysis, chi-square tests and Welch one-way ANOVAs (robust to unequal variances) on the data set to understand the extent to which SES factors varied by parcels in different types of communities: old and new and HOA and NA parcels. Given the high number of independent variables relative to observations, we then used forward and backward stepwise Ordinary Least Squares (OLS) regression utilizing all independent variables represented in Table 2 to reduce the number of variables influencing parcel-scale tree species richness. Ultimately, results from this analysis allowed for simpler manual model construction using sensitivity analysis. In consideration of the influence individual communities (n=14) may have on parcel-scale richness, we followed the OLS model with a mixed model that accounted for the random effect of community (this was not our initial step as we believed that variation by community would be captured by the suite of independent variables included in the parcel-scale OLS regression). In all statistical tests, results were considered significant at $\alpha \leq 0.05$ and marginal when $\alpha > 0.05 \leq 0.10$.

III. Results

Description of sample

Parcels varied in their biophysical, social, and institutional characteristics (Table 3.). In terms of biophysical variables (for all tests, n=104), the average parcel contained 7.19 tree species (standard deviation (s) = 4.99) and 18.18 individual trees (s = 20.84). Plantable area ranged from 152.64 m^2 (0.03 acre) to 2566.44 m^2 (0.62 acre) with an average of 978.24 m^2 (0.24 acre) per parcel. Average tree density per parcel was ~18 trees / 1011.71 m^2 (quarter acre).

Table 3. Summary of SES data by average parcel overall and average parcel by community type: HOA vs. NA and Old vs. New. Data shown are averages and in parentheses, standard deviation, and for ordinal data (indicated by italics), mode and in parentheses, proportion at mode.

Variable Operationalized	N	Average parcel	Average parcel by Association Type		Average parcel by Age Type	
			HOA	NA	Old	New
Tree species richness	104	7.19 (4.99)	7.05 (4.69)	7.33 (5.33)	***8.56** **(5.15)**	**6.18** **(4.67)**
Tree abundance	104	18.18 (20.84)	18.37 (21.71)	17.98 (21.11)	***23.04** **(23.29)**	**14.61** **(18.23)**
Plantable area (m^2)	104	978.24 (556.67)	*****781.44** **(556.03)**	**1182.76** **(482.80)**	*****1234.55** **(694.88)**	**790.29** **(322.72)**
Tree density (m^2)	104	0.018 (0.015)	****0.022** **(0.017)**	**0.013** **(0.012)**	0.017 (0.011)	0.018 (0.018)
Age of parcel (years)	104	24.06 (16.51)	*****17.39** **(9.35)**	**31.00** **(19.34)**	*****40.06** **(12.21)**	**12.33** **(6.00)**
Number of children	98	0.43 (0.82)	0.38 (0.78)	0.48 (0.86)	0.38 (0.69)	0.48 (0.91)
Resident age category (years)	101	55-64 (23.8%)	>65 (33.3%)	55-64 (28%)	>65 (33.3%)	<35 (27.1%)
Resident time at parcel (years)	103	9.34 (8.96)	8.59 (7.84)	10.11 (9.91)	*****13.88** **(10.95)**	**6.10** **(5.26)**
Resident's tree management level	103	Planting + removal (38.8%)	Planting + removal (20.4%)	Planting or removal (20.4%)	Planting + removal (21.4%)	Planting or removal (26.2%)
Highest level of education attained	101	College grad or + (78.2%)	College grad or + (82.4%)	College grad or + (74%)	College grad or + (76.2%)	College grad or + (79.7%)
Income category (dollars)	89	75,000+ (32.5%)	****75,000+** **(43.1%)**	**25-50,000 & 50-75,000 (both 26.6%)**	75,000+ (36.11%)	25-50,000 & 75,000+ (both 30.1%)
Importance of rule compliance	103	Somewhat (38.8%)	*****Very** **(26.2%)**	**Somewhat** **(20.4%)**	Somewhat (48.8%)	Very (40.0%)
Importance of norms	104	Somewhat (46.2%)	****Very** **(24%)**	**Somewhat** **(24%)**	Somewhat (52.3%)	Somewhat (41.7%)
Community supports collective problem solving	102	Agree (37.3%)	***Neutral** **(17.6%)**	**Agree** **(21.6%)**	****Agree** **(44.1%)**	**Disagree** **(55.8%)**

Note: Significant differences are bolded values and a (*) at p≤ 0.05, (**) = p≤ 0.01, (***) = p≤ 0.001.

In terms of "user" characteristics, the average household had 0.43 children (s= 0.82, n=98) and the average respondent ranged in age from 55-64 years old (n=101). On average, respondents had lived at the parcel inventoried for 9.34 years (s = 8.96, n = 103) and most (38.8%) had removed and planted a tree on that parcel. Of 101 respondents, 78.2% reported their highest level of education as college graduate or higher, while 17.8% had completed some college or technical

training, and 4% had completed high school or equivalent. Fifteen respondents (14.4%) declined to report household income, while of the 89 respondents that did, 32.5% made $75,000 or greater, 24.7% made between $50,000 and $75,000, 29.3% made between $25,000 and $50,000, and 13.5% made below $25,000.

In terms of governance and institutions, respondent households (n = 103) reported that compliance to city and community rules was somewhat important (38.8%) or very important (35.9%) in terms of their landscaping, but a full 25.2% felt such rules were not important in maintaining landscaping. The importance of neighborhood norms for parcel-scale landscape management varied among respondents (n = 104). Norms were very important for 34.6% households, somewhat important for 46.2%, and not important to 19.2%. Of 102 respondents, 33.3% did not agree that their community supported collective problem solving, 26.5% were neutral on the topic, while 37.3% agreed.

Households were split between NAs (n= 53) and HOAs (n= 51) which subjected them to different institutional arrangements. As expected, HOAs were more institutionally formalized than NAs in our sample, all having a formal governing board with rule-making and sanctioning power over private parcels. CCRs existed for all HOAs in the sample, and were utilized as community by-laws which addressed both collective-choice and operational rules of the communities. For all HOAs, rules required a governing board and determined its functions, as well as operational rules in terms of board and home-owner responsibilities. Common operational rules were related to maintaining a particular community aesthetic; most required approval of parcel-scale changes or additions to built structures, and many simply restricted potential "visual negatives" including open garage doors or outdoor laundry lines. The extent to which these by-laws governed tree management was minimal in our sample. HOA rules generally required the removal of dead or dying trees, avoidance of planting within infrastructure easements, and pruning trees to avoid obstructing sight-lines at intersections of roads.

Compared to HOAs, NAs sampled were institutionally informal. Only one NA within our sample was determined to have CCRs and the only tree management related rule within those requirements stated that no tree was to be planted in the PROW of the community. Given that

this NA had no governing board with formal sanctioning power and had numerous trees in the PROW, its CCRs were considered largely unenforced.

Suggestive of these institutional differences by association type, HOA households compared to NA households were significantly more influenced by operational rule compliance and community norms in terms of their landscaping choices, and were also less agreeable that their communities supported collective problem solving (Table 3). Although there was no significant difference between the two association types in terms of parcel-scale tree abundance or species richness, HOA parcels had significantly higher tree density and significantly smaller plantable areas than NA parcels. Despite attempting to control for average age across parcels of differing association types, NA parcels were significantly older than HOA parcels. In addition, HOA households had significantly different income distributions than NA households with a far larger proportion of households falling in the highest income bracket.

The division of parcel data by old (n=44) and new (n=60) communities resulted in significant differences in SES factors (Table 3). The average old and new parcel differed significantly in terms of tree species richness, with older parcels having on average 8.56 (s = 5.15) species compared to 6.18 (s = 4.67) species on new parcels. Tree abundance was significantly higher for old parcels with 23.04 trees on average (s = 23.29) compared to new parcels with an average of 14.61 trees (s = 18.23). While tree density was not significantly different between age categories, plantable area was significantly higher for older parcels. Additionally, resident time at parcel was significantly greater for old parcels compared to new parcels and chi-square tests demonstrated significantly more agreement by older parcels that their community supports collective choice opportunities.

Regression Analysis

Regression analysis produced a significant model (F = 56.61, p < 0.001) that explains 72% of the variation in parcel-scale tree species richness (in natural log form) with four independent variables (Table 4) of the twelve available (Table 2). Tree density, parcel age and resident time at parcel positively influenced species richness while the importance of rule compliance negatively influenced richness.

Table 4. Forward stepwise regression model of tree species richness (transformed to its natural log for normality) of private residential parcels in Bloomington, Indiana communities.

Parameter	Unstandardized Coefficients		Standardized Coefficients	t	Significance (p-value)
Model Summary Adjusted R^2= 0.72 Root Mean Square = 0.42 F= 56.61 p < 0.001					
	Beta	**Std. Error**	**Beta**		
Intercept	-.065	.249		-.263	.793
Tree Density (ln)	.576	.046	.742	12.642	.000
Parcel Age	.009	.003	.191	3.040	.003
Importance of rule compliance	-.178	.059	-.177	-3.040	.003
Resident Time at Parcel	.013	.005	.144	2.297	.024

Because parcels fell within 14 different communities, we also produced a mixed model utilizing only the significant independent variables from the OLS regression while accounting for community as a random effect (Table 5). Results of this model demonstrate that the same SES factors are considered significant or marginally significant when accounting for variation at the community scale except for resident time at parcel. Parameter estimates between the two models' common variables were quite similar, but the covariance parameter in the mixed model finds that individual community is a significant driver of parcel tree species richness.

Table 5. Mixed model regression results of tree species richness on private residential parcels in Bloomington, Indiana communities.

Fixed Parameter	Estimate	Std. Error	Degrees of freedom	t	Significance (p-value)
Intercept	-.177	.250	77.212	-.711	.479
Tree Density	.597	.039	97.717	15.225	.000
Parcel Age	.008	.004	26.046	1.961	.061
Importance of rule compliance	-.128	.050	96.179	-2.556	.012
Resident Time at Parcel	.006	.004	94.912	1.406	.163

Co-variance Parameter		Estimate	Std. Error	Wald Z	Significance (p-value)
Residual		.111	.017	6.557	.000
Neighborhood	Variance	.067	.034	1.970	.049

IV. Discussion

A key to interpreting our results is an understanding that humans ultimately decide the pattern of tree cover in urban areas through active and passive decisions (Zipperer *et al*., 1997) and that those decisions are influenced by incentives which are altered by institutions at multiple scales (Ostrom, 2005). Human decisions influenced by such institutions result in a patchy distribution of planted, emergent and remnant vegetation (Zipperer *et al*., 1997). We recognize that there is great variation in our sample in terms of the mix of these vegetation types as well as the social and institutional factors at play. Thus, to interpret our findings we must not only rely on ecological theory to explain remnant and emergent tree structure, but developing urban ecological theory and social and institutional theory to link structural outcomes to human decision-making.

Given the broad array of theory from which we draw, the SES framework proved useful in structuring this analysis. A number of frameworks in urban ecology and urban forest management aim to account for dynamics that govern social-ecological system outcomes but the prevalence and clarity of "institutions" varies among them. Those that include them frequently embed them as minor exogenous factors or broadly define them as "dynamic solutions to universal needs, including health, justice, faith, commerce, education, leisure, government, and sustenance" (Grove, 2009, 288; Machlis *et al*., 1997), or inclusive of "kinship, economy, religion, polity, governance, and education" (Beddoe *et al*., 2009). Such broad definitions make institutions difficult to operationally incorporate into empirical analysis. Alternatively, a rich history of institutional analysis in social-ecological systems supports the SES framework offering an ideal extension to "institutionally thick" (per Hardy and Koontz, 2010) urban systems.

Variation in parcels by community type and age category

Our results related to variation in community parcels by association type and age category demonstrate not only the importance of these community types in consideration of parcel-level tree structure, but also, their covariance. The fact that we attempted to control for age distributions of communities between association types but still found a significantly younger

population of parcels in HOAs compared to NAs is somewhat confounding but not unexpected. We struggled to identify older HOAs within city limits, likely due to their recent popularity in Bloomington which reflects a larger trend; abundance of HOAs in the U.S. grew from a few hundred to several thousand in the 1960s and has continued to increase in recent decades (Stabile, 2000; Low, 2003).

The physical form of residential parcels varied greatly by association type and age of communities; the significantly smaller size of HOA parcels and new parcels in terms of plantable area, and the significantly higher tree density for HOA parcels is not surprising. According to CCRs, most HOAs in our sample contained common land, owned and managed by the HOA. The inclusion of common land within HOA communities may reduce overall parcel size as we found significantly smaller parcels within HOAs compared to NAs, although we also noted significantly smaller parcels in new communities compared to old (data not shown). Larsen and Hall (2008) report that older parcels in the Phoenix area are larger due to increasing demand (and price) of land over time, and that with smaller parcel size, the recreational value of landscaping shifts from yards to common areas in HOAs. Based on our data, the same appears to be true for Bloomington communities; thus, we conclude that higher tree density on private parcels in HOAs compared to NAs is related to significantly smaller area because there is no significant difference in tree abundance between association types. Although new parcels have significantly smaller plantable areas (and parcel area), their tree density was not significantly different from older parcels which appears to be driven by significantly fewer trees on new parcels.

The variation in household social characteristics by association type and age category was limited, while education and income were skewed toward relatively high values, likely linked to the presence of a large university within the city. However, the variation observed was not unexpected. Older parcels were certainly more likely to house residents whose tenure at the parcel was significantly longer than households at new parcels. Likewise, more tree management on older parcels, although not significantly different statistically from new parcels, was expected given that the presumed older tree populations would be more likely to require tree removal and subsequent planting. Significant differences in the income distribution of households by association type was somewhat expected given that, unlike NA households, HOA households are

required to pay association fees for maintenance of common property, a likely financial barrier to residence for those with lower incomes.

Institutional variation by association type and age category largely reflected the relative abundance and type of institutions to which each was subjected. Given that all parcels were subject to city rules, and HOA parcels were subject to additional CCRs, it is not surprising that rule compliance was very important to a significantly larger proportion of HOA versus NA respondents. A greater threat of sanctions exists for HOA parcels, which we reason would incentivize the importance of rule compliance. The greater importance of norms to households within HOAs is likely linked to the presence of CCRs and development legacies. Ostrom (2005) argues that in order for norms to be important to a community, "all households must share the knowledge that all the other households are [following the norms]" (p. 123). The common physical structure expected of HOA parcels cannot be entirely codified in rules but is partially present from developer legacies, supporting a sense of normative landscaping. Moreover, the presence of rules within HOAs likely establishes a sense of shared responsibilities to institutions in general. Cooter and Ulen (1996) explain that shared norms play an important complimentary role to community rules.

Interestingly, significantly divergent levels of agreement regarding the opportunities afforded for collective problem solving were noted for both association and age type. Although formal institutions establish collective choice arrangements within HOAs, including regular board meetings, the informal institutions within NA communities appear to provide more opportunities for community problem solving, according to our survey data. One possible explanation for this is that such "opportunities" do not require formalities which may even hinder collective problem-solving. For instance, a collective choice requirement that a particular proportion of a community be present to vote on an issue, a common collective choice rule in our HOAs, may hinder resolution if attendance is not met. NAs, without strict collective choice rules may be more flexible to make such arrangements at the convenience of households. This is not to suggest a superior strategy, as formal collective choice arrangements establish predictable structure that can, theoretically, offer alternative benefits to flexibility, such as equitability. However, this result aligns with theory that formality is not a prerequisite of successful collective action (Gibson *et al*., 2005).

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Alternatively, the influence of the confounding factor related to age of community may explain this outcome as older communities were more likely to agree while newer communities were more likely to disagree that their respective communities provide opportunities for collective problem solving. Ostrom (2005) explains that individuals learn from experiences, that learning is enhanced when situations are repeated through improved understanding of others' strategies which facilitates convergence of individuals' mental models. Presumably, the older a community, the more chance that repeated interactions have taken place between neighbors allowing for convergence toward problem solving.

Predicting tree species richness

The degree to which these variations in the biophysical, social and institutional characteristics of households influence species diversity is demonstrated in our regression models. Perhaps most interesting is the absence of income, a theoretically important variable. Although important for explaining species diversity of perennial vegetation in front yards of Phoenix residences, household income did not constitute a significant variable in our OLS model of parcel-scale tree species richness. We believe this result may be attributable to the difference in sampling procedure via taxa included and/or spatial extent of sampling. Our tree census data represented full parcels, while the "luxury effect" was proposed based on front and side yard perennial plant diversity (Hope *et al*., 2003). Because Larsen and Harlan (2006) found significant differences in front and back yard vegetation structure and predictors, we believe full parcel-scale data, with greater attention to trees alone, are required to further address this theory.

Tree density as a positive predictor of tree species richness was an important finding in that it considered tree abundance while accounting for parcel area. It is not necessarily surprising that the more trees a parcel has, the more species found. It is interesting, however, because it stands regardless of parcel area. Area is theorized to have a positive effect on species richness through, among other mechanisms, the increase of ecological niches in natural landscapes (Darlington, 1957; Preston, 1962). However, our results demonstrate that this does not strictly apply to urban residential parcels, most certainly due to the anthropogenic input/extraction of trees. But, this also suggests that human preferences tend toward species diversification in parcel-scale tree

planting, regardless of parcel size. In fact, research suggests this link; according to Galliano and Leoffler (1999), landscapes with high diversity have some of the greatest scenic value for people.

The finding that parcel age positively predicts tree species richness falls in line with findings from Baltimore, given significant positive bivariate correlations between species richness, tree abundance, and parcel age in our sample (data not shown). If we assume development of urban residential parcels generally results in clearing of vegetation, the more trees we would expect to find with increasing time since development due to increasing probability of both natural recruitment of tree seedlings and anthropogenic plantings over time. We suggest the link to species richness may be as simple—more time increases the probability that a new species will either be planted or naturally recruited, increasing richness. In fact, Porter *et al.* (2001) found species diversity peaked in urban residential areas along a urban-to-rural transect due to the introduction of exotic and ornamental species *planted* by residents coupled with the representative native tree population (assumed to be remnant or emergent in nature).

However, ecological theory suggests that, through succession, species richness peaks at some intermediate time point after disturbance (Grime, 1973). For instance, after land clearing, pioneer species dominate; at an intermediate point in time, richness peaks as pioneer species coexist with later successional species, after which, richness decreases as later successional species outcompete pioneer species. Thus, we further suggest that our parcels' maximum age (52 years) may represent an intermediate time since disturbance given the long-lived nature of trees. Alternatively, we recognize that homeowners play a significant role in staving off natural successional dynamics (Zipperer *et al.*, 1997), and may actually be more likely to facilitate increased richness to a point, after which richness remains relatively steady. Of course, a broader range of parcel ages within our sample would be required to test such a hypothesis, but Kaplan and Kaplan (1989) review "experimental aesthetics" research related to psychological preferences for nature and report that people prefer intermediate complexity in visual patterns which seems relatable to the potential for some preferential, intermediate complexity of tree species.

Further, it is not unexpected that our results related to parcel age appear to fall more in line with those of Baltimore studies, given the climatic similarity between Bloomington and Baltimore. In

fact, Lowry *et al*. (2011) attributed the variation in findings between Baltimore and Phoenix studies to the stark environmental differences between the two cities, and the types of landscaping common in each. This suggests that further studies are needed to tease out the relationship between tree abundance and species richness at the parcel scale, and the relationship of both to parcel age. However, our results are most supportive of theory drawn from research in Baltimore.

While distinction of parcels by association type produced significant differences in terms of multiple SES factors, association type as a proxy for "constitutional rule" did not constitute a significant variable in either model to explain variation in parcel tree species richness. However, operational rule compliance was a significant negative predictor of tree species richness, a surprising finding. We believe this negative relationship is related to the fact that tree management-related rules from both city ordinances and community CCRs examined within this study relate to tree removal and barriers to planting and diversifying trees on private properties. No rules exist related to requiring diversifying tree species or increasing tree abundance. For instance, CCRs and city rules require the removal of dead or dying trees; some CCRs require gaining approval for planting trees or restrict plantings in easement areas or in setbacks; and even city ordinances that encourage preferences for native trees species—meant to reduce the likelihood of non-native exotic spread—reduce the palette of species for planting even within NAs. If rule-compliance is important to a household, it makes sense then that fewer trees and likely fewer species will exist on the parcel regardless of association type. Supporting this argument, a negative correlation between operational rule compliance and tree abundance was noted across the entire data set.

Moreover, this finding represents an important link between institutions and robustness. Sustainability in human-designed social-ecological systems is perhaps better viewed as "robustness," an engineering term that implies that increased sustainability to one parameter may deflate sustainability or robustness to another (Anderies *et al*., 2004). Such appears to be the case in our sample. Rules meant to increase robustness to invasive tree species and to the liability from dead or dying trees (as perceived by homeowners) decreases the robustness of the system to species specific pests and pathogens by facilitating decreased tree species richness.

These findings lead to a potential policy implication of this research—that tree management rules influencing private residential parcels should not only incentivize the avoidance of negative impacts from trees, but the provision of positive structural characteristics. For those households for which city or association rule compliance is important, might rules that incentivize tree planting and diversification have the opposite, positive effect on species richness? We believe our data suggest this possibility. While cities may struggle to gain the support to impose such detailed requirements on private parcels, it is certainly not unimaginable that private HOAs could institutionalize diversity, potentially increase the ecological resilience of tree structure and the robustness of community urban forest management.

Results of the mixed model demonstrate the significance of community-effects on parcel scale tree species richness while discounting the influence of resident time at parcel. We suggest that these influential factors may be related and thus, resident time at parcel drops in significance. In fact, significant differences were determined between communities in terms of resident time at parcel. However, we believe it is important to consider given its potential to influence results in other studies, particularly those interested in further exploring legacy effects on parcel vegetation. Through recent research in Spain, Garcia-Llorente *et al.* (2012) found a strong positive effect between respondents' place attachment and the level of support for non-use values of a landscape. In other words, respondents were more willing to invest in landscapes when their sense of belonging to it was greater. If we assume a sense of belonging to a landscape intensifies with time spent there and that high tree species richness on residential parcels is linked to not only natural recruitment but tree plantings which constitute an investment in the landscape, then we can make sense of the positive influence that resident time at parcel may have on tree species as well as other structural metrics.

However, it is not appropriate to relegate neighborhood effect only to resident time at parcel. In fact, there are a number of unique circumstances at the neighborhood scale that cannot be accounted for in our statistical models which may have important influences on species richness at the parcel scale. For instance, one of the new HOA communities within our sample constitutes a development that included tree preservation easements. For the most part, these easements fall within common property of the HOA, however, for some parcels sampled there, it was clear to

researchers that remnant trees composed a portion of the parcels and heavily influenced the tree abundance and species richness of the parcels and thus, the community. Another unique community case includes a small, new HOA in which multiple homes were outfitted with solar panels. Low tree abundance, density, and richness may be linked to parcel age in the case, but it is also likely that the presence of solar panels requires homeowners to maintain low numbers of trees to avoid future interference of canopy with sunlight. Because we could not account for all these and other unique community-level circumstances in the OLS model, it appears that a portion of the unexplained variance (28%) in parcel-scale species richness should be attributed to the individual communities' respective circumstances, according to the mixed model. Therefore, our data appear to support the importance of neighborhood effects which Sampson (2012) claims remain durable despite social theorists' assertions to the contrary. Thus, future analysis should consider the mechanisms by which individual communities may influence tree species composition.

V.   Conclusion

This research demonstrates to urban ecological researchers the usefulness of the SES framework and institutional analysis and theory to define specific types of institutions and their respective influences on urban ecological outcomes at the residential parcel scale. Moreover, it points to the role of operational rules and their potential for unintended consequences related to determining one important aspect of parcel-scale tree structure involved in sustainable urban forest management, species richness. This is an important finding given our gap in understanding the influence of institutions on residential parcel tree species richness. Further, results demonstrate that association type and development age co-vary with many important SES characteristics, some of which relate to parcel and tree structure.

Because this research demonstrates the importance of city and home-owner association rules in structuring urban forests, it implicates the nature of those rules in sustainable urban forest management. Consideration for additional rules that proactively support sustainable urban forest structure, such as requiring species diversification on parcels, is appropriate and perhaps most effective at the community association scale. Cities such as Bloomington may also reconsider

refining their policy on discouraging non-native species, as it plays a role in the decreased palette of species available for diversification, and alternatively, focus rules on discouraging invasive species.

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Chapter Six

CONCLUSIONS

I.   **Introduction**

Three key messages presented in the United Nation's report, *Cities and Biodiversity Outlook* (UNCOB 2012), include: 1) maintaining functioning urban ecosystems can significantly enhance human health and wellbeing, 2) ecosystem services must be integrated in urban policy and planning, and 3) successful management of biodiversity and ecosystem services in cities must be based on multi-scale, multi-sectoral, and multi-stakeholder involvement. These key messages are reflected in the research presented here. I have argued that urban trees provide many of these ecosystem services, a function dependent upon the structure of the urban forest as a whole. In cities, "the whole" is composed of many small and moving parts—"where people are on the move, preoccupied, and living in heterogeneous, dense, assemblages, often in relative anonymity" according to Nagendra (2012). In many ways, this description indicates the difficulty of urban forest management in which the true managers constitute diverse individual households. And yet, the UNCOB's key messages point to the mechanisms by which these disparate management units can be tied together toward a common goal; institutions and collective action across scales may produce small and moving parts working together as "urban fractals… containing the essential characteristics that we want to see in the whole urban system, including nature, ecosystem services, and urban agriculture," according to Downton (2012).

And yet, despite a rising inclusion of urban forest management institutions in municipal policies and despite increasing collective efforts to "plant a million" and preserve urban trees and green space, urban forests are declining. How and when, then, do these mechanisms—institutions and collective action—work to produce sustainable outcomes for urban forests? The answer to this question, in part, has been the goal of this research. In consideration of top-down policies, I have demonstrated that municipalities must be willing to explore the influence of their zoning policies at fine scales, even those related only to land use, because related variation in urban forest outcomes is evident at disaggregated zone scales. At the level of neighborhoods, I have established the importance of collective action alone, along with institutions structured to support monitoring and sanctioning, to aid in the establishment of community planted trees and in the

ability of communities to continue to work for the common good. Finally, I have contributed to the understanding that at the finest scale—the urban parcel—not only do biophysical and social factors structure urban forests, but so do association and city rules which can have unintended consequences when efforts to produce robust systems yield tradeoffs.

This final chapter summarizes the findings in greater detail and suggests important implications from this research. I consider both the theoretical and practical implications, followed by the limitations of each project. Finally, I conclude by considering the research questions that arise and constitute next steps in a continued research agenda in sustainable urban forest management.

II.   Summary of findings and implications

Chapter 2: Institutional analysis in urban ecosystems

The theoretical analysis presented in Chapter 2 argues for linking urban forest management and urban ecosystem research to frameworks established in new institutional economics, including the Institutional Analysis and Development Framework (Kiser and Ostrom, 1983), the precursor to the SES framework (Ostrom, 2009). This article demonstrates through the application of the IAD framework the usefulness of this approach for exploring the role of institutions in driving urban forest outcomes, a necessary approach given the lack of focus on institutions in urban ecosystems research and the recent call for a policy-based research agenda in urban forest management (Wolf and Kruger, 2010). Further the article argues that institutional and collective action theories developed in rural community resource systems are relevant to urban community resource management, and should be tested in cities.

Implications of this work are beginning to unfold. Beyond the usefulness of Chapter 2 for subsequent research presented within this dissertation, the article itself has been submitted to *Urban Ecosystems* and was recently accepted there contingent upon revisions. Upon its publication, this article promises to introduce terminology that will allow for expanded cross-disciplinary fertilization between rural and urban resource management scholars. After all, Ostrom (2009) argued, "Understanding of the processes that lead to improvements in or deterioration of natural resources is limited because scientific disciplines use different concepts and languages to describe and explain complex social-ecological systems (SESs)" (419). By

explicitly demonstrating how frameworks utilized largely by social scientists can contribute to urban ecology and urban forest management research, this work begins to bridge such a gap between groups of scientists that arguably have much to share once a common language is established.

Practical implications of this work are unfolding, as well. This article was a starting point for other research that has evolved from the Bloomington Urban Forestry Research Group (BUFRG) at Indiana University that is now contributing to a national effort developed by the Urban Tree Growth & Longevity (UTGL) Working Group. The UTGL is currently drafting a framework for an urban tree monitoring protocol which promises to:

1. Provide technical guidelines for long-term data collection in urban forests, ensuring that data generated by monitoring programs is scientifically rigorous;

2. Provide urban forestry practitioners with methods for monitoring that can be adapted to local management needs and organizational capacities, ensuring that the data generated is useful to practitioners;

3. Develop a national network of local urban forestry organizations collecting longitudinal data in the same manner, which will facilitate comparisons across programs and cities.

The framework being drafted relies on the concepts addressed in Chapter 2 of this dissertation and multiple conference presentations by members of BUFRG; the UTGL plans to cite our work (pointing to the IAD and SES framework) in demonstrating the importance of the biophysical, socioeconomic, and institutional factors that impact the growth and survival of urban trees. Thus, this chapter has contributed to a national monitoring effort that will likely have long-term implications for framing future research in urban forest management.

The major limitation of this article is that the application of the framework we propose is demonstrated by a *hypothetical* example of street tree management. In fact, the *Urban Ecosystems* reviewer of the article suggests her interest in how the framework could handle private residential tree management scenarios. Since this article was submitted, we have conducted data collection and analysis on private residential parcels in Bloomington; a revision

of this text may offer the opportunity to address this limitation by applying the IAD framework to actual data from private parcels.

Chapter 3: Municipal zoning institutions and canopy cover

The research presented related to zoning and canopy cover demonstrates the importance of land cover institutions and the scale at which they are applied and analyzed, continuing the dissertations emphasis on expanding the study of institutions in urban SES outcomes. For scholars, the work demonstrates the variation in existing, potential, and relative canopy cover by zone typologies following the research recommendation by Kenney *et al*. (2011) to move beyond simply examining existing canopy cover. Our use of relative canopy cover demonstrates the emergent properties of institutions in urban ecosystems in terms of the active and passive decisions of land managers in regard to how much potential canopy area is allowed to support tree cover. Overall, we found that conclusions drawn differ according to scale of analysis and therefore partly counter the oft-cited conclusion that "residential lands" have the most canopy cover. Specifically, high density residential lands had significantly lower canopy than other residential lands which refines theory and suggests the importance of the fine-scale analysis of zoning typologies. This research also allowed us to conclude that we see a tradeoff in canopy cover within PUD zones, a form of new urbanism in Bloomington. Few scholars have examined the influence of mixed-use design on urban tree canopy, thus this work contributes to a growing body of knowledge related to an increasingly popular form of urban development.

The implications of this work are not only in expanding theory, but in its application. For instance, we argue that urban forest managers must consider zoning's influence over the urban forest even if zoning does not directly regulate trees; policies that influence impervious cover appear to influence existing canopy by affecting potential canopy area. More importantly, fine-scale variation in zoning should be considered when institutionalizing canopy cover goals for a city; why set a goal for a particular zoning designation if it is unlikely to be met due to constraints in the form of potential cover? Moreover, this work suggests that urban planners may wish to consider the spatial distribution of zoning at fine scales as it will likely influence the existing, potential, and relative canopy cover of those lands and therefore the distribution of ecosystem services. Further, we suggest that urban planners need to consider not only the

benefits of mixed-use design for sustainable cities, but also the potential drawback in relatively lower canopy cover. The potential of this work to influence practitioners is more likely because the article has been submitted to *Urban Forestry and Urban Greening* and accepted contingent upon revisions.

The limitations of this work are largely related to the lack of parcel-scale data. As stated in the article, canopy cover analysis at the parcel scale while controlling for zone district and type would have yielded further information. Moreover, parcel scale analysis would allow for consideration of the relationship between zone districts/types and age of development which is likely an additional determinant of canopy cover. However, the focus of the paper was not to suggest that zoning is the most important driver of canopy cover, but one that may shed light on variation in canopy across zones.

Chapter 4: The impact of watering strategies on trees and communities

Analysis of watering strategies of community planted trees in Indianapolis NeighborWoods concluded that tree survival was greatest in communities that collectively watered, used signed watering agreements, monitored watering, and where monitoring and sanctioning changed watering behavior. Likewise, tree growth was greater where signed watering agreements alone or in combination with collective watering were used and where monitoring and sanctioning changed behavior. These effects were attributed to more consistent and thorough watering inferred through interview data. Finally, collective activity post-tree planting and management was greater for communities that collectively watered.

This data clearly supports collective action theory and extends it to urban forest management scenarios. Namely, the results fall in-line with the theory that collective action has higher returns than individuals acting alone as argued by Ostrom (1990). Additionally, the results suggest that at least some of the Design Principles (Ostrom, 1990) may help support long-enduring community urban forest management when applied within neighborhoods and at higher-scale institutional contexts of facilitating organizations such as KIB. Practically, the implications of this research are likely to impact the strategies suggested by nonprofits to communities that wish to improve their own planted tree survival. However, as we suggest in the paper, there are no panaceas (Anderies *et al.*, 2007). We hope that the research itself will be shared with

communities to allow them to use the information in devising the management strategies that work best for their particular circumstances. Because this article was submitted to *Arboriculture and Urban Forestry* and was recently revised and resubmitted, its impact is likely broadened from simply influencing the practices of KIB to other non-profits and organizations focused on community tree planting.

Limitations of this research are largely related to the inferred link between watering institutions and tree outcomes; based on interview data, we infer that the link is the consistency and thoroughness of watering. We cannot, however, prove that watering strategy affected watering behavior, nor that watering behavior directly affected tree outcomes. Future research that addresses similar questions would be improved by seeking data related to causal mechanisms such as the consistency and thoroughness of watering, perhaps by utilizing water monitors (people in neighborhoods) or instrumentation for such data collection.

Chapter 5: The relative influence of social institutions on residential urban forest structure

The negative influence of institutions in the form of operational rules (that largely addressed tree removal) along with the positive influence of tree density and parcel age were demonstrated on parcel-scale tree species richness, a measure of diversity, in Chapter 5. The OLS regression also established the positive influence of resident time at parcel, while the mixed model alternatively indicates the importance of neighborhood-context in influencing parcel-scale tree species richness. More generally, we found that parcels varied in other SES characteristics by association type and development age.

This chapter demonstrates to urban ecological researchers the usefulness of the SES framework and the importance of institutional analysis and theory to define various types of institutions and their respective influences on urban ecological outcomes. It points to the role of operational rules in predicting tree species diversity, an important metric for sustainable urban forest management. More specifically, the article demonstrates that operational rules may have unintended consequences; a tradeoff between robustness to invasive species or liability from perceived dead/dying trees appears to result in a tradeoff with robustness to species-specific pests and pathogens (by reducing tree species diversity). Further, results demonstrate that constitutional rules co-vary with many important SES components related to the parcel-scale, many of which

appear to influence other aspects of parcel and tree structure. These are novel findings in urban systems particularly given our full-parcel tree inventory data and the fact that little research addresses the influence of institutions on private residential parcel vegetation. Thus, this paper suggests the relative importance of institutions and a framework and theory that support continued research in that regard.

Policy implications from this research are related to the importance of city and home-owner association rules in structuring sustainable urban forests. While the influential institutions at play in this study are not considered incompatible with sustainable urban forest structure, it seems that their overall focus on tree removal and limiting tree species has negatively impacted diversity. Therefore, consideration for additional rules that require diversification on parcels is appropriate and perhaps most effective at the home-owner association scale. Cities such as Bloomington may also reconsider their stance on encouraging the planting of only native species (as it may play a role in the decreased palette of species available and due to its controversy in the literature) and instead focus institutions more specifically on discouraging invasive species.

The primary limitations of this research are related largely to the research design. Neighborhoods were determined to be an important factor through mixed-model analysis, but could not be considered alongside other SES variables within the OLS model given that the inclusion of 13 neighborhood dummy variables would require a much larger sample size than 106 parcels. Further, with only 14 neighborhoods, modeling neighborhood tree structure was limited. Additionally, parcel samples within neighborhoods were fairly small due to constraints on time and budget thus making it somewhat difficult to characterize each neighborhood and link unique institutional arrangements to outcomes. Therefore, some generalization of institutions by association type was necessary for analytical purposes.

III. Future research

Short term research goals

A number of research questions arise from this dissertation that constitute a future research agenda. First, and likely most tangible, is further exploration regarding canopy cover and the influence of zoning; specifically, do our conclusions hold at the parcel scale? In order to answer

this question, we must either sample parcels that fall within zoning boundaries or work to correct the misalignment between zoning and parcel boundaries. The former strategy could be effective and given its relative simplicity, will likely constitute a next step.

Of likely greater significance are the subsequent questions that arise from research in Indianapolis NeighborWoods: As we increase our sample size, do our conclusions hold? What exactly is the mechanism behind improved tree survival and growth in collective watering scenarios? What exactly is the mechanism behind increased collaboration in communities that collectively water? The opportunity to answer these questions will be facilitated by the recent receipt of a National Urban and Community Forestry Advisory Council (NUCFAC) Grant to the Bloomington Urban Forestry Research Group to expand this study to five additional cities across the Eastern U.S. This will certainly increase our sample size in order to test whether our current conclusions hold. In order to address the link between watering strategies and biophysical outcomes, we have considered the possibility of studying neighborhoods in which trees were more recently planted (and still requiring watering) and seeking watering data through either field collection by neighborhood watering monitors, or through instrumentation to measure soil moisture over time. In order to best address the link between neighborhood collective watering and subsequent collective action, we plan to survey communities rather than relying solely on interview data. Survey data will seek to understand the perception of participating residents as to their neighborhoods' ability to work together before and after tree planting and management. We will explore the mechanisms behind these perceptions, seeking to understand the variation in trust and reciprocity among community members at both time points.

The data collected from residential parcel in Bloomington HOAs and NAs will facilitate a great deal of additional research. Another manuscript is currently under revision which addresses the carbon content of trees and soils across the sampled parcels. This is an important undertaking as little research has addressed the above *and* below ground carbon content of residential parcels. Additionally, a wealth of survey data leads to numerous research questions: What factors motivate households in the management of their trees and vegetation? Do those motivations differ by community association type or neighborhood? How do motivations relate to the physical properties measured on the parcels? Exploratory statistical analysis is underway to begin to answer these research questions.

An additional data set gathered this summer will complement the private residential data set. Public right of way soil samples were collected under street trees, which were re-inventoried in the same HOAs and NAs sampled for the research addressed in Chapter 5. The natural extension of this work is to compare public and private data, asking the questions: How do urban trees and associated soils vary by public and private property divisions? How do structure and function vary by property type, and what are the implications of this for sustaining urban forests?

Long term research goals

In the long-run, an important research agenda for sustainable urban forest management is comparative in nature. To build on the objectives of this dissertation, it will be important to continue to understand how theory developed in rural community resource management relates to urban settings. For instance, I hypothesize that one important difference between rural and urban forest management is the salience of the resource. Timber and non-timber forest product markets are arguably important in rural forest management. Such markets do not necessarily exist for the myriad of ecosystem services provided by urban trees even though significant monetary benefits can be attached to those services (e.g., energy saving from shade, etc.) . Thus, do urban dwellers perceive the importance of urban trees? Is there a difference between the values that urban and rural residents attach to their properties' trees?

Expanding this work to different socio-economic and political settings for comparison will further shed light on the role of institutions in sustaining urban trees. Colleagues familiar with the IAD and SES framework and related institutional theory have explored the influence of institutions on street tree structure in Bangalore, India (Nagendra and Gopal, 2011). A promising research project would compare the role of social, economic, and political variation in urban forest management between Bloomington and Bangalore. Furthermore, the NUCFAC grant will offer years-worth of comparative urban forest data from six cities across the U.S. An important question that was not possible to explore in Indianapolis, working with only one non-profit, is: what is the role of the nonprofit in sustaining urban forest management within communities? How do cities, non-profits, and neighborhoods work polycentrically to coproduce the urban forest? As suggested in Chapter 4, the institutional context of KIB follows several design principles (Ostrom, 1990). The variation in management of these community tree planting

programs may offer important insight into their role in successful urban tree management. Additionally, this large data set will offer insight into the role non-profits, or the third sector, play in adaptive management to climate change, likely a key focus of future research.

IV. Conclusion

As urbanization and associated clearing of rural forests and agricultural land increases, and as cities struggle to maintain urban trees as development infill progresses, the problems associated with growing urban populations are increasingly clear. At the same time, opportunity exists. If ecosystem services that support global processes are lost to expanding cities, the necessity of compensation from functional urban forests grows. As rural populations move to cities, the need for local ecosystem services from resilient and equitably distributed urban trees expands. There is no longer any question as to the importance of sustainable urban forests for their functions to which humans are indebted.

The questions that remain are related to understanding the strategies that communities use to undertake sustainable, robust urban forest management. While the metrics of structure that yield function are relatively well known, the ways in which communities work together and incentivize individual and emergent action for sustainable structure is less well known. Applying frameworks and lessons learned from rural resource management to urban systems is an important line of inquiry that promises to improve our understanding of the mechanisms that produce sustainable urban forest solutions. Perhaps more importantly, this cross-disciplinary fertilization holds the potential to find urban ecosystem solutions more rapidly than urban researchers might otherwise find, a salient concern in a rapidly urbanizing world facing a cascade of new and emergent problems associated with global change. This research portfolio represents one step in that direction, identifying key institutions, collective action, and the characteristics of each that contribute to sustainable structure and function of urban forests.

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Education

Indiana University School of Public and Environmental Affairs (SPEA)   **Bloomington, Indiana**
- ◦ Doctor of Philosophy in Environmental Science (GPA: 3.88)   December 2012
- ◦ Master of Public Affairs in Environmental Policy, Natural Resource Management   December 2007
- ◦ Master of Science in Environmental Science, specialized in Forest Resources   December 2007

Morehead State University   **Morehead, Kentucky**
- ◦ Bachelor of Arts in English *(major),* Environmental Science *(minor)*   May 2003

Ph.D. Research Committee
- ◦ Burnell C. Fischer, PhD., Committee Chair, Clinical Professor – Indiana University, SPEA
- ◦ J.C. Randolph, PhD. Professor – Indiana University, SPEA
- ◦ Tom Evans, PhD. Associate Professor – Indiana University, Department of Geography
- ◦ Michael Cox, PhD. Assistant Professor – Dartmouth College, Environmental Studies
- ◦ Catherine Tucker, Ph.D. Associate Professor – Indiana University, Department of Anthropology
- ◦ (Posthumously) Elinor Ostrom, PhD. Distinguished Professor – Indiana University, Political Science/SPEA

Professional Experience

Indiana University-School of Public and Environmental Affairs, Bloomington, Indiana
Visiting Professor, 2013
- ◦ Primary instructor for two sections of E162 Environment and People, spring 2013
- ◦ Co-instructor for V600 Masters Capstone Course, spring semester 2013

Associate Instructor, 2010-2012
- ◦ Primary instructor for E332/532 Introduction to Applied Ecology, fall semesters 2010, 2011
- ◦ Co-instructor for V600 Masters Capstone Course, spring semester 2012
- ◦ Co-instructor for E555 Urban Ecology, fall semester 2010

Teaching Assistant, 2008-2010
- ◦ E422/E522-Urban Forest Management, spring semesters 2008, 2009, 2010
- ◦ E555-Sustainable Forestry, fall semester 2008

Graduate Assistant, Graduate Programs Office, 2007- 2008
- ◦ Coordinated Graduate Admissions and Financial Aid, spring and summer 2008
- ◦ Assisted Director of Student Services with recruitment research, communication, event planning

Service Corps Community Coordinator, 2006-2007
- ◦ Coordinated the professional work of 30 Graduate Student Fellows in 18 nonprofit agencies
- ◦ Developed and managed biweekly professional development seminars for Fellows
- ◦ Designed and directed four service activities for Fellows that supported local nonprofit agencies

Professional Experience (continued)

AmeriCorps, Morehead, Kentucky

Rowan County Middle School Youth Services Center Literacy Coordinator, 2004-2005

◦ Instituted literacy development program and tutored twenty at-risk youth for one academic year

◦ Taught weekly sessions in after-school program including theatre, reading, Girl Scout troop

◦ Supported Youth Services Administration including program development and communications

Publications

Cox, M., **S.K. Mincey,** T. Ruseva, S. Villamayor-Tomas, B. Fischer. 2013. Evaluating the USFS state & private forestry redesign: A first look at policy. *Ecological Economics* 85: 35-42.

Mincey, S.K and J. Vogt. **In review (revise and resubmit).** The influence of watering strategy on the survival and growth of community-planted trees. Submitted to *Arboriculture and Urban Forestry.*

Mincey, S.K., M. Schmitt-Harsh, R. Thurau. **In review (revise and resubmit).** Zoning, land use, and urban tree canopy cover: The importance of scale. Submitted to *Urban Forestry and Urban Greening.*

Epstein, G., J.M. Vogt, **S.K. Mincey**, M. Cox, and B.C. Fischer. **In review.** Towards an interdisciplinary social-ecological systems framework. Submitted to the *International Journal of the Commons.*

Mincey, S.K., M. Hutten, B.Fischer, T. Evans, S. Stewart, J. M. Vogt. **In review (revise and resubmit).** Structuring institutional analysis for urban ecosystems: A key to sustainable urban forest management. Submitted to *Urban Ecosystems.*

Fleischman, F. D., K. Boenning, G. A. Garcia-Lopez, **S. K. Mincey**, M.Schmitt-Harsh, K. Daedlow, M. Lopez, X. Basurto, B. Fischer and E. Ostrom. 2010. Disturbance, response, and persistence in self-organized forested communities: Analysis of robustness and resilience in five communities in southern Indiana. *Ecology and Society* 15 (4): 9.

Basurto, X., K. Boenning, K. Daedlow, F. Fleischman, B. Fischer, G. Garcia-Lopez, M. C. Lopez Perez, **S.K. Mincey,** M. Schmitt, E. Ostrom, F. Souza. 2008. The Community and Forests of Painted Hills. International Forestry Resources and Institutions Research Program, The Vincent and Elinor Ostrom Workshop in Political Theory and Policy Analysis Paper, W09I-3, 35p.

Arnold, G, J. Bauer, B. Fischer, F. Fleischman, M. Midgley, **S.K. Mincey,** M.P. Priebe, K. Souzq, C. Tucker, S.V. Tomas, 2007. An initial visit to the Yellowwood Lake Watershed Forest. International Forestry Resources & Institutions Research Program, The Vincent and Elinor Ostrom Workshop in Political Theory and Policy Analysis Paper, W07I-24, 31p.

Conference Presentations and Proceedings

Mincey, S.K., M. Schmitt-Harsh, M.S. Patterson, T. Evans, B.C. Fischer. 2012. Analysis of private urban forest management in Bloomington, Indiana: A social-ecological systems perspective. 97th Annual Meeting of the Ecological Society of America, August 10, 2012. Portland, Oregon.

Schmitt-Harsh, M., **Mincey, S.K.,** M.S. Patterson, T. Evans, B.C. Fischer. 2012. Carbon storage and ecosystem services of urban trees in Bloomington, IN as a function of development age. 97th Annual Meeting of the Ecological Society of America, August 10, 2012. Portland, Oregon.

Vogt, J.M., **S.K. Mincey**, and B.C. Fischer. 2012. Evaluating the biophysical and social results of Keep Indianapolis Beautiful's urban tree plantings as a social-ecological system. 97th Annual Meeting of the Ecological Society of America, August 10, 2012. Portland, Oregon.

Mincey, S.K., M. Schmitt-Harsh, M.S. Patterson, R. Thurau, T. Evans, B.C. Fischer. 2012. The relative influence of homeowner attitudes and social institutions on residential urban forest structure. International Society of Arboriculture 2012 Annual Conference, August 13, 2012. Portland, Oregon.

Vogt, J.M., S.K. Mincey, M. Patterson & B.C. Fischer. 2012. Predicting urban tree success: Establishment, maintenance, growth and condition of neighborhood-initiated tree plantings. International Society of Arboriculture 2012 Annual Conference, August 13, 2012, Portland, Oregon.

Vogt, J.M., **S.K. Mincey**, and B.C. Fischer. 2011. Establishment and Growth of Community-Planted Trees in Indianapolis, Indiana. ISA Tree Growth and Longevity Symposium, September 12-13, 2011. Lisle, Illinois.

Fischer, B.C., and **S.K. Mincey.** 2010. Characteristics of sustainable urban forest social-ecological systems: Analysis tools for the local level. Presented at 2010 Partners in Community Forestry National Conference. November 9-11, 2010. Philadelphia, Pennsylvania.

Mincey, S.K., L. Ahern, B.C. Fischer, M. Widhelm. 2010. Planning sustainable urban forest for college campuses. Presented at the Association for the Advancement of Sustainability in Higher Education (AASHE) 2010 National Conference. October 12, 2010. Denver, Colorado.

Mincey, S.K., B.C. Fischer, S. Stewart. 2010. Managing urban forests as commons: Implications for sustainability. The International Society for Arboriculture 2010 Annual Conference, Chicago, Illinois.

Thurau, R., **S.K. Mincey,** M. Schmitt-Harsh, M.E. Cox. 2010**.** Spatial distribution and socio-economic contexts of urban tree canopy cover in Bloomington, Indiana, USA. The Urbanization and Global Environmental Change (UGEC) Conference. October 18, 2010. Tempe, Arizona.

Mincey, S.K. and B.C. Fischer. 2008. Indiana University and Bloomington, Indiana: Partnering for the Urban Forest. Presented at the International Society for Arboriculture 2008 Annual Conference, St. Louis, Missouri.

Mincey, S.K. and B.C. Fischer. 2007. STRATUM (Street Tree Resource Analysis Tool for Urban Forest Managers): An analysis and recent case study from Bloomington, Indiana. Abstracts, Indiana Academy of Science, 123rd Annual Meeting, p. 62-63.

Mincey, S.K. 2007. A fight against mountain top removal coal mining: Female influence on natural resource policy in Appalachia. Presented at the 9th Annual National Conference on Women of Appalachia at Ohio University-Zanesville. October 19-20, 2007. Zanesville, Ohio.

Grants

National Urban and Community Forestry Advisory Council, 2012, "Trees and People, a Two-Way Street: A Research Program to Assess the Direct and Indirect Effects of Urban Tree Planting Programs in the Face of Climate Change" PIs B.C. Fischer, J. Vogt and S.K. Mincey. Project Partners: Alliance for Community Trees (College Park, MD); Keep Indianapolis Beautiful, Inc. (Indianapolis, IN); Pennsylvania Horticultural Society (Philadelphia, PA); Trees Atlanta (Atlanta, GA); Greening of Detroit (Detroit, MI); Forest ReLeaf of Missouri (St. Louis, MO); Trees Forever (Des Moines, IA). **$173,206 awarded (plus $188,365 provided in matching funds).**

USDA Forest Service, Northeastern Area Forest Legacy Program, 2011, "A Review of the USDA Forest Service's Forest Legacy Program: Identifying Organizational Designs that Ensure Conservation Easement Monitoring and Stewardship," PIs B.C. Fischer, S.K. Mincey, J. Farmer, **$15,000 awarded**.

Efroymson Fund and Keep Indianapolis Beautiful, Inc., 2011, "Sustainable Urban Forest Management: Predicting Tree Success in Indianapolis NeighborWoods," PIs B.C. Fischer, S.K. Mincey, J. Vogt., **$17,500 awarded.**

IU Office of Sustainability, 2011, "The Impact of Institutional Mechanisms on Sustainable Urban Development," PI B.C. Fischer, Co-PIs S.K. Mincey, M. Schmitt-Harsh, R. Thurau **$15,000 awarded.**

IU-School of Public and Environmental Affairs, Seed Grant, 2010, "Comparing the design and outcomes of competitive funding programs in federal natural resource agencies," PIs B.C. Fischer, S. Fernandez, S.K. Mincey, M.E. Cox, S. Villamayor-Tomas, T. Ruseva. **$20,000 awarded**.

IU-Center for Research in Environmental Science, Research Grant, 2009, "The influence of institutional dynamics on urban tree canopy in Bloomington, IN: Spatial analysis of sustainable urban forestry," PIs T.P. Evans, B.C. Fischer, S.K. Mincey, R. Thurau, **$18,700 awarded**.

Indiana Department of Natural Resources, Division of Forestry, Community and Urban Forestry-Putting Trees to Work Grant, 2009, "Urban Forestry Resources and Institutions (UFRI) Pilot Project," P.I.s B.C. Fischer and S.K. Mincey, **$3,000 awarded**.

USDA Forest Service, State and Private Forestry, Research Grant, 2009, "The Impacts of Redesign Competitive Funding," PIs B.C. Fischer, S.K. Mincey, S.V. Tomas, M.E. Cox, T. Ruseva, **$49,999 awarded.**

IU-School of Public and Environmental Affairs, Sustainability Research Grant, 2008, "Development of the Urban Forestry Resources and Institutions (UFRI) Program," PIs B.C. Fischer, S.K. Mincey, R. Thurau, **$10,000 awarded.**

Indiana Department of Natural Resources, Division of Forestry, Community and Urban Forestry-Putting Trees to Work Grant, 2007, "The Indiana University Woodland Campus Brochure," PIs B.C. Fischer and S.K. Mincey, **$4,410 awarded**.

Fellowships and Awards

◦ **Garden Club of America Zone VI Fellowship in Urban Forestry**, 2010, **$4,000**

◦ **IU-School of Public and Environmental Affairs, Service Corps Fellowship**, 2005-2007, tuition remission

◦ **Morehead State University, Presidential Scholarship**, 2000-2003, Full tuition remission and housing award

Community and University Service

◦ **Indiana University SPEA Teaching and Learning Faculty Group Steering Committee**, *2011-2012*

◦ **Indiana University SPEA Undergraduate Honors Thesis Advisor**, *Spring 2012*

◦ **Indiana University Tree Board**, Member, *2009-2012*

◦ **Association of SPEA PhD Students**, Student Conference Logistics Coordinator, *2009-2011*

◦ **Association of SPEA PhD Students**, Graduate Professional Student Liaison, *2009*; Secretary, *2010, 2011*

 ◦ **IU-B School of Public and Environmental Affairs publication** (Authors: Mincey, S.K. and B.C. Fischer) "The Woodland Campus: A historic walking tour of Indiana University-Bloomington." URL: www.indiana.edu/~spea/pubs/woodland.pdf. 2008

 ◦ **United Way of Monroe County**, Vision Council Member, *2006-2007, 2007-2008, 2008-2009*

 ◦ **Kentucky Heartwood**, Board Member, *2007-2012*, Forest-Watch Volunteer, *2002-2005*

 ◦ **Indiana University's Student Professional Enrichment Auction,** Co-Chair, *2006-2007*

Special Skills

 ◦ **Leadership:** Featured on Student Stories Project, Indiana University Alumni Association, 2007; URL: http://alumni.indiana.edu/profiles/students/mincey.shtml

 ◦ **Public Speaking:** Featured on the International Society of Arboriculture's ArborViews Podcasts. May 27, 2011, "Managing the Urban Forest as a Commons." URL: http://itunes.apple.com/us/podcast/arborviews/id356690928

Professional Memberships

Association for Environmental Studies and Sciences (AESS), American Association for the Advancement of Science (AAAS), Ecological Society of America (ESA), International Society for Arboriculture (ISA), Indiana Urban Forest Council (IUFC), American Forests.